Groundwater-connected systems:
A social-ecological framing, global data-driven applications, and sustainability implications

by

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A social-ecological framing, global data-driven applications, and sustainability implications

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Abstract

Groundwater systems and groundwater science are both at critical moments characterized by rapid change. Human activities continue to transform the land surface and climate, pump groundwater at rates beyond physically renewable limits, and attempt to govern and manage the resource. In recognition of this, groundwater science has broadened in recent decades to account for the interactions between people, ecosystems, Earth systems, and groundwater. Separately yet simultaneously, sustainability science has emerged as a problem-oriented field aimed at understanding interactions between social and natural systems within the contested and normative contexts of sustainability. In this dissertation, I integrate leading sustainability science concepts and methods with groundwater science and demonstrate the utility of this approach through global studies that combine large, multidimensional datasets with spatial data science methods. This work makes contributions under two overarching themes: to support a more comprehensive understanding of large-scale groundwater systems as social-ecological systems, and to explore possible uses of these insights in support of global groundwater sustainability.

The fundamental contribution of this study is the development of the *groundwater-connected systems* framing that provides a language, conceptual foundation, and pathway to consider groundwater systems as social-ecological systems (**Paper I**). This framing centers a relational understanding of groundwater where groundwater systems are explicitly considered on the basis of biophysical and socioeconomic system interactions rather than on the basis of the resource's hydrogeological characteristics or physiographic setting. I argue that this framing has useful implications across data collection, scientific investigations, education, governance, and management. The remainder of the dissertation begins to explore some of these opportunities through global-scale data-driven applications.

As all global analyses I conduct (**Papers III-VI**) are based on open-access datasets, I first perform a scoping review of the existing open data landscape to study groundwater systems as social-ecological systems (**Paper II**). Over 130 datasets are identified and reviewed, and 40 unique datasets are used to generate findings across Papers III-VI.

I first apply the groundwater-connected systems framing to develop a global classification and mapping of groundwater's large-scale (order of 10^4 km^2) biophysical and socioeconomic functions (**Paper III**). The resulting *groundwaterscapes* (n = 15) are landscape units that represent specific and broadly occurring configurations of groundwater functions across Earth systems, ecosystems, food systems, and water management systems. The groundwaterscapes are derived using an iterative, two-stage self-organizing map clustering algorithm. Groundwaterscapes contrast with existing groundwater resource maps as all large aquifer systems of the world are characterized by multiple groundwaterscapes. Thus, groundwaterscapes offer a new lens and spatial tool to study groundwater dynamics, inform groundwater data collection priorities, and manage groundwater resources based on an understanding of groundwater systems as social-ecological systems.

I subsequently investigate the groundwater sustainability implications of the groundwaterscapes. I do so by developing a complementary global classification of groundwater system risk types, informed by an Anthropocene risk framing. This approach includes both conventional risks such as groundwater storage loss and land use change in addition to unconventional and increasingly prioritized risks such as gender development inequalities and hydro-political tension. Overlaying groundwaterscapes with groundwater risk types generates a spatially explicit mapping of hundreds (n = 270) of unique groundwater sustainability challenges, providing the most comprehensive social-ecological evaluation of the global groundwater crisis to date (**Paper IV**). Conceptualizing and mapping the global groundwater crisis in this way provides a tool to support solution transfer and network development between regions.

I conclude by conducting two studies that demonstrate the broad applicability of the groundwater-connected systems framing in contexts where groundwater considerations are often overlooked or omitted. In **Paper V**, I assess the social-ecological vulnerability of river basins to experience impacts from the linked threats of freshwater stress and freshwater storage loss, which embeds representation of groundwater storage trends, and identify a set of most vulnerable basins as hotspots for global prioritization. In **Paper VI**, I delineate the groundwater catchments of the world's protected areas to highlight how groundwater flow can transmit human impacts occurring outside protected areas to ecosystems within protected area boundaries.

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Papers included in this dissertation

I. **Groundwater connections and sustainability in social-ecological systems.** (2023) Huggins, X., Gleeson, T., Castilla-Rho, J., Holley, C., Re, V., & Famiglietti, J.S. *Groundwater, 61*(4), 463 – 478. https://doi.org/10.1111/gwat.13305

II. **The open data landscape to study groundwater dynamics in social-ecological systems: A scoping collection and review of global datasets and an aspirational future outlook.** Huggins, X., Gleeson, T., & Famiglietti, J.S. *Manuscript*.

III. ***Groundwaterscapes*: A global classification and mapping of groundwater's large scale socioeconomic, ecological, and Earth system functions.** (in review). Huggins, X., Gleeson, T., Villholth, K.G., Rocha, J.C., & Famiglietti, J.S. *Water Resources Research.* Preprint: https://doi.org/10.31223/X5M382

IV. **Global groundwaterscape risks comprehensively map groundwater sustainability challenges.** Huggins, X., Gleeson, T., Moore, M.-L., & Famiglietti, J.S. *Manuscript*.

V. **Hotspots for social and ecological impacts from freshwater stress and storage loss.** (2022). Huggins, X., Gleeson, T., Kummu, M., Zipper, S.C., Wada, Y., Troy, T.J., & Famiglietti, J.S. *Nature Communications, 13*, 439. https://doi.org/10.1038/s41467-022-28029-w

VI. **Overlooked risks and opportunities in groundwatersheds of the world's protected areas.** (2023). Huggins, X., Gleeson, T., Serrano, D., Zipper, S., Jehn, F., Rohde, M.M., Abell, R., Vigerstol, K., & Hartmann, A. *Nature Sustainability, 6*, 855 – 864. https://doi.org/10.1038/s41893-023-01086-9

Note on authorship:

Xander Huggins (X.H.) is the primary author of this dissertation and all enclosed documents. Paper specific author contribution statements are included in each paper's respective chapter. A summary of co-author involvement in each paper in summarized below in an abridged Contribution Roles Taxonomy (CRediT).

Summary of contributions

■ >90% X.H. contribution
⧹ Notable shared credit with co-authors (50 – 90% X.H. contribution)
● Predominantly co-author led contribution (<50% X.H. contribution)

	Conceptualization	Methodology	Data curation	Analysis	Software/ programming	Visualization	Writing – original draft	Writing – review and editing
Paper I	■	N/A	N/A	N/A	N/A	■	■	⧹
Paper II	■	■	■	■	■	■	■	⧹
Paper III	■	■	■	■	■	■	■	⧹
Paper IV	■	■	■	■	■	■	■	⧹
Paper V	■	■	■	■	■	■	■	⧹
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Publications outside the dissertation

VII. **Groundwater-dependent ecosystem map exposes global dryland protection needs**. (2024). Rohde, M.M., Albano, C.M., Huggins, X., Klausmeyer, K.R., Morton, C., Sharman, A., Zaveri, E., Saito, L., Freed, Z., Howard, J.K., Job, N., Richter, H., Toderich, K., Rodella, A.-S., Gleeson, T., Huntington, J., Chandanpurkar, H.A., Purdy, A.J., Famiglietti, J.S., Singer, M.B., Roberts, D.A., Caylor, K. & Stella, J.C. *Nature*, *632,* 101–107. https://doi.org/10.1038/s41586-024-07702-8

VIII. **Chapter 6: Groundwater and Ecosystems**. (2022). Gleeson, T., Huggins, X., Connor, R., Arrojo-Agudo, P., & Vázquez Suñé, E. In *UNESCO Water Development Report 2022: "Groundwater: Making the invisible visible"*. Available online at: https://unesdoc.unesco.org/ark:/48223/pf0000380742.locale=en

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XII. **The potential of Hydrogeodesy to address water-related problems and sustainability challenges**. (in review). Jaramillo, F., Aminjafari, S., Castellazzy, P., Fleischmann, A., Fluet-Chouinard, E., Hashemi, H., Martens, H., Papa, F., Schöne, T., Tarpanelli, A., Virkki, V., Wang-Erlandsson, L., Abarca del Río, R., Borsa, A., Destouni, G., Di Baldassere, G., Moore, M.-L., Posada- Marín, J., Wdowinski, S., Allen, G., Argus, D., Elmi, O., Fenoglio-Marc, L., Frappart, F., Huggins, X., Kalantari, Z., Munier, S., Palomino-Ángel, S., Robinson, A., Rubiano, K., Simard, M., Song, C., Spence, C., Tourian, M., Wada, Y., Wang, C., Wang, J., Berghuis, W., Crétaux, J.-F., Famiglietti, J., Fassoni-Andrade, A., Fayne, J., Girard, F., Kummu, M., Larson, K., Eguivar, J.M.M., Moreira, D., Pavelsky, T., Reager, J., Rulli, M.C., Salazar, J., Nilsen, K., Peña, F., Siles, G., Yao, F., & Hübinger, C. Preprint: https://doi.org/10.22541/au.170379692.29590839/v1

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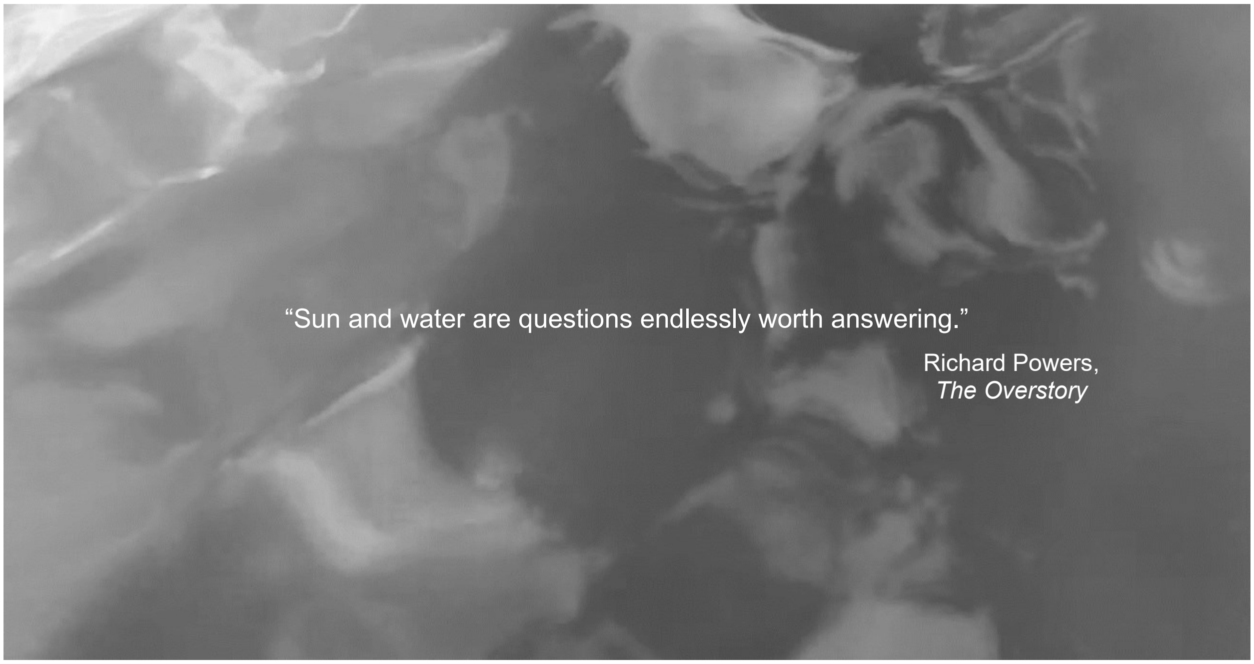
Dedication

to my adoring and adored parents

&

for sharing in joy with me so often



“Sun and water are questions endlessly worth answering.”

Richard Powers,
The Overstory

Chapter 1

INTRODUCTION

Groundwater is an enigmatic resource characterized by contradictions. Residing underground and out of sight, groundwater is often called "the invisible resource" (United Nations, 2022) yet its presence is made visible through the ecosystems, streams, agriculture, households, cities, cultures, and industries that depend on the resource. Groundwater is a characteristically local resource as substantial political, social, legal, and hydrological heterogeneity can be found within individual aquifers (Foster et al., 2013). Yet, it is globally distributed and serves crucial global-scale functions such as storing the largest volume of liquid freshwater and regulating hydroclimatic and hydroecological systems (Gleeson et al., 2020). Groundwater systems are characteristically slow, with typical response times that far exceed human timescales (Rousseau-Gueutin et al., 2013), yet the rapid expansion and intensification of groundwater use in the Anthropocene (Abbott et al., 2019; Bierkens & Wada, 2019) is driving dramatic groundwater changes that are perceptible over increasingly short and alarming periods (Jasechko et al., 2024). That groundwater is both local *and* global, invisible *and* ubiquitous, slow *and* rapidly changing makes the resource a particularly unique, counterintuitive, and complicated common-pool resource. In this dissertation, I work along a similar counterintuition to study groundwater systems by centering my approach not on groundwater itself but rather on the interactions between groundwater and its connected socioeconomic, ecological, and Earth systems.

Common across natural resources is an intertwinement and interplay between science and an encompassing sustainability discourse (Montanari et al., 2013). For instance, improved understanding of aquifer-stream interactions is useful not only to advance hydrological science but also to support improved conjunctive water management (Zipper et al., 2024). This is certainly the case for groundwater, which faces a global and intensifying crisis (Famiglietti, 2014; Famiglietti & Ferguson, 2021). Over half of the major aquifer systems of the world are in states of depletion (Richey et al., 2015), approximately half of the world's dryland groundwater-dependent ecosystems are in regions experiencing groundwater storage loss (Rohde et al., 2024), and groundwater pumping may transgress environmental flow limits in up to 80% of watersheds by the middle of the century (de Graaf et al., 2019). These examples highlight some of the quantifiable impacts of the groundwater crisis, yet there are innumerous, less tangible or more qualitative impacts such as on the relational values, cultural practices, senses of place, and identities (Tapsuwan et al., 2011) emanating from or related to groundwater. It is a call to action in itself that the severity with which we recognize the global groundwater crisis is informed from a partial, ‘tip of the iceberg’ understanding of these impacts.

It is a truism that both science and action are needed to address this crisis, yet doing so is challenging in both theory and practice. There exists a wide array of physical groundwater sustainability metrics (see review in Gleeson et al., 2020), providing a spectrum of approaches which are further complicated by persistent, over-simplified concepts of sustainable groundwater use (Bredehoeft, 2002). Indeed, a diversity of conflicting perspectives have been articulated using the language of groundwater sustainability, that range from solely focusing on physical groundwater systems to discipline-spanning approaches that consider linkages with the biosphere and society (Alley & Leake, 2004). This groundwater sustainability discourse traces the tensions and debates found in the general sustainability science literature, such as the divide between strong and weak sustainability (Ayres et al., 2001) which hinge on the substitutability or dependence of social, economic, and environmental dimensions and contributions to well-being (Folke et al., 2016; Purvis et al., 2019). Thus, engaging with sustainability literature is both an opportunity to apply new concepts and methods to the groundwater domain and simultaneously a challenge that requires careful navigation as the field is diverse and debate-filled, and thus brings sizable ‘conceptual baggage’ with it.

In practice, regulatory approaches have widely failed to curb groundwater depletion (Molle & Closas, 2020) and some have even intensified the problem (Balasubramanya et al., 2024). In global sustainability initiatives, such as the United Nations Sustainable Development Goals or the planetary boundaries framework, groundwater is often omitted, underrepresented or misrepresented (Gleeson, 2020; Guppy et al., 2018). Grassroots initiatives have called for more urgency and mobilization on groundwater sustainability (Global Groundwater Statement, 2019) but little evidence of concrete mobilization can be found. The scale and magnitude of human impacts on the global hydrological cycle (Konikow & Kendy, 2005; Kuang et al., 2024), in addition to the lack of applied successes in confronting groundwater challenges, collectively demand new scientific approaches dedicated to conceptualizing and facilitating sustainability in these systems (Di Baldassarre et al., 2019).

In my personal reading of the groundwater sustainability literature, and for the purpose of situating the research included herein, I have found it useful to consider groundwater science and groundwater sustainability science as interrelated but unique pursuits. I have drawn this distinction largely following that which is commonly drawn between the general forms of science and sustainability science. Doing so has provided clarifying oversight regarding the language, framing, goals, and intended audience of a given study. In this typology, I understand groundwater science as investigations that operate primarily in the language of hydrogeology and natural sciences, are framed around physical groundwater systems, and aim to reveal insights about groundwater resources, dynamics, and underlying mechanisms, and are written for disciplinary audiences (i.e., the audience largely reflects the authorship). Conversely, I understand groundwater sustainability science as the application of concepts and methods from the emerging field of sustainability science to groundwater topics. The language of groundwater sustainability science more closely mirrors the interdisciplinary and system-agnostic language of sustainability science, is framed around an embedded understanding of natural resources as interconnected with biophysical, social, and economic systems, is explicitly problem-oriented, and is written for the diverse audiences that shape sustainability outcomes (i.e., the audience is broader than the authorship). While this distinction is fuzzy in application, the body of work I present in this dissertation can be broadly interpreted as an effort to develop one possible approach to conduct groundwater sustainability science.

There is a substantial and growing groundwater science literature focused on system interactions between groundwater and Earth systems, ecosystems, and society. This literature is primarily populated by the disciplines of socio-hydrogeology (Re, 2015), hydro-social systems (Linton, 2010; Wesselink et al., 2017), and eco-hydrogeology (Cantonati et al., 2020), and recognized by frameworks and concepts such as the food-energy-water nexus (D'Odorico et al., 2018), integrated water resources management (Global Water Partnership, 2000), and the One Water paradigm (Villholth, 2021). These research communities have established a wide, baseline understanding of the system dynamics connecting groundwater with humans, ecosystems, and the Earth system and have generated a substantial volume of data in this process. The recognition that these pairwise system interactions do not occur in isolated contexts and that significant work remains to integrate insights from these research directions into a cohesive, system-of-systems understanding of groundwater represents a foundational motivation of this dissertation.

Sustainability science, which has matured substantially since its modern foundation was laid in the Brundtland Commission Report's definition of sustainable development (Bettencourt & Kaur, 2011; WCED, 1987), is a characteristically undisciplinary and problem-oriented domain (Kates, 2011; Haider et al., 2018). The application of sustainability science to groundwater presents several opportunities and potential benefits. The most obvious opportunity is that the groundwater crisis demands a direct engagement with the language and approaches of leading sustainability science. Groundwater has not been at the forefront of the sustainability science literature in comparison to land systems, forests, and fisheries; and these domains provide a fertile roadmap for groundwater to adapt and follow. A second opportunity concerns addressing needed methodological innovation in groundwater science, which has been provocatively described as a "zombie science" (Schwartz, 2013) for a reliance on dominant conventions and a pattern of revisiting "old" research questions ("How much more analytical well hydraulics do we need when most every solution likely to be used was finished by 1970?"; Schwartz, 2013). If this stagnancy in groundwater science and the severity of the groundwater crisis are at all related, innovating within the science should be treated as not only a curiosity but equally as an obligation. In this dissertation, I look for such innovation by drawing on the undisciplinary and diverse methods of sustainability science, which engender multiple worldviews, mental models, and system boundaries in comparison to groundwater hydrology. Given the 'fuzziness' and intertwinement between science and sustainability science overviewed above, I foresee the potential for reciprocity between groundwater science and groundwater sustainability science, where insights from groundwater science can have useful sustainability implications while applications of groundwater sustainability science may also unlock perspectives, lines of inquiry, and methods that support novel groundwater science insights.

This dissertation is broadly motivated to meet this present and pivotal moment for groundwater science and sustainability, characterized by the intersecting crises presented above: a crisis in the resource (i.e., the global groundwater crisis); and a crisis in the science (i.e., groundwater hydrology as a zombie science). This moment is further defined by the opportunities presented in the emergence of sustainability science as a robust field with overdue application to groundwater, by the rapid growth of global data documenting groundwater processes and interactions across a wide array of biophysical and socioeconomic systems, and in the development of data science methods to process and 'make sense' of large, multidimensional, and interdisciplinary datasets.

My overarching intent is to develop an approach to groundwater sustainability science that operates within the overlapping opportunity space sketched above, and to demonstrate the approach through global-scale data-driven applications. I draw from the core fields of physical groundwater science, data science, spatial science, and sustainability science with a special focus on data-driven assessments of novel conceptual models of groundwater systems as social-ecological systems. This work follows a tradition of scholarship that recognizes humans, ecosystems, and the water cycle as intertwined systems (e.g., Falkenmark, 1977), however it remains that groundwater has been underrepresented in these efforts and in the broader social-ecological system literature (Kajikawa et al., 2014).

Social-ecological systems, known also as coupled human-environmental systems, are a central and defining concept of sustainability science (Berkes & Folke, 1998; Clark & Harley, 2020). Social-ecological systems are defined as integrated social and biophysical systems that exhibit complex adaptive system behaviours such as thresholds, multi-scalar dynamics, multiple stable states, and feedback mechanisms (Preiser et al., 2018). My work to align groundwater science and groundwater sustainability with sustainability science rests principally on a translation of social-ecological system concepts, frameworks, and perspectives to groundwater. Though there are multiple approaches and competing terminologies to describe human-environmental system interactions, I opt to use the term ‘social-ecological systems’ as it is associated with an ecosystem of frameworks (Binder et al., 2013) which further guide language usage and conceptual model development.

I opt to work at the global scale for several reasons. The first is that global-scale analysis accommodates a greater degree of system abstraction and generalization in comparison to place-based investigations, and that I view as more compatible with early stages of conceptual development. For instance, groundwater process representation in global hydrological models is considerably more rudimentary in contrast to local-scale and catchment-specific models (Condon et al., 2021; Gleeson et al., 2021). I argue the same pattern holds for groundwater when studied as a social-ecological system. Furthermore, global analyses and maps are powerful education instruments, which can serve an important role in facilitating uptake of the concepts developed in this dissertation within the broader groundwater community. Thus, I view exploratory, global-scale case studies as a suitable approach to begin uncovering the nascent potential of studying groundwater systems as social-ecological systems. Gleeson (2020) argues six additional reasons why global perspectives are important for groundwater, which are paraphrased below, and that simultaneously act as motivations for the global-scale analyses presented in this dissertation.

- Reason 1: To inform water governance and management in transboundary aquifers in an increasingly globalized world.
- Reason 2: To systematically analyze problems and solutions to enable regional prioritization and solution transfer.
- Reason 3: To create visualizations to improve understanding and appreciation of groundwater.
- Reason 4: To emphasize the importance of groundwater in global sustainability frameworks.
- Reason 5: To quantify interactions between groundwater and other components of the Earth system.
- Reason 6: To build global networks of interdisciplinary experts.

My lines of inquiry and research outcomes build first on and then out from each other (Figure 1.1). I begin by developing an overarching framing to approach the study of groundwater systems as social-ecological systems, and later chapters investigate specific questions (such as how can the groundwater catchments of the world's protected areas be derived, and what is their protected status?) to demonstrate the utility of taking a social-ecological systems approach to global groundwater sustainability challenges. I investigate six research questions whose interrelationships are illustrated in Figure 1.1. As sustainability science is an inexact science (cf. Helmer & Rescher, 1959) that integrates the natural and social sciences, developing a strict, formal method is not a realistic or intended pursuit of this work. Rather, I seek to build a robust foundation for a potential form or theme of *groundwater sustainability science* that is characterized by the intersection of geospatial data science methods, social-ecological systems frameworks, and sustainability science foundations. This dissertation primarily makes contributions within the first and second stages of discipline development (Shneider, 2009) by introducing the subject matter, framing, and language for this new approach (first-stage contributions) and identifying and testing a toolbox of methods (second-stage contributions) through preliminary, global-scale investigations.

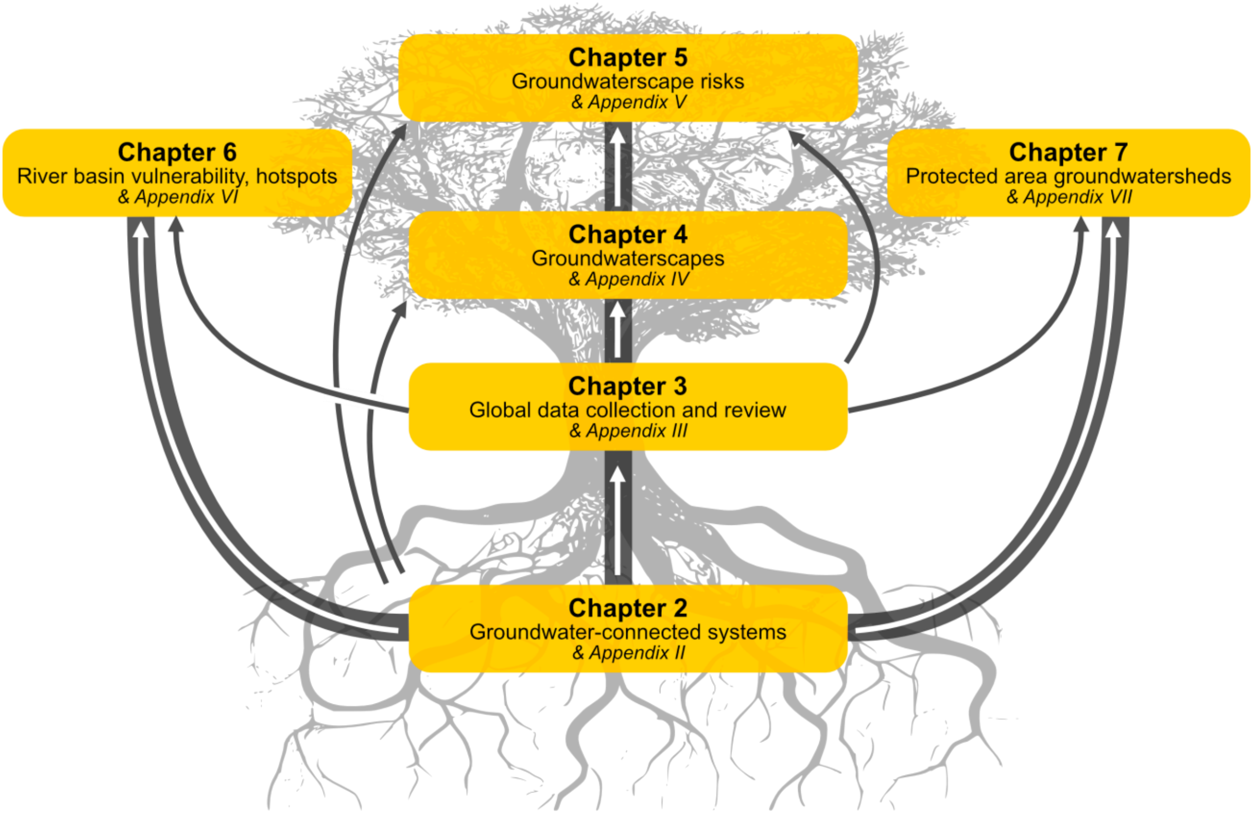


Figure 1.1. Relationships between research chapters.

This dissertation is comprised of a central column of research chapters (Chapters 2-5), and two extended research chapters (Chapters 6 and 7). Chapter 2 serves as the conceptual foundation which is directly built on in a data review (Chapter 3), and two explicit data-driven applications (Chapters 4 and 5). The extended research chapters ‘branch out’ from this central column by investigating research questions related to groundwater systems as social-ecological systems but do not continue to linearly build on the outcomes from the central column chapters.

Chapter-specific research questions and brief overviews of chapter contents and research question motivation are listed below.

Research Question 1: How can groundwater systems be conceptualized and studied as social-ecological systems? (Chapter 2)

> As the groundwater literature grows and becomes increasingly interdisciplinary, it is evident that groundwater systems are deeply intertwined with social, ecological, and Earth systems. Yet, there remains no dedicated, overarching set of guidelines or framing to support the conceptualization of groundwater systems as social-ecological systems. This

chapter reviews relevant sustainability science and groundwater sustainability literature, interprets their relevance and interconnections, and develops the *groundwater-connected systems* framing. This work then discusses the potential implications of the developed groundwater-connected systems framing for data collection, scientific investigations, governance, management, and training of groundwater scientists and professionals.

Research Question 2: What global-scale data are available to study groundwater systems as social-ecological systems? What strengths, omissions, and biases exist in these data? What forms of data collection should be prioritized to strengthen global assessments of groundwater systems as social-ecological systems? (Chapter 3)

Alongside the global-scale groundwater literature, the volume and diversity of data documenting groundwater system attributes, functions, and system interactions has grown considerably in recent decades. Summarizing this data landscape is both useful in its ability to facilitate analyses of groundwater-connected systems and to inform and motivate future data collection efforts to address current deficiencies. This chapter reviews an assembled collection of datasets using categories that consider dataset treatment of time (static, time series, or historical records), explicitness of groundwater representation or consideration (direct, explicit, implicit), and the primary system the data represents (hydrosphere, atmosphere, lithosphere, biosphere, governance, food systems, and other socioeconomic systems). Understanding that data generation is simultaneously a technical and social process (i.e., data do not exist independently of the scientists and institutions developing the data), datasets are further analyzed to identify geographic bias in the institutions generating these data. This work provides a foundational reference on existing global groundwater data and discusses possible pathways for future global groundwater data initiatives.

Research Question 3: What are the predominant forms of groundwater systems when conceptualized as social-ecological systems? How do these forms, and their spatial patterns, compare to existing global classification systems of groundwater resources? And what are the groundwater science and groundwater sustainability implications of generating a global social-ecological typology of groundwater systems? (Chapter 4)

This chapter builds on the groundwater-connected systems framing presented in Chapter 2 and applies a subset of data summarized in Chapter 3 to develop a novel landscape unit classification of global groundwater systems based on groundwater's large-scale (order of ~10^4 km^2) Earth system, ecosystem, food system, and water management system functions. This study uses leading, previously published, geospatial datasets documenting these functions and applies a sequential self-organizing map (SOM) algorithm to derive the system classification. The identified system classes, that I call *groundwaterscapes*, capture the dominant spatial patterns in groundwater's social-ecological functions. These groundwaterscapes support a context-rich understanding of global groundwater systems and can support (1) process-based insights on the cooccurrence and linkages between groundwater systems functions, and (2) groundwater sustainability strategies by bridging global and regional scales by facilitating consideration of regional context in global initiatives. Landscape metrics are calculated for groundwaterscape distributions within the large aquifer systems of the world. The distribution of monitoring wells in the global groundwater monitoring network (GGMN) is evaluated across groundwaterscapes to assess the current ability to observe groundwater behaviour across these characteristically unique social-ecological systems.

Research Question 4: How are global groundwater sustainability challenges distributed across groundwaterscapes? How do configurations of sustainability challenges and system functions interrelate and enhance problem definition for the global groundwater crisis? (Chapter 5)

This chapter directly builds on the groundwaterscapes developed in Chapter 4 by deriving a complementary global classification of groundwater system risks. Groundwater sustainability challenges are identified by interpreting Anthropocene risk (cf. Keys et al., 2019) through a groundwater-connected system lens. This process generates a set of risks that relate to Earth system change such as groundwater depletion, land use change, change in heavy rainfall, land subsidence, and agricultural intensification, and risks that relate to inequality emergence such as unmet conservation needs, hydro-political conflict, gender development inequality, and water crowding. The pairing of these Anthropocene risk patterns with groundwaterscapes generates a first social-ecological global mapping of the global groundwater sustainability problem space. Unique combinations of groundwaterscapes and groundwater system risk types, which I call *groundwaterscape risks,* offer a data-driven tool to identify groundwater sustainability challenges and challenge similarity across the global domain. These groundwaterscape risks provide a baseline mapping to consult when considering solution transfer between regions and

> support more explicit consideration of social-ecological context in the development of groundwater sustainability strategies. Furthermore, the groundwaterscape risks work to pluralise approaches to conceptualize the global groundwater crisis by generating data-driven regional narratives that contest sweeping narratives that are prevalent in the global groundwater sustainability discourse.

Research Question 5: How do trends in groundwater storage, alongside other components of terrestrial water storage and in conjunction with freshwater stress, contribute to social-ecological system vulnerability at the river basin scale? (Chapter 6)

Research Question 6: How can groundwater catchments be derived for the world's protected areas? What is the spatial extent and protected status of these groundwater catchments? How can groundwater catchment delineation support more robust area-based conservation? (Chapter 7)

> These final two final research questions represent broader applications of the groundwater-connected systems framing. Through their inclusion, these research questions demonstrate the ability of the framing to support and strengthen research questions on topics where groundwater considerations are typically omitted or underrepresented. In these studies, the groundwater-connected systems framing guides the inclusion of groundwater in conceptual models built around specific, targeted research questions. (This is in contrast to Chapters 4 and 5, which develop and apply conceptual models explicitly built on the groundwater-connected systems framing.) Two research studies are presented: a global assessment of patterns in social and ecological vulnerability to the linked hydrological threats of freshwater stress and freshwater storage loss, which encompass trends in groundwater storage (Chapter 6), and a study on the protected status of the groundwater catchments of the world's protected areas that contain groundwater-dependent ecosystems (Chapter 7).

I hope to make the case that the conceptual foundation laid in Chapter 2 and built on throughout the rest of the dissertation is wide and fertile. This dissertation does not exhaust the potential uses of the groundwater-connected systems framing. Rather, my hope for this dissertation is to establish a foundation of groundwater-connected system scholarship and enable a generative future for its research directions.

Lastly, I find it important to draw attention to the relevance of this dissertation with respect to civil engineering and to the role of engineering in shaping this dissertation. Sustainability challenges, such as the global groundwater crisis, are characteristically different from conventional engineering problems in that they operate across scales, present as not only technical problems but *wicked* challenges (with contested goals, underspecified boundary conditions and no clear stopping criterion; Lönngren & van Poeck, 2021; and that require approaches that transcend traditional disciplines; Jerneck et al., 2011). On this basis, engineering and sustainability science can be viewed as interdependent competencies, where engineering expertise is necessary but insufficient to realise sustainability goals while sustainability science concepts are necessary tools to inform engineering practice and design. Engineering bodies in Canada recognize this growing interdependence. For instance, Engineers Canada (2016) argues “the implications of sustainability for engineers are major” and notes the shifting role of the engineer: “engineers must become more effective at identifying real needs … [that] will require them to become problem framers so they can decide on the most effective directions for technology to take”. Simultaneously, Engineers and Geoscientists British Columbia (EGBC)’s Code of Ethics is clear that the engineer’s duty includes a role to promote sustainability through engineering practice by “safeguard[ing] human life and welfare and the environment.”

Yet, integrating sustainability science with engineering is not trivial and doing so casts light on differences in research traditions and competencies and that can be challenging to navigate. For instance, natural and social science data are often incompatible (Strang, 2009) and different epistemologies characterize these research fields. Positivist philosophies tend to dominate natural science and engineering disciplines (Baillie & Douglas, 2014), which view knowledge as singular and obtainable through objective and impartial means (Willig, 2008). Conversely, constructionist philosophies, which dominate social and sustainability sciences, recognize the interplay between scientist and study object and understand knowledge claims (where there are multiple “knowledges” rather than one “knowledge”) as positioned by factors including social location and educational training (Baillie & Douglas, 2014; Cockburn, 2022; Moon & Blackman, 2014). These differences are not superficial and have deep implications on how science is conducted, including on what constitutes as robust scientific assessment, on how research questions are formulated and accepted by research communities, and on how research outputs are reported. Thus, one reading of this dissertation may be as a case study on the process of engagement with sustainability science and its associated methodologies from the perspective of a researcher (myself) trained in water resources engineering. Let us return to this dynamic in the conclusion (Chapter 8) to reflect on the insights and lessons learned in this regard through the doing of this research.

There are indeed many demands placed on the engineering profession related to the grand challenge of global groundwater sustainability. These range from designing satellites and other measurement instruments to monitor trends in groundwater quantity and quality, to designing managed aquifer recharge infrastructure, geotechnical work to limit impacts of land subsidence, saltwater desalination technologies, and hydrological modelling (UNESCO & ICEE, 2021). The research I conduct in this dissertation can be understood as problem-oriented sustainability science with groundwater science implications. I argue that it simultaneously presents a case study of a crucial but underemphasized competency in engineering research: extensive, technical, and system-spanning problem definition and framing. Such discipline-spanning and technical assessments of the social-ecological context in which engineering designs operate and broader sustainability transformations situate are crucial areas for growth in the engineering profession in order for it to effectively address the challenges presented by global change. In many ways, this dissertation reflects the changing role and expertise required from engineers. My hope is that one final impact of this work is found through a demonstration of the need for, and the benefits incurred by, incorporating sustainability science into the toolkit of the 21st century engineer.

1.1 Dissertation organization

This dissertation is organized in eight chapters, including the introduction, six chapters presenting original research, and one final chapter synthesizing the contributions of this work. Each research chapter includes study-specific introductions and reviews of the relevant literature, so a separate chapter dedicated to literature review is not included to reduce repetition.

CHAPTER 2 presents the article “Groundwater connections and sustainability in social-ecological systems” that was published in the journal *Groundwater* in March 2023.

CHAPTER 3 presents the manuscript “The open data landscape to study groundwater dynamics in social-ecological systems: a scoping collection and review of global datasets and an aspirational future outlook”.

CHAPTER 4 presents the manuscript “*Groundwaterscapes*: a global classification and mapping of groundwater’s large-scale socioeconomic, ecological, and earth system functions” that is in review with the journal *Water Resources Research*.

CHAPTER 5 presents the manuscript “Groundwaterscape risks comprehensively map global groundwater sustainability challenges”.

CHAPTER 6 presents the article “Hotspots for social and ecological impacts from freshwater stress and storage loss” that was published in the journal *Nature Communications* in January 2022.

CHAPTER 7 presents the article “Overlooked risks and opportunities in groundwatersheds of the world’s protected areas” that was published in the journal *Nature Sustainability* in March 2023.

CHAPTER 8 presents conclusions and reflections.

1.2 References

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Chapter 2

GROUNDWATER CONNECTIONS AND SUSTAINABILITY IN SOCIAL-ECOLOGICAL SYSTEMS

This chapter was published as an *issue paper* in *Groundwater* in 2023.

Huggins, X., Gleeson, T., Castilla-Rho, J., Holley, C., Re, V. & Famiglietti, J.S. (2023). Groundwater connections and sustainability in social-ecological systems. *Groundwater*, *61*(4), 463-478.
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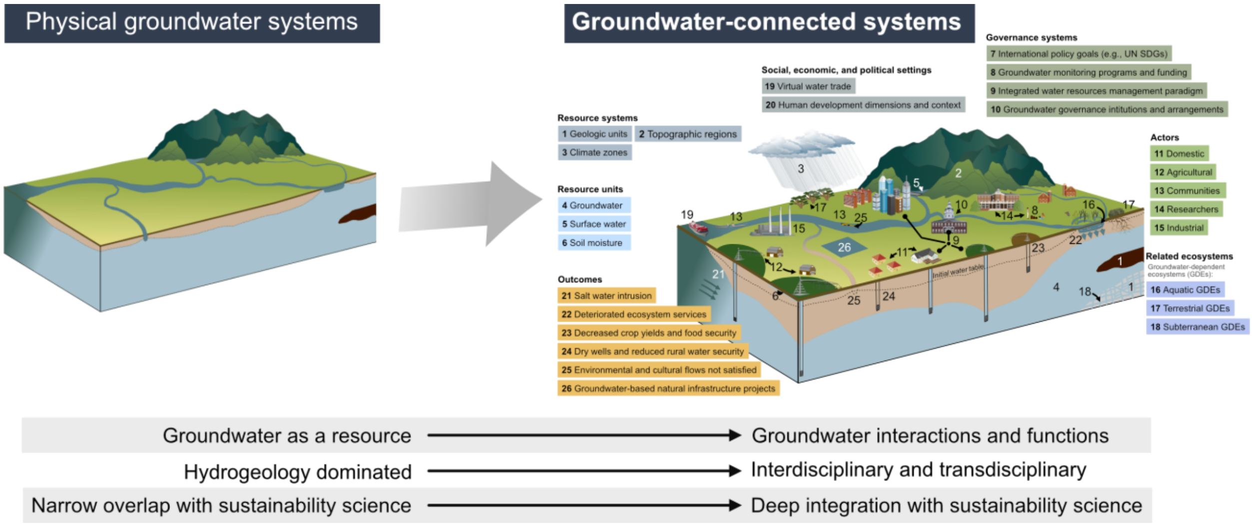
Key points:

- Presents the groundwater-connected systems framing to conceptualize groundwater systems as social-ecological systems.
- The framing redirects the central focus of groundwater research towards explicit consideration of the relationships between groundwater and social, economic, ecological, and Earth systems.
- The framing has potentially wide-ranging implications across data collection, scientific investigations, groundwater education, governance, and management.

Figure 2.1. Graphical abstract.



2.1 Abstract

Groundwater resources are connected with social, economic, ecological, and Earth systems. We introduce the framing of groundwater-connected systems to better represent the nature and complexity of these connections in data collection, scientific investigations, governance and management approaches, and groundwater education. Groundwater-connected systems are social, economic, ecological, and Earth systems that interact with groundwater, such as irrigated agriculture, groundwater-dependent ecosystems, and cultural relationships to groundwater expressions such as springs and rivers. Groundwater-connected systems form social-ecological systems with complex behaviors such as feedbacks, nonlinear processes, multiple stable system states, and path dependency. These complex behaviors are only visible through this integrated system framing and are not endogenous properties of physical groundwater systems. The framing is syncretic as it aims to provide a common conceptual foundation for the growing disciplines of socio-hydrogeology, eco-hydrogeology, groundwater governance, and hydro-social groundwater analysis. The framing also facilitates greater alignment between the groundwater sustainability discourse and emerging sustainability concepts and principles. Aligning with these concepts and principles presents groundwater sustainability as more than a physical state to be reached; and argues that place-based and multifaceted goals, values, justice, knowledge systems, governance, and management must continually be integrated to maintain groundwater's social, ecological, and Earth system functions. The groundwater-connected systems framing can underpin a broad, methodologically pluralistic, and community-driven new wave of data collection and analysis, research, governance, management, and education. These developments, together, can invigorate efforts to foster sustainable groundwater futures in the complex systems groundwater is embedded within.

2.2 Seeing groundwater through its connections

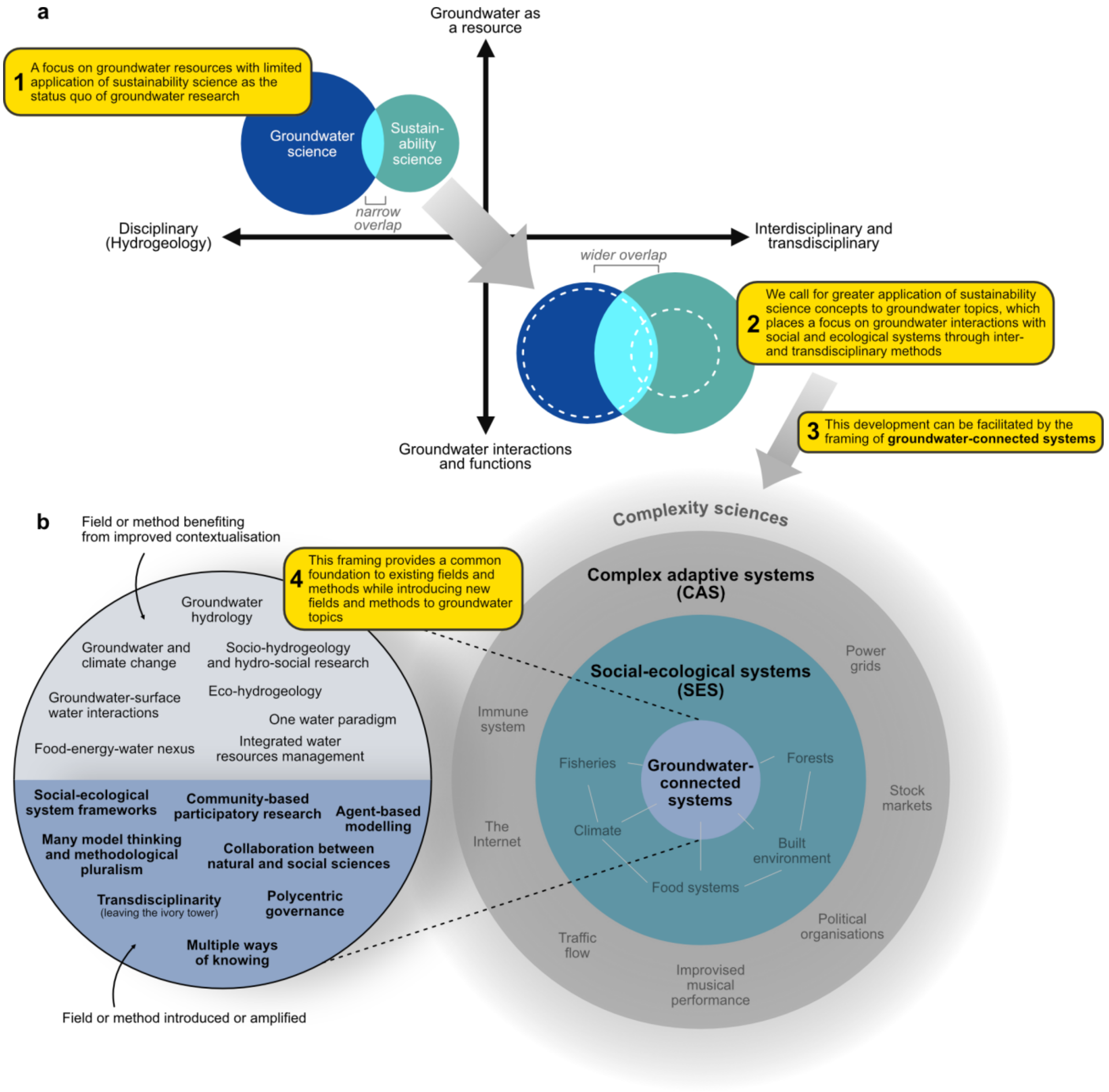
Groundwater is often described as a uniquely invisible, slow, and distributed resource (Villholth & Conti, 2018; Gleeson et al., 2020a). In this work, we seek to add a fourth quality to this description: groundwater as a connected resource. We make the case that a focus on groundwater's connections to social, economic, ecological, and Earth systems can generate novel insights, and more effective, socially relevant outcomes.

Groundwater is linked to many societal and environmental challenges and is a resource deeply embedded in a global crisis (Famiglietti, 2014). Yet, it is often under-prioritized or omitted in political and social agendas (Global Groundwater Statement, 2019). Simultaneously, there are calls for creativity and greater methodological experimentation in groundwater research (Schwartz, 2013). To what degree might a reliance on dominant conventions be linked or even contribute to the depleted and overlooked state of groundwater today? And, in what direction should groundwater practice and research expand to better address these intersecting challenges?

Amid calls for innovation in groundwater research, substantial progress has been made to document groundwater interactions and relationships in social, ecological, and Earth systems. This progress is found in the emerging disciplines of socio-hydrogeology (Re, 2015), eco-hydrogeology (Cantonati, et al. 2020), groundwater in Earth systems science (Gleeson et al., 2020b), and transdisciplinary methods (Zwarteveen et al., 2021); and in the more established social science domains of common-pool resource governance (Mukherji & Shah, 2005; Curtis et al., 2016) and analysis of hydro-social systems (Wesselink et al., 2017). The intricate nature and complexity of these interactions reveal the need to study, use, and manage groundwater resources on the basis of the functions and services that groundwater provides to systems that interact with it. Taking methodological and practical steps in this direction is necessary to ensure long-term sustainability and resilience in systems connected to groundwater.

We introduce a new framing for groundwater systems that we call groundwater-connected systems. The potential for this framing is two-fold. First, it can provide a common conceptual foundation for both traditional research programs and emerging, diverse research programs that document groundwater interactions with a broad and expanding set of systems. Second, it can facilitate the application of paradigms, methods, and theories from the field of sustainability science to groundwater topics that, in our view, have been underutilized to date.

This new framing supports the growth of groundwater research from a predominantly disciplinary pursuit—focused on groundwater as an isolated resource and one dominated by hydrogeologists' perspectives, methods, and paradigms—to an interdisciplinary pursuit focused on documenting groundwater interactions and relationships with social, ecological, and Earth systems through inter- and transdisciplinary methods and collaborations (Figure 2.2a).



1 A focus on groundwater resources with limited application of sustainability science as the status quo of groundwater research

2 We call for greater application of sustainability science concepts to groundwater topics, which places a focus on groundwater interactions with social and ecological systems through inter- and transdisciplinary methods

3 This development can be facilitated by the framing of **groundwater-connected systems**

4 This framing provides a common foundation to existing fields and methods while introducing new fields and methods to groundwater topics

Figure 2.2. Groundwater-connected systems as a framing for groundwater practice and research.

(a) We argue that groundwater investigations and assessments should increasingly move from disciplinary pursuits focusing on physical groundwater systems to inter- and transdisciplinary collaborations that focus on understanding groundwater interactions and functions in larger, connected systems. (b) This new framing is enabled by understanding groundwater-connected systems as social-ecological systems, which introduces new methods or amplifies existing methods for data collection, research, governance and management approaches, and education. To support the interpretation of this figure, consult the yellow text boxes in their numbered order.

There is a long history in the social sciences of documenting many of these interactions and dynamics (Ostrom, 1990). Yet, motivating this paper and the groundwater-connected systems framing are two notions. The first is that these foundational concepts and research questions remain largely unknown or rest in the peripheral awareness of many hydrogeologists, the dominant discipline in groundwater dialogs. A greater ability to engage in inter- and transdisciplinary discourse and science among hydrogeologists is needed for effective participation in applied groundwater studies and management initiatives. The second is that we perceive unfulfilled potential for social scientists to represent biophysical (e.g., hydrogeological, ecological, Earth system) dynamics with greater process specificity, and to operate at larger spatial scales of analysis, which are both needed to address a wider array of groundwater related interactions and challenges.

Our intention for the framing is to facilitate novel, methodologically pluralistic work on diverse groundwater topics to produce outputs more aligned with issues of ecological and societal concern. By making relationships between groundwater and social, economic, ecological, and Earth system processes better understood and more visible, our framing can help redress the often-overlooked nature of groundwater and elevate the relevance and prioritization of groundwater in social and policy discourses.

We begin by introducing our framing of “Groundwater-connected systems.” We then discuss the wider potential for sustainability science methods and concepts to be applied to groundwater sustainability topics in “Invigorating groundwater sustainability with sustainability science.” We end by providing a set of possible implications the framing can impart on data collection, scientific investigations, governance and management, and education in “Wide applicability to groundwater science and beyond.” Key terms are defined in Table 2.1.

Table 2.1. Summary of terminology used in this chapter.

Term	Definition	Core Properties	Key References (➜ Review Article)
Groundwater-connected system	A system that is formed between physical groundwater systems and any social, ecological, or Earth system(s)	Shared with social-ecological systems and complex adaptive systems	This work
Social-ecological system	An integrated system formed by interactions between social and biophysical systems	Social-ecological systems are forms of complex adaptive systems, with thresholds, multi-scalar dynamics, feedbacks, nonlinear processes, multiple stable states, time lags, and path dependency	Ostrom (1990) Berkes and Folke (1998) Ostrom (2009) ➜ de Vos et al. (2019)
Complex adaptive system	A system of interacting components which are "defined more by the interactions among their constituent components than by the components themselves" (Preiser et al., 2018)	Dynamic processes, relational networks, open systems, context-dependent behaviour, and emergent behaviour	Levin et al. (2013) ➜ Preiser et al. (2018)
Sustainability science	A science that focuses on the "interactions between natural and social systems, and with how those interactions affect the challenge of sustainability" (Kates, 2011)	Undisciplinary, problem oriented, complexity, collaborative institutions, multiple ways of knowing, no panaceas, and adaptation	Kates (2011) Jerneck et al. (2011) Loring (2020) ➜ Clark and Harley (2020)

Wicked problem	Problems that are not easily defined or solved due to their embeddedness in complex social contexts, having no single or straightforward solution	Unintended consequences, no clear stopping criterion, multiple, contradictory perspectives framing problem, and unclear definitions of ‘good’ or ‘bad’ outcomes	Rittel and Webber (1973) Crowley and Head (2017) ➔ Lönngren and van Poeck (2021)

2.3 Groundwater-connected systems

Here, we introduce the framing of groundwater-connected systems. Groundwater-connected systems are formed between physical groundwater systems and any social, ecological, or other biophysical system(s) that interacts with groundwater (Table 2.1). Thus, groundwater-connected systems take many forms. Groundwater-irrigated agriculture, domestic well owners' water security, groundwater institutions, management initiatives, and the cultural values associated with surface expressions of groundwater, such as river baseflow and springs, are a few human-oriented examples of groundwater-connected systems. Ecological and biophysical examples include terrestrial, aquatic, and subterranean groundwater-dependent ecosystems, groundwater-atmosphere process coupling, coastal ecosystems that rely on groundwater discharge, and groundwater-aquatic biodiversity relationships such as ecological responses to transgressed environmental flow requirements. Groundwater-connected systems are also the network of interactions between these often-intertwined systems.

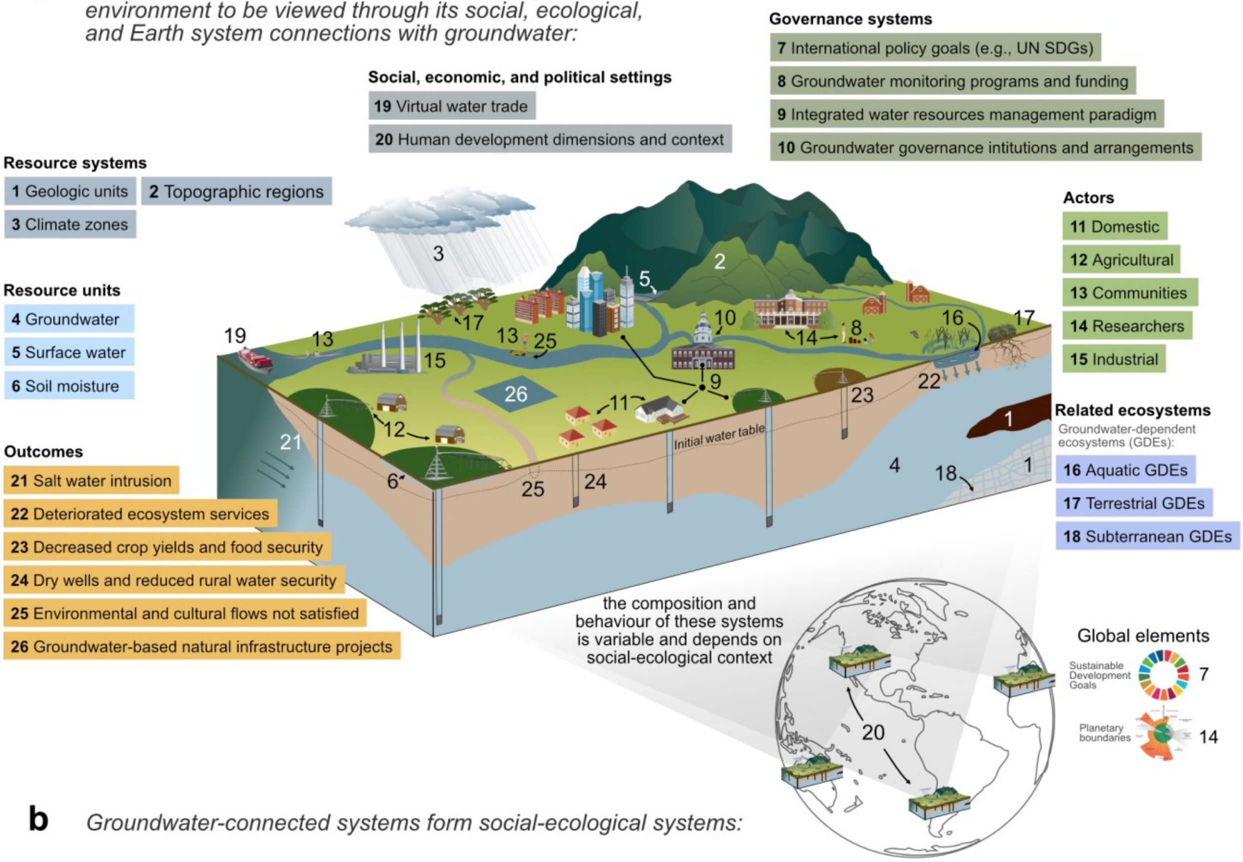
We understand groundwater-connected systems as forms of social-ecological systems (Figure 2.3). Social-ecological systems offer a way of viewing human-environmental system interactions as a single, interconnected system with physical, ecological, and social components (Berkes & Folke, 1998). Social-ecological systems are characterized by complex adaptive system behaviors (Levin et al., 2013; Preiser et al., 2018) such as thresholds, feedbacks, nonlinear processes, multiple stable system states, path- and context-dependent behavior, and emergent phenomena (Table 2.1). While physical groundwater systems are naturally dissipative and are themselves not social-ecological systems, these physical systems (i.e., aquifers) are components of social-ecological systems through their social, ecological, and biophysical interactions.

The groundwater-connected systems framing is flexible and does not provide an explicit or finite set of system interactions to study. Rather, the framing argues that a focus on relationships and interactions between groundwater and other systems offers critical insights that are unattainable when studying the resource in isolation.

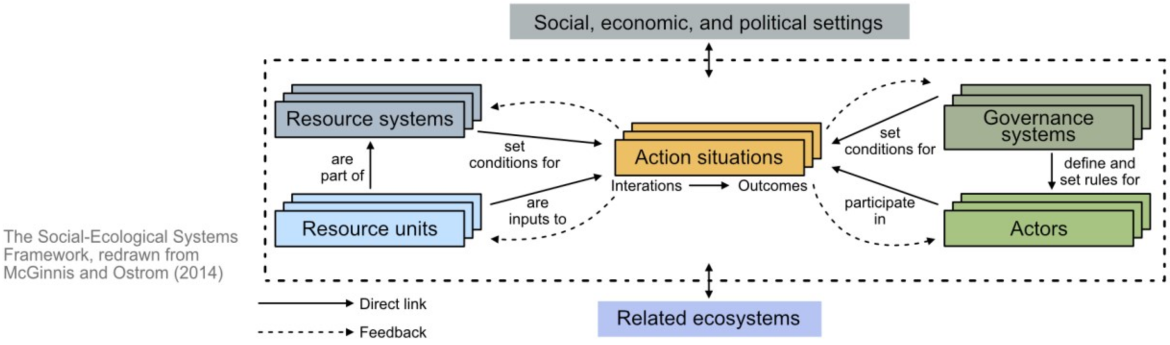
This focus on relationships rather than entities is consistent with motivations of the broader social-ecological systems literature (Reyers & Selomane, 2018). The subsetting of groundwater-connected systems, social-ecological systems, and complex adaptive systems (shown by the nested circles in Figure 2.2b) locates groundwater-connected systems research as a complexity discipline.

In Figure 2.3a, we present a conceptual diagram of groundwater-connected systems as social-ecological systems. For this illustration, we use the structure of the Social-Ecological Systems Framework (McGinnis & Ostrom, 2014; Figure 2.3b), the predominant framework used in the study of social-ecological systems (Partelow 2018). We associate features and processes of groundwater-connected systems to the generic structure of the Social-Ecological System Framework. These attributions are not comprehensive but provide evidence to support the view of groundwater-connected systems as social-ecological systems. For an extended description of Figure 2.3a, see Section II.1.

a *The groundwater-connected systems framing enables this environment to be viewed through its social, ecological, and Earth system connections with groundwater:*

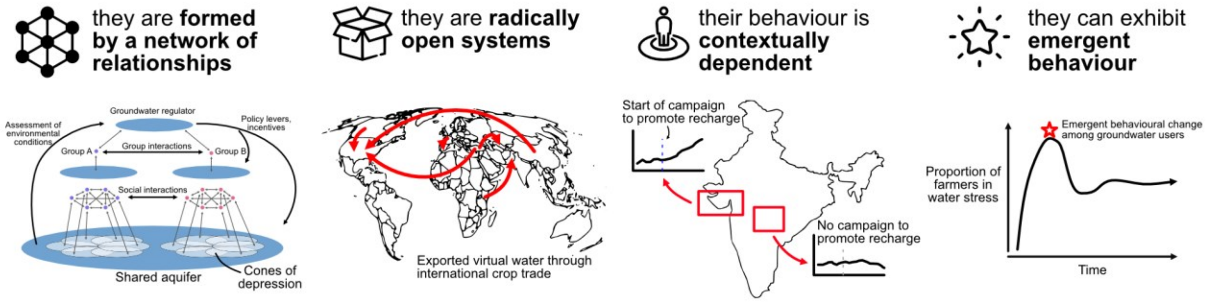


b *Groundwater-connected systems form social-ecological systems:*



The Social-Ecological Systems Framework, redrawn from McGinnis and Ostrom (2014)

c *Groundwater-connected systems behave as complex adaptive systems, with properties including:*



e.g., multi-scalar modes of interaction in managed groundwater systems (redrawn from Castilla-Rho et al. 2015)

e.g., groundwater depletion embedded in international food trade (Dalin et al. 2017)

e.g., social adoption of civil society and state efforts to promote groundwater recharge (Patel et al. 2020)

e.g., drawdown across farmers' wells across alternative regulation scenarios (Castilla-Rho et al. 2015)

Figure 2.3. Groundwater-connected systems are social-ecological systems.

(a) Mapping a regional environment's groundwater-connected systems to elements of the Social-Ecological Systems Framework (shown in b). (b) The Social-Ecological Systems Framework, redrawn from McGinnis and Ostrom (2014). (c) Properties of groundwater-connected systems that reflect how these systems behave as complex adaptive systems, with examples from Castilla-Rho et al. (2015), Dalin et al. (2017), and Patel et al. (2020).

Interactions and feedbacks in social-ecological systems occur across multiple space and time scales (Chapin et al., 2009). The relationship between international food trade, groundwater depletion, and environmental flows represents one example of cross-scale interactions in groundwater-connected systems. International food trade networks drive groundwater depletion (Dalin et al., 2017) that manifests as local to regional scale drawdown of the water table. Falling water tables can subsequently have cascading impacts on aquatic ecosystems that depend on groundwater discharge. For example, environmental flow transgressions driven by reduced groundwater discharge can lead to reach-scale impacts on fish populations, aquatic ecologies, and riparian vegetation (Gleeson & Richter, 2018). Thus, social-ecological system analysis attempts to understand how outcomes emerge through biophysical and social interactions, which often embody properties of complex adaptive systems (Figure 2.3c). For instance, groundwater-pumping-induced land subsidence can irreversibly change aquifer storage capacity, reducing the ability of groundwater to act as a buffer in times of drought which can decrease agricultural productivity and force shifts to alternative land uses (Dinar et al., 2021). These dynamics offer examples of thresholds, feedback mechanisms, path-dependent behavior and regime shifts common to complex adaptive systems. See Table II.1 for more information on complex adaptive system properties and behaviors of groundwater-connected systems.

While many of these interactions and outcomes remain undocumented, excluded, or under-analyzed, a growing body of literature across the natural and social sciences is beginning to examine the complex characteristics, processes, and outcomes of groundwater interactions in social-ecological systems. Example studies from the natural sciences include nonlinear influences of groundwater on ecosystem services (Qiu et al., 2019), groundwater depth thresholds to maintain tree canopy condition (Kath et al., 2014), regional precipitation patterns driven by distal groundwater irrigation (Lo & Famiglietti, 2013), and alternate stable states in groundwater-stream interactions (Zipper et al., 2022). In the social sciences, from which the social-ecological systems concept emerged, example studies include general design principles for self-sustaining irrigation institutions (Ostrom, 1993), identification of nested institutional arrangements in local irrigation communities (Cox, 2014), farmer adaptations to reduced groundwater availability (Running et al., 2019), the perception of fairness in groundwater allocation (Hammond Wagner & Niles, 2020), socio-historical studies on the social and political contexts that lead to successful implementation of managed aquifer recharge projects (Richard-Ferroudji et al., 2018), Indigenous knowledge systems in relation to water (McGregor, 2012), and analysis on the ability of low income, rural stakeholders to meaningfully participate in groundwater governance processes (Dobbin, 2020). There is also a third grouping of emerging interdisciplinary studies (Barthel & Seidl, 2017), which include suitability analysis of managed aquifer recharge that considers both physiographic setting and institutional design (Ulibarri et al., 2021), studies on interactions between groundwater user behaviors, social norms, and physical groundwater dynamics to establish rules for more sustainable groundwater management (Hammani et al., 2009), and evaluations of the effect and timing of initiatives to promote groundwater recharge (Patel et al., 2020).

Thus, we are far from the first to recognize the potential for a social-ecological framing to be applied to groundwater topics and to the groundwater sustainability discourse. However, amid this rich and diverse set of studies, we perceive a lack of foundational literature that integrates emerging trends in groundwater research though a common conceptual foundation. Furthermore, while these outcomes are often included in discussion sections of hydrogeological studies, they remain rarely modeled or explicitly considered in analysis. These relationships and outcomes become the explicit focus of analysis for groundwater-connected systems. Thus, our framing is syncretic in that it aspires to tie together and build on emerging trends in groundwater-related disciplines. Viewing these various research trends, overviewed above, through the common foundation of groundwater-connected systems can facilitate greater awareness, dialog, and collaboration between these research communities. Furthermore, the framing can provide a useful foundation to support the construction of hypotheses and to generate narratives about change in social-ecological systems connected to groundwater.

To illustrate the potential of the groundwater-connected systems framing to facilitate more systematic, holistic problem understanding that brings together multiple knowledge bases and data formats, we use an example outcome from Figure 2.3a: “dry wells and reduced rural water security” in the setting of California's Central Valley (Box 2.1). We argue that taking such a holistic systems view, regardless of the type of analysis to be conducted, supports a more rigorous identification of study assumptions, limitations, and potential in-roads across disciplines than when approached exclusively from narrowly defined disciplinary perspectives. Other benefits of this framing extend across data collection, scientific investigations, governance and management, and education topics, which the remainder of this paper is dedicated to.

Box 2.1. Understanding the outcome of "dry wells and reduced rural water security" through the groundwater-connected systems framing. For this example, we use the setting of California's Central Valley and use a narrative approach to weave together multiple perspectives, data sources, and formats.

In California's Central Valley, groundwater pumping accelerates during times of drought (Liu et al., 2022), further depleting groundwater resources. As this occurs, wells across the state run dry (Jasechko & Perrone, 2020).

"The whole time you're going, 'Oh please, let it be something else. Let it be a switch. Let it be the pump — let it be anything but being out of water,'" a domestic well owner in California's Central Valley (Becker, 2021).

The majority of groundwater withdrawal in the Central Valley occurs for agricultural irrigation, and the Valley is one of the most agriculturally productive areas in the world. Simultaneously, tens of thousands of domestic wells provide rural water security across the state (Pauloo et al., 2020). While the conventional drivers of groundwater behavior (e.g., geology, topography, and climate) remain important, the human fingerprint of groundwater pumping, climate change-induced drought, and land-use change are dominant drivers in this setting (sensu Abbott et al., 2019). Global processes also factor into this situation as the Valley is an exporter of virtual water (Marston & Konar, 2017). Thus, multiple tensions exist in the Central Valley, including but not limited to those between residents' water security and importing regions' food security, and between rural well owners and industrial agriculture regarding groundwater access.

> *"We want to be at the table. I know we are little, but we don't want to be left behind. We want to know what's going on."*
>
> *"What is your biggest problem? Farming? Who got all the control? Farmers. So good luck fixing the problem."*
>
> *"Who's representing the small people or the city or what not?"*

Excerpts from interviews conducted with rural community members in the Central Valley by Dobbin (2020).

Absent or ineffective regulations on groundwater use and a lack of policy coordination between food, water, and energy goals are common in areas experiencing groundwater depletion (Villholth & Conti, 2018; Molle & Closas, 2020). Despite the accelerating rate of groundwater depletion in the Valley, placing the state's groundwater resources on pathways to sustainability has been a policy objective since the development and subsequent enactment of the Sustainable Groundwater Management Act. The Act's decentralized approach delegates the process of defining groundwater sustainability to local groundwater sustainability agencies, creating nested, context-based opportunities for managing groundwater. Yet, risks to rural water security may occur in locations where existing power and economic inequalities come to dominate this process. This is possible through the setting of management targets, often water table depths, that may be derived without engagement with rural, disadvantaged communities and that favor dominant, richer, and industrial users who are able to afford the drilling costs of deeper wells (Bostic et al., 2020). This process can thus entrench existing bias found in newsprint and science in favor of the interests of the agricultural industry, leaving interests of disadvantaged rural communities “underrepresented, understudied, and underserved” (Bernacchi et al., 2020; Fernandez-Bou et al., 2021).

The Yocha DeHe Wintun Nation stewards over 40,000 acres in the Yolo Subbasin of the Sacramento Valley. On these lands, Yocha DeHe Wintun Nation practises both traditional food cultivation and production agriculture. The Nation's name, Yocha DeHe, translates to “home by the spring water” (Romero-Briones et al., 2020).

Simultaneously, falling water tables also place at risk groundwater-dependent ecosystems (GDEs) (Rohde et al., 2019), with estimates indicating nearly half of all GDEs in California have experienced declining groundwater levels (Rohde et al., 2021). Yet not only are the subterranean, terrestrial, and aquatic ecosystems placed at risk through groundwater depletion, but so too are the myriad set of ecosystem services and cultural values of GDEs (Kreamer et al., 2015). Thus, a focus on only human-groundwater relationships overlooks processes that link groundwater use with ecosystem health, and the feedback mechanisms that can impact humans through deteriorated ecosystem services provided by these GDEs. These include services that directly support water security, such as water purification, aquifer storage, and buffering hydrological extremes, and broader services that support social well-being including the cultural services associated with groundwater's recreational, spiritual, religious, and esthetic values (Gleeson et al., 2022).

This application of the groundwater-connected systems framing to California's Central Valley demonstrates how integrating multiple perspectives, data sources, and formats develops a more holistic understanding of the system than can be provided by each study in isolation. In doing so, it argues that it is necessary to look beyond strict hydrogeological assessments and methods to understand the dynamics and impacts of changes in groundwater-connected systems.

2.4 Invigorating groundwater sustainability with sustainability science

Groundwater sustainability, as a subdiscipline, lies at the intersection of groundwater science with sustainability science (see intersecting circles in Figure 2.2a). Sustainability science has blossomed over recent decades into a rich and robust literature (Table 2.1), yet our view is that groundwater topics have been underrepresented in sustainability science studies in contrast to other common pool resources such as forests and fisheries (Kajikawa et al., 2014). As social-ecological systems and their associated language and concepts permeate the sustainability science discourse, we see significant potential for greater application of sustainability science concepts to groundwater through the groundwater-connected systems framing. Doing so moves groundwater work toward increasingly interdisciplinary, relationship-centric, and complexity-based approaches (see arrow in Figure 2.2a).

To facilitate this, we provide below a brief sustainability science primer for hydrogeologists through a set of core sustainability science concepts: wicked problems, the multiple scales and dimensions of sustainability, and an introduction to analysis frameworks. Though this set of terms is limited, we view their collection as a minimum but representative set of introductory concepts alongside the key references provided in Table 2.1. We briefly summarize and connect these key concepts to our framing of groundwater-connected systems.

Wicked problems are problems with no single solution, where conflicting values and a variety of standpoints between partners, collaborators, and stakeholders lead to different situational understandings and desired outcomes (Lönngren & van Poeck, 2021). Wicked problems are found in social-ecological systems where interactions among social, economic, and biophysical systems are poorly understood, highly variable, and can produce undesirable consequences from well-intentioned actions. Owing to these properties, wicked problems are not solved as much as they are continuously managed (DeFries & Nagendra, 2017).

Whereas the physical renewability of a groundwater system can be objectively defined through, for instance, a water balance, sustainability in groundwater-connected systems should be approached as a wicked problem. Drivers of groundwater depletion and misuse are complex and diverse (see Box 2.1), and the challenge of steering groundwater systems on pathways toward sustainability is well reflected in the literature (Ostrom, 1993; Zellner, 2008; Aeschbach-Hertig & Gleeson, 2012; Zwarteveen et al., 2021). Important groundwater-connected processes occur across a wide range of spatial and temporal scales, which span well-head to catchment, aquifer, and transboundary domains, to the global scale; and across seasonal to century and longer time ranges (Figure 2.4). These interactions between processes of dramatically different spatial and temporal scales contribute to the 'wicked' nature of sustainability in groundwater-connected systems.

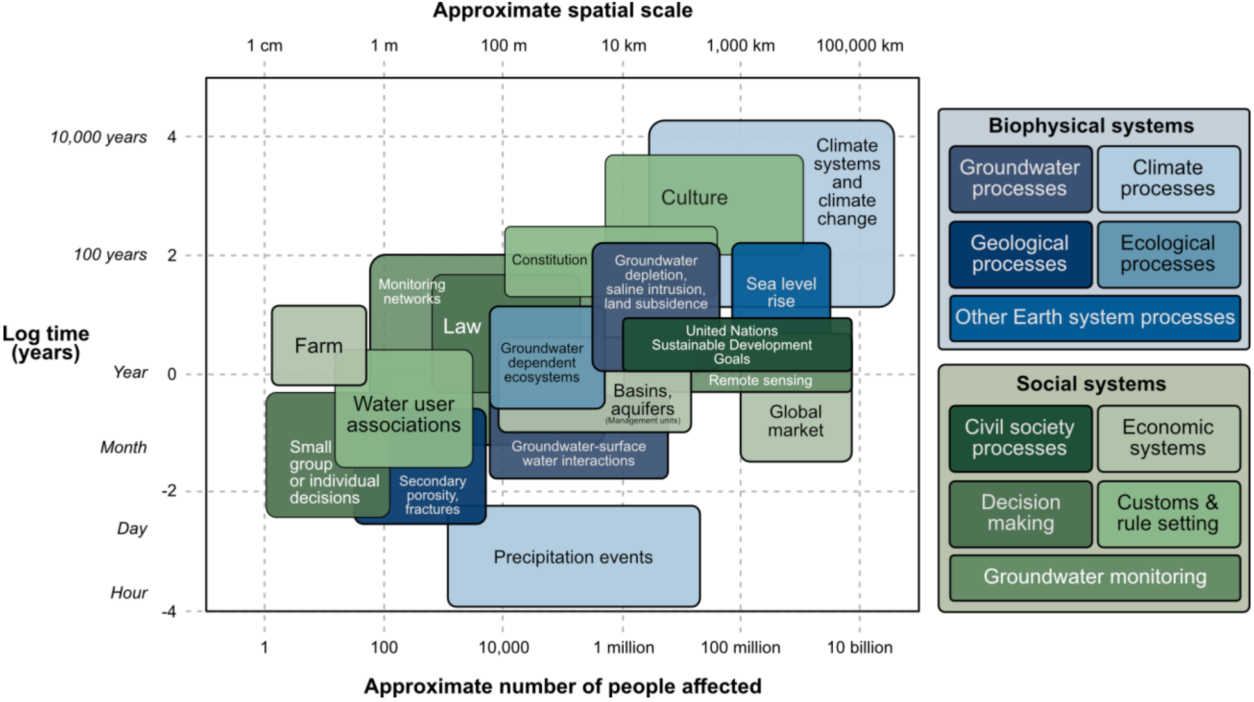


Figure 2.4. Spatial, temporal, and social scales of biophysical and social processes of groundwater-connected systems.

The processes shown are not comprehensive but are intended to illustrate the diversity of processes across scales.

Sustainability is a deeply normative concept and is tightly coupled to notions of justice (Jerneck et al., 2011; Wijsman & Berbés-Blázquez, 2022). The contemporary concept of sustainability is rooted in the Brundtland Report's (WCED, 1987) definition of sustainable development: “development that meets the needs of the present without compromising the ability of future generations to meet their own needs” (Purvis et al., 2019). While this foundational definition concerned intergenerational equity, current definitions have expanded to also include considerations of equity across spatial and social dimensions (Jerneck et al., 2011). Thus, sustainability is a multidimensional concept expressed through determinations of what is equitable across generations (temporal dimension), regions (spatial dimension), and identities (socio-economic or cultural dimension). These determinations hinge on normative judgments of ‘what should be’ (Lélé & Norgaard, 1996). Finding consensus in these discussions can be elusive with contested understandings of what goals should be pursued.

Sustainability-focused and framed groundwater research is rapidly growing (Elshall et al., 2020), and application of sustainability science concepts are already present in the existing literature. Notable examples include increasingly expansive groundwater sustainability definitions (Gleeson et al., 2020a), modeling approaches that consider complex social and institutional dynamics (Castilla-Rho et al., 2015), and transdisciplinary approaches that directly engage groundwater users as research partners (Zwarteveen et al., 2021).

Applying sustainability science frameworks to groundwater sustainability topics is an important step to further align these literatures and can provide additional insights to better delineate the groundwater sustainability problem space, understand its complexity, and guide more effective and engaged work. A framework is the “most general form of conceptualization; [providing] checklists or building blocks for consideration in constructing theories or models” (Clark & Harley, 2020). In our illustration of groundwater-connected systems as social-ecological systems (Figure 2.3), we used the Social-Ecological Systems Framework of (McGinnis & Ostrom, 2014). Many other frameworks exist to study social-ecological systems. For a comparison of common frameworks, see Binder et al. (2013).

Table 2.2. Added considerations for groundwater sustainability through the application of the groundwater-connected systems framing.

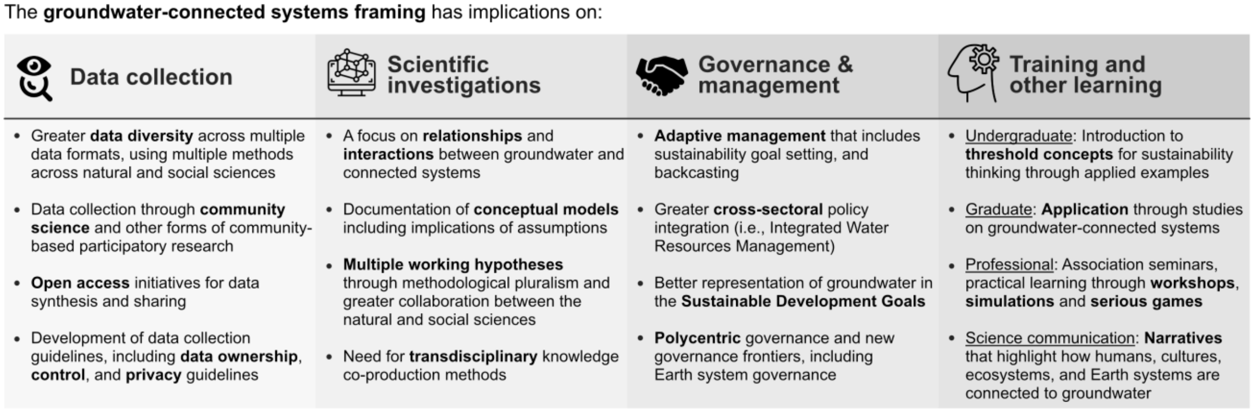
Conventional considerations for groundwater sustainability	Additional considerations for groundwater sustainability through the groundwater-connected systems framing
Flux-based approaches: • Recharge rate (Döll & Fiedler, 2008) • Mean renewal time (Bierkens & Wada, 2019) • Groundwater development stress (Alley et al., 2018) • Water balance (Richey et al., 2015) • Groundwater footprint (Gleeson et al., 2012b) • Environmental flow needs (de Graaf et al., 2019) Long-term goal setting and backcasting (Gleeson et al., 2012a) Calls for equitable, inclusive, and long-term governance and adaptive management (Gleeson et al., 2020a)	How do changes in groundwater quantity and quality lead to changes in ecosystem services? How does groundwater access change with trends in groundwater storage? Are impacts faced evenly across the affected population? Are access inequalities being formed or amplified? And, how do social and economic attributes affect individuals' abilities to cope with changing groundwater quality and quantity? Are existing power and economic inequalities dominating groundwater governance processes? Are cultural values and other social relationships to groundwater acknowledged and valued in sustainability plans and management decisions? How are groundwater storage trends altering the Earth system? How are changes in Earth system components impacting local to regional scale groundwater resources, such as through altered rates and spatial patterns of groundwater recharge?

The groundwater-connected systems framing does not call to replace existing definitions of physical groundwater sustainability. Instead, the framing provides additional considerations to apply alongside determinations of physical sustainability (Table 2.2). Physical renewability therefore becomes a necessary but insufficient condition for broader social-ecological sustainability in groundwater-connected systems. These broader considerations can include equity of groundwater access across different user groups and communities, determination of ecological thresholds for groundwater use, identification of cultural sites that depend on groundwater, tracking of community participation and engagement levels in monitoring and management initiatives, and broader considerations of environmental justice. In applied settings, this could take the form of quantitative analysis, such as calculating horizontal inequality ratios

(Boyce et al., 2016) for groundwater accessibility across user groups, tracking citizen science participation rates, or using satellite imaging to determine the proportion of a landscape whose terrestrial ecosystem thresholds for water table drawdown have been exceeded. Likewise, applied qualitative analysis could take the form of tracking community member perceptions of fairness in groundwater allocation decision-making processes, sense of well-being in relation to the services and functions provided by groundwater, or routine analysis and synthesis of community member perceptions of hydrological, ecological, and socio-economic change. These possible additions reflect the multi-objective nature of sustainability in groundwater-connected systems.

2.5 Wide applicability to groundwater science and beyond

The groundwater-connected systems framing does not provide an explicit roadmap to follow. Rather, we provide here a set of possible implications across the core domains of data collection efforts, scientific investigations, governance and management approaches, and education (Figure 2.5). Our aim is to provide an overview of the breadth of work we believe the groundwater-connected systems framing can contribute to.



The **groundwater-connected systems framing** has implications on:

Data collection	Scientific investigations	Governance & management	Training and other learning
Greater **data diversity** across multiple data formats, using multiple methods across natural and social sciences	A focus on **relationships** and **interactions** between groundwater and connected systems	**Adaptive management** that includes sustainability goal setting, and backcasting	Undergraduate: Introduction to **threshold concepts** for sustainability thinking through applied examples
Data collection through **community science** and other forms of community-based participatory research	Documentation of **conceptual models** including implications of assumptions	Greater **cross-sectoral** policy integration (i.e., Integrated Water Resources Management)	Graduate: **Application** through studies on groundwater-connected systems
Open access initiatives for data synthesis and sharing	**Multiple working hypotheses** through methodological pluralism and greater collaboration between the natural and social sciences	Better representation of groundwater in the **Sustainable Development Goals**	Professional: Association seminars, practical learning through **workshops**, **simulations** and **serious games**
Development of data collection guidelines, including **data ownership**, **control**, and **privacy** guidelines	Need for **transdisciplinary** knowledge co-production methods	**Polycentric** governance and new governance frontiers, including Earth system governance	Science communication: **Narratives** that highlight how humans, cultures, ecosystems, and Earth systems are connected to groundwater

Figure 2.5. Implications of the groundwater-connected systems framing.

Implications are list for data collection, scientific investigations, governance and management approaches, and education, training, and communication.

2.5.1 Implications for data collection

Empirical, grounded analysis of groundwater-connected systems requires observational data on the relationships that constitute these systems. The relevant data space to study groundwater-connected systems includes all social-ecological systems that interact with groundwater resources (e.g., Figure 2.3). Thus, this data space is more expansive and diverse in comparison to the data requirements for hydrogeological studies. These data can include conventional types of hydrogeological data, such as water table levels, but also extends to less traditional data such as the extent and type of groundwater-dependent ecosystems, governance, and economic and social dimensions including data on social norms, drivers of groundwater user behaviors, the effectiveness of rules, and community values in relation to groundwater. At present, little of this multidimensional data is collected and shared.

Yet, this expanded delineation of relevant data for groundwater studies introduces data formats that do not easily integrate with the typical data workflows and numerical models of groundwater hydrologists. For example, dominant data types in the social sciences are in the form of qualitative case study outcomes, surveys, and interviews. There is a long list of applied environmental topics and research communities also navigating the challenges of integrating the social and natural sciences (Strang, 2009; Hirsch Hadorn et al., 2010) for groundwater-connected systems to learn from and build on. While some notable groundwater studies do exist that integrate multiple data formats (e.g., Castilla-Rho et al., 2017), the enduring challenge remains to integrate data while preserving the subtlety and fidelity of each data format (Pooley et al., 2014). Noting that social sciences often face situations of reduced power and influence when in collaboration with natural scientists (MacMynowski, 2007), great care and methodological attention is needed to ensure that social science data is not “compressed into extinction” (Strang, 2009; Pooley et al., 2014). To accomplish this requires significant amounts of time dedicated to understanding the different research philosophies and methods used among interdisciplinary collaborators, which can help avoid collaborative work from only using data that integrates easily with the methods of the dominant discipline (Strang, 2009).

Pursuing more comprehensive data collection is accompanied by the additional need to synthesize such efforts via open-access initiatives. This call to collect more diverse data requires careful consideration of what data is not only practical but ethical to obtain and share. Zipper et al. (2019) provide guidance in navigating the open science-data privacy dilemma in socio-hydrology, which can also apply to groundwater-connected systems data.

One opportunity to address data deficiencies is to embrace the potential of community or citizen science (Buytaert et al., 2014) and other forms of community-based participatory research.

Community science not only fills observation deficiencies but also leads to increased social awareness of change in human-environmental systems (Kimura & Kinchy, 2016). Thus, these initiatives are particularly relevant in regions where groundwater-connected systems are undergoing rapid change.

2.5.2 Implications for scientific investigations

As an overriding implication on scientific practice, the groundwater-connected systems framing forces a recognition of the role and influence of the researcher. This calls on researchers to examine the impact of their technical expertise and research philosophy on study design and outcome. The groundwater-connected systems framing challenges the conventional view in the natural sciences of doing ‘good’ science while holding no opinions and urges against claims of objectivity in study outcomes.

To facilitate this reflexivity, greater focus needs to be placed on documenting conceptual models in these higher-dimensional, more complex studies. Doing so not only aids in identifying the strengths of a given approach but also explicitly highlights the processes considered and omitted from representation, the limitations of these decisions, and the uncertainties they introduce. Documenting limitations and uncertainty does not undermine a study's value but rather is a core research output that aids in locating knowledge gaps and informing subsequent work (Wagener et al., 2021). Such clarification requires stating and justifying assumptions underpinning analyses. This focus on uncovering assumptions is consistent with recent calls in the groundwater modeling literature (“assumption hunting” in Peeters, 2017) but extends across a wider, interdisciplinary domain for groundwater-connected systems. Furthermore, this methodological introspection can facilitate more effective collaborations by increasing mutual understanding across disciplines (Strang, 2009).

To address uncertainty given stark structural differences between models, the method of multiple working hypotheses via an ensemble-of-models approach is already being used in the groundwater and hydrological modeling communities (Clark et al., 2011; MacMillan, 2017). This many-model paradigm can lead to wiser choices, more accurate predictions, and better constrained uncertainty. Ensemble-of-model approaches should be pursued for topics concerning groundwater-connected systems which are characterized by less process understanding and greater uncertainty relative to physical groundwater systems. This approach does not need to take any particular form and can be used to integrate methodologically diverse studies, each fit for a specific purpose, to identify common outcomes and areas of convergence and divergence (Castilla-Rho et al., 2020).

Research on groundwater-connected systems necessarily must focus on the relationships and interactions between system components rather than on groundwater in isolation. Such research often aims to identify complex system attributes and behaviors (e.g., Figure 2.3c). For instance, methods to detect early-warning signals for regime shifts in complex systems (Scheffer et al., 2009) are only just beginning to be applied to groundwater-connected systems (e.g., Zipper et al., 2022). Alternatively, the heterogeneity of groundwater-connected systems requires that actions to promote sustainability in these systems fit the local context. For example, studies (e.g., Richard-Ferroudji et al., 2018, Ulibarri et al., 2021) that identify the combination of socio-economic, institutional, infrastructural, and hydrogeological conditions that lead to successful implementation of managed aquifer recharge projects are a useful advance beyond conventional feasibility studies that focus exclusively on the physical system and setting. Lastly, quantitative studies that identify macro-level conditions that characterize a social-ecological system's composite state or behavior can be found in the broader social-ecological literature (Leslie et al., 2015; Williamson et al., 2018) but have yet to be adapted for groundwater-connected systems.

The groundwater-connected systems framing also creates space for greater adoption of community-based participatory research that enables data and knowledge co-production in transdisciplinary settings. Such knowledge co-production can facilitate the integration of multiple knowledge bases and can help ensure that research better reflects local partner and stakeholder values and relationships with groundwater. Simultaneously, community-based participatory research strengthens scientific practice and output by canvassing a larger evidence base to inform studies (sensu Tengö et al., 2014). These transdisciplinary interactions between academics and stakeholders can create synergistic interactions across knowledge systems and worldviews (Castilla-Rho et al., 2020).

2.5.3 Implications for governance and management

Shifting from a resource-centric to a social-ecological systems approach can avoid traditional tendencies of disconnecting groundwater resources from their social context. Doing so rejects the types of simplistic and uniform thinking that have led to failed top-down, technical, and one-sizefits-all governance designs (Villholth & Conti, 2018). Instead, the social-ecological systems lens recognizes integrated and connected governance systems as social and political phenomena (Closas & Villholth, 2020). In this way, it unlocks opportunities for more tailored and orchestrated polycentric governance solutions that, under the right conditions, can support more democratic, sustainable, and resilient outcomes (McGinnis, 2016).

Complex adaptive systems provide an alternative paradigm to equilibrium-based approaches and support the linking of adaptive management and participatory modeling processes (Crevier & Parrott, 2019). Such adaptive management needs to be underpinned by sustainability goal setting and backcasting (Gleeson et al., 2012a). Sustainability goals in groundwater-connected systems can be informed by multi-objective initiatives such as the Sustainable Development Goals, and multi-scalar objectives such as downscaled planetary boundaries (Zipper et al., 2020). However, global and downscaled objectives require reconciling with place-based values, preferences, and norms. Thus, the pursuit of bottom-up approaches that can include self-regulation or peer-to-peer monitoring that also fit within broader multi-scalar sustainability goals is a grand challenge for governance in groundwater-connected systems.

Underrepresentation of groundwater in global sustainability initiatives limits such multi-scalar approaches. Most notably, groundwater is largely absent from the Sustainable Development Goals (Gleeson et al., 2020a) despite being connected to nearly half of the initiative's targets (Guppy et al., 2018). The groundwater-connected systems framing supports the consideration and thus inclusion of groundwater in such interdisciplinary, multi-objective initiatives and helps confront the overlooked and invisible history of groundwater in policy discourses.

Other works calling for social-ecological approaches to groundwater elaborate more extensively on management implications. See Bouchet et al. (2019) for a discussion on strategic adaptive groundwater management, and Barreteau et al. (2016) for a description of an integrated groundwater management landscape across water, land, and energy sectors.

2.5.4 Implications for education, training, and communication

Groundwater-connected systems span conventional academic disciplines and require different skill sets than those used in traditional, discipline-specific groundwater work. This discipline spanning is common across sustainability science and challenges conventional education pathways. Fruitful uptake and implementation of the groundwater-connected systems framing will rely on its incorporation into the training of groundwater academics, practitioners, policy makers, users, and stakeholders. Below we highlight how the framing can interface with education at the undergraduate and graduate levels, to existing professionals, and in science communication efforts.

As it is crucial to develop a strong disciplinary foundation, we do not advocate for any fundamental changes to training at the undergraduate level. Yet, in such disciplinary programs, we believe it is possible and important to expose students to core concepts of sustainability science at an introductory level. Doing so fosters an awareness of the interdisciplinarity and complexity of groundwater-connected systems and underscores the need for disciplinary specialists to participate in diverse teams when identifying and solving problems in applied settings. In our own teaching of upper-year civil engineering courses on water sustainability and groundwater hydrology (Huggins and Gleeson, 2022), we have begun introducing sustainability science fundamentals, including the “threshold concepts” of sustainability science (Loring, 2020), through applied case examples and in-class activities. These are often tied to multimedia resources such as the Water Underground Talks (https://www.waterundergroundtalks.org/), an initiative that shares short interviews and research talks on groundwater connections to climate, food, and people.

We perceive graduate degrees as the appropriate level for more rigorous application of the concepts discussed in this paper. There is already a rich global ecosystem of graduate programs, schools, and research institutes that focus on social-ecological systems, resilience, and complex adaptive systems (e.g., the Stockholm Resilience Centre, the Centre for Sustainability Transitions, the Ashoka Trust for Research in Ecology and the Environment). Yet, we see potential for the graduate courses and research theses conducted at these institutes to place a greater focus on groundwater. The groundwater-connected systems framing can be used to facilitate this uptake of groundwater topics in social-ecological systems education and research.

There is also a need for professional training and development initiatives to introduce professionals to the framing of groundwater-connected systems. These could include practitioner-focused seminars; online guides to groundwater-connected systems concepts, methods, and data; and interactive workshops that could use agent-based models or serious games (e.g., Ouariachi et al., 2018) that would enable participants to grapple with complexity, adaptation, feedback mechanisms, and uncertainty in a risk-free environment while gaining practice working in inter- and transdisciplinary teams.

Finally, the framing of groundwater-connected systems can be a powerful tool to build public awareness on the importance of groundwater in everyday life and sustainable, equitable futures. While groundwater is often ‘advertised’ to the public through impressive statistics (e.g., as the world's largest store of unfrozen freshwater), we perceive that few aside from groundwater hydrologists will find interest in groundwater presented this way amid global pandemics, conflicts, and social movements. With the same motivation as the groundwater-connected systems framing, we argue that we should present groundwater in a more relational sense. Presenting groundwater in relatable narratives is a compelling and effective way to increase public interest in groundwater. One way to do this is by telling stories about the ways people are connected to groundwater, such as through the food we eat and the activities we enjoy and find important, such as swimming or ceremonies, among other social and cultural relationships to groundwater.

2.6 Conclusion

Groundwater-connected systems are formed by social, economic, ecological, and Earth system interactions with physical groundwater systems. We present the framing of groundwater-connected systems to facilitate greater representation of these interactions in groundwater research and practice through data collection, scientific investigations, governance, management, and education. However, the framing does not provide a specific blueprint for all to follow. Rather, we present this framing as an invitation to the groundwater community to revisit foundational concepts and explore a wide set of methods that can be used to advance groundwater science and sustainability in diverse hydrogeological, social, and ecological contexts. Thus, the groundwater-connected systems framing can provide a useful basis for growth and collaboration within the groundwater community. Equally, the framing is an invitation to other disciplines and the social-ecological research community at large to join us in advancing this uncertain, complex, and needed research on groundwater connections and sustainability in social-ecological systems.

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2.8 Author contributions

X.H. conceived the issue paper with advice from T.G., J.C.R., and J.S.F. X.H. produced all figures, with input on Figure 2.2 from: T.G., J.C.R., and J.S.F.; Figure 2.3 from: T.G. and J.C.R.; Figure 2.4 from: J.C.R.; and Figure 2.5 from: T.G., J.C.R., V.R., and C.H. X.H. lead writing, and all co-authors (T.G., J.C.R., C.H., V.R., and J.S.F.) edited and discussed the manuscript at multiple stages.

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Chapter 3

THE OPEN DATA LANDSCAPE TO STUDY GROUNDWATER DYNAMICS IN SOCIAL-ECOLOGICAL SYSTEMS: A SCOPING COLLECTION AND REVIEW OF GLOBAL DATASETS AND AN ASPIRATIONAL FUTURE OUTLOOK

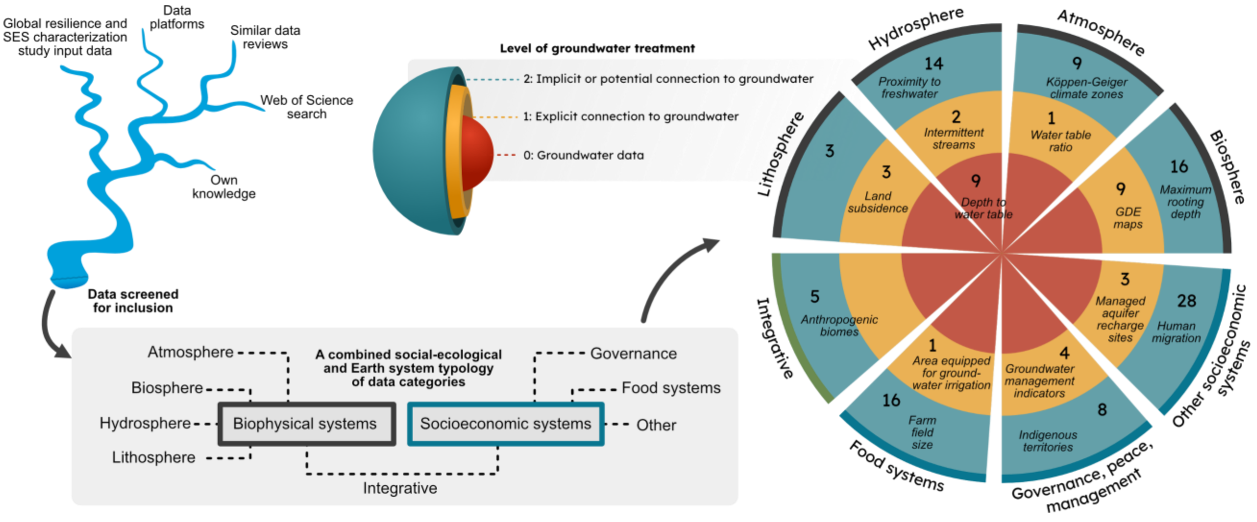
This chapter will be formatted for submission.

Huggins, X., Gleeson, T., & Famiglietti, J. S. (2024). The open data landscape to study groundwater dynamics in social-ecological systems: A scoping collection and review of global datasets and an aspirational future outlook. Manuscript.

Key points:

- Develops and reviews a collection of over 130 global open geospatial datasets relevant for the study of groundwater systems as social-ecological systems.
- Most datasets are temporally static, time series data peak during the 2000-2010 decade, and nearly all data are generated from institutions in the global North.
- Discusses essential variables and data cubes as concepts to guide future data collection.

Figure 3.1. Graphical abstract.



3.1 Abstract

Global data have served an integral role in developing large-scale knowledge on groundwater systems. As global groundwater science moves towards greater interdisciplinarity, where ecological, socioeconomic, and Earth system interactions with groundwater receive greater attention, the success of this shift will depend on the development of a wider variety of compatible interdisciplinary datasets. We identify a collection of 131 global datasets that span elements of the hydrosphere, biosphere, atmosphere, lithosphere, food systems, governance, management, and other socioeconomic systems relevant for groundwater studies. As the global groundwater literature grows, this study offers a reference of existing data for use in interdisciplinary assessments while simultaneously providing motivation and guidance for future work to address gaps and biases in the existing collection of data. We find that datasets with implicit or potential connections to groundwater systems outnumber datasets with explicit representation of groundwater, followed by direct groundwater observations which count the fewest number of datasets. The majority of data explicitly addressing groundwater systems are temporally static, while we find time series data to peak over the 2000-2010 decade. These findings call for greater collection, collation, and development of temporally dynamic data to enable social-ecological system analysis of groundwater at the global scale. We identify a clear geographic bias in the institutions producing the reviewed data, highlighting the process of generating global datasets as a near-exclusive activity of researchers located in the global North. We conclude with a discussion on the concept of essential variables for groundwater-connected systems as a potential aspirational pathway to address existing weaknesses in the global data landscape and catalyze interdisciplinary data-driven groundwater sustainability science.

3.2 Introduction

Conceptualizing groundwater systems as social-ecological systems has been proposed as a more effective approach to understand groundwater and capture its changing role in the Anthropocene (Gleeson et al., 2020; Huggins et al., 2023; Kuang et al., 2024; Scanlon et al., 2023). Taking such a framing requires a broader consideration of groundwater systems, where hydrogeological systems are understood as part of a larger, intertwined network of socioeconomic, ecological, and Earth systems (Gleeson & Cardiff, 2013; Huggins et al., 2023). This argument to expand conceptual models about groundwater systems is supported not only by groundwater and social-ecological systems theory (Abbott et al., 2019; Berkes et al., 1998; Zellner, 2008), but also by a wide and yet unquantified variety and volume of global-scale data. Understanding the scope of data available to study groundwater through its social-ecological and Earth system connections is a fundamental but yet untaken step to support the development of social-ecological system conceptualization and theories for groundwater systems and to inform future data collection efforts.

Two perspectives contextualize our intentions to review the global groundwater data landscape. The first is the aforementioned development of a conceptual approach to study groundwater through a social-ecological systems framing. This groundwater-connected systems framing (Huggins et al., 2023) provides a novel and broad scaffolding with which to evaluate existing groundwater data. This framing recognizes groundwater's connections to socioeconomic, ecological, and Earth systems and visualizes these connections through the structure of the Social-Ecological Systems Framework, which explicitly considers human-environmental systems through the framework's elements: resource systems, resource units, related ecosystems, governance systems, actors, and social, political, and economic settings (McGinnis & Ostrom, 2014). Applied to groundwater-connected systems, these elements become, for instance, climate zones, groundwater storage, groundwater-dependent ecosystems, groundwater institutions, irrigation rates, and human development dimensions, respectively. Taking such a framing expands the scope of potential datasets for review, as the relevant system boundaries extend beyond hydrological, geological, and climate, and include socioeconomic and ecological data. The second is found in the growing awareness of subjectivity and bias in the scientific process, particularly among the natural sciences that have traditionally lacked self-reflection on the value-laden processes of their work and output (Lélé & Norgaard, 1996). Recognizing that different disciplines, research groups, and scientists bring with them biases in any given study, as argued by a critical realist scientific position (Cockburn, 2022), provides useful grounding to approach understanding these biases. A preliminary question that advances this perspective is: where are the institutions generating global groundwater datasets located? To our knowledge, no study has explicitly evaluated the international distribution of global-scale groundwater data generators and providers. Yet, doing so can be instructive to appraise what communities are leading the discourse on global groundwater systems and can facilitate reflections on implications of biases identified through this process.

Here, we identify a wide collection of global datasets that can be used to study physical groundwater system systems as social-ecological systems. We focus on global rather than regional or national-scale data as the global scale supports extensive analyses of groundwater systems that can enable systematic comparison between regions, is consistent with scales of Earth system processes (Gleeson et al., 2020), and mirrors the scale of global groundwater crisis and sustainability framings and discourse (Famiglietti, 2014; Gleeson et al., 2020; Scanlon et al., 2023). Furthermore, global datasets can be used as place-holder data in regions where localized data are not available, and thus reviewing global data inherently has a broader potential usage than focusing first on any specific region. We include exclusively open-access data, ensuring all datasets included in this collection are available for all to use without barrier. This criterion biases toward data generated within recent years (circa 2015 and later), reflecting the rise of open publishing and data deposition practices (Clark et al., 2021; Hall et al., 2022).

In total, we identify 131 global datasets and collect and derive metadata on each. These metadata include the primary Earth system or social-ecological system component represented by the data, spatial and temporal resolution, explicitness of groundwater representation or consideration, and the country of affiliation for lead and senior authorship (see Section 3.3). We evaluate strengths and limitations in the existing data landscape and conclude with discussion on a possible aspirational future for global groundwater data collection.

3.3 Review methodology

As the groundwater-connected system framing works to expand the scope of system considerations in groundwater science, this study faces additional complications in contrast to other reviews work with a more defined and established delineation of system boundaries. Furthermore, as the framing is nascent without an established literature or self-identifying language, we were required to adopt a piecemeal approach to consider multiple sources and wide search criteria to locate relevant datasets. We were interested in not only identifying data that have already been used to study global groundwater systems but also in identifying data that have the potential to enrich the study of global groundwater systems that may yet to be recognized or realized. Thus, a considerable amount of data with potential or implicit connections and relevance to groundwater is included in our reviewed collection.

This interest in performing a wide sweep of unconventional data for groundwater system applications doubles as a process that introduces notable bias. This bias is found in our judgments on which data are considered to have potential implications for groundwater studies and will reflect our disciplinary and conceptual model biases. Our decisions on data inclusion stem from a conceptual model that focuses on social-ecological system dynamics in relation to shallow, terrestrial, and physical groundwater systems. This system and review scope informs our decision to not consider offshore aquifers (Post et al., 2013), deep groundwater systems (Ferguson et al., 2023), or geochemical data (Edmunds & Smedley, 1996). These point to some of the biases in our collection of data which can be addressed through complementary initiatives in the future. Rather than only including the small set of existing global datasets with explicit consideration of physical groundwater systems, we argue that considering a wider selection of data to be more useful and insightful. While simultaneously recognizing the data we summarize are not exhaustive, this collection takes an aspirational form where the diverse collection of data sketches a broad, ambitious outline of what the global groundwater data landscape could evolve into.

3.3.1 Metadata categories

We developed a number of classification schemes that were applied across the data collection. These classifications include: (1) the identification of the primary system the data relate to, (2) the explicitness with which groundwater is treated and considered in the dataset, and (3) the dataset's treatment of time. We describe these three classification schemes in the paragraphs below.

To summarize the primary system of each dataset, we developed a composite classification system that combined elements from social-ecological system and Earth system framings. In all, we settled on a set of nine system categories: hydrosphere, lithosphere, biosphere, atmosphere, socioeconomic systems, food systems, governance, and an integrative category for datasets that span multiple systems (Figure 3.2a). We found that a combined approach (Figure 3.2b) helped address disciplinary biases in the two frameworks. That is, we found that using Earth system elements (natural science developed) aided to counteract the social system biases of a social-ecological system framing (i.e., the atmosphere, biosphere, hydrosphere, and lithosphere would all be classified under the broad term of biophysical systems). Similarly, borrowing elements from the Social-Ecological Systems Framework (social science developed) (SESF; McGinnis & Ostrom, 2014) helped balance overgeneralized human system representation common when using an Earth systems framing (i.e., governance, food systems, and other socioeconomic systems would all be classified under the Earth system term ‘Anthroposphere’). Here, the governance category is drawn directly from the ‘governance systems’ element of the SESF, while the food systems category is drawn from the ‘actors’ element, which we highlight due to agriculture representing the dominant form of groundwater use at the global scale (Wada et al., 2012).

To reflect the explicitness of groundwater consideration in each dataset, we classified all data into three ‘levels’ (Figure 3.3a). Level 0 data represent direct physical groundwater data (e.g., water table depth), level 1 data represent data with explicit groundwater treatments or connections (e.g., groundwater-driven wetlands, groundwater management indicators), and level 2 data represent data with implicit or potential groundwater connections (e.g., cropland area, freshwater ecoregions, gross domestic product). We applied a literal approach when assigning ‘explicit’ versus ‘implicit’ classes, where data were classified as implicit (level 2) unless the data generation process included explicit consideration of groundwater. For instance, global cropland datasets that do not specify irrigation sources were classified as implicit, whereas datasets that explicit identify area equipped for groundwater irrigation were classified as ‘explicit’. Similarly, global wetland maps that do not specify which wetland type were identified as implicit while those that do specify groundwater-dependent wetlands are identified as explicit (level 1). Thus, our procedure to assign explicitness classes was not based on the strength of underlying theory connecting the dataset with groundwater but rather on the data generation process itself.

We additionally collect metadata on the type and format of datasets. We use the term ‘data type’ to represent the nature of the dataset as either a zonal data product (e.g., Köppen-Geiger climate zones), a static dataset (e.g., farm field size), a time series (e.g., annual population estimates), or historical records (e.g., international water events). As zonal and static data can be challenging to classify, we used an approach where data were classified as zonal if their principal reason for derivation was to be used as a unit for data summary (e.g., IPCC reference regions), while we classified a dataset as static data if its primary purpose for derivation is the documentation of an underlying system property or attribute (e.g., near-surface porosity), even if a secondary purpose of the data can be as a zonal layer. To differentiate between time series and historical record classification, we used a definition of time series as data which are generated at regular time steps with consistent spatial extent whereas historical record data are often more irregular (e.g., water related conflicts) with potentially inconsistent time ranges depending on individual entries within the dataset (e.g., water levels within individual monitoring wells). Should historical records be synthesized into a dataset with regular time steps, this data would be considered as a time series. Data formats are either rasters, vectors (polygon, polyline, or point), or tabular. For raster data, we record the spatial resolution of the dataset, while for vector data we specify dataset specific spatial information such as the median size of polygons or map scale.

To map the geographic distribution of institutions generating these data, we record the country of the institution affiliated with the lead author. We also include this information for the corresponding author if lead and corresponding author differs for the dataset. For data with institutions as the data provider, we use the location of the institution's headquarters. We include the license or policy of each dataset in our metadata to identify and share the conditions on which the data are free to access and use. These data sharing agreements include creative commons licenses, dataset specific user agreement, or an explicit statement encouraging the use of data where a license or agreement is not readily identifiable.

We were challenged by the prospect of including data from large, self-contained research communities such as the wide variety of global hydrological model output (Reinecke et al., 2021; Schellekens et al., 2017), global precipitation datasets (Sun et al., 2018), and other Earth observation datasets (McCabe et al., 2017). As these communities have respective data reviews and repositories (see preceding references), we include only the datasets most relevant for social-ecological system assessments in our collection to prevent this work from developing into an intractable review of reviews. Yet, our decisions on what data are included provides an additional source of bias in this work and reflects an underlying weakness of the groundwater-connected systems framing that encourages interdisciplinary thinking but does not offer guidance on a stopping or boundary criterion.

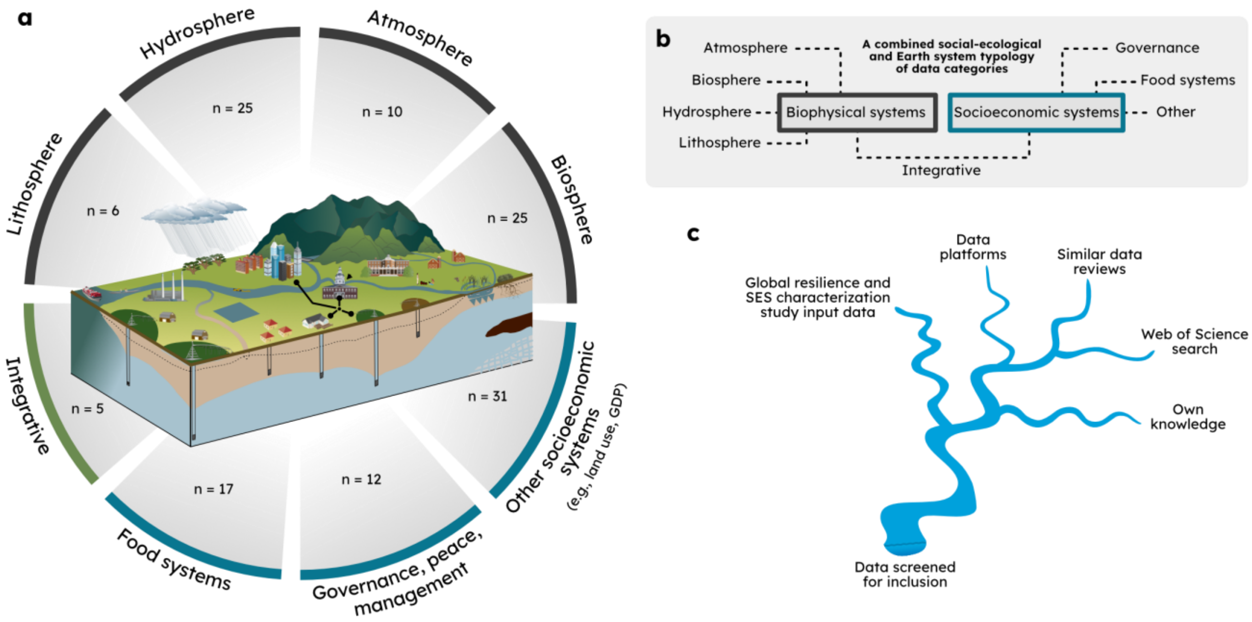


Figure 3.2. Overarching scope and sources of data.
(a) System classification scheme used to structure the data included in this review. Counts are provided for the number of datasets in each class. (b) The combined social-ecological and Earth system typology of data categories used to sort datasets included in our collection. (c) Sources used to identify datasets.

3.3.2 Dataset identification

We drew from multiple sources to identify data for inclusion in our reference collection. Each 'source' of data is visualized as a data 'tributary' (Figure 3.2c). To develop this list of data tributaries, we sought to include as many avenues as possible for data to appear in our search. Thus, we not only reviewed datasets associated with published studies, but also reviewed input datasets used in thematically aligned global social-ecological system assessments, leading global data platforms, and compatible global data reviews. These five tributary sources are described in Table 3.1.

Table 3.1. Description of data sources consulted to develop our dataset collection.

Data 'tributary'	**Sources *and how they were screened***
Data used in global social-ecological system characterization studies	Ellis & Ramankutty (2008), Gain et al. (2016), Sietz et al. (2011), Václavík et al. (2013), Varis et al. (2019) ➜ *We screened all input datasets used in each study.*
Global data platforms	WRI Aqueduct (https://www.wri.org/aqueduct) WRI Resource Watch (https://resourcewatch.org/) IWRM data portal (https://iwrmdataportal.unepdhi.org/) IGRAC GGIS (https://ggis.un-igrac.org/) WWF Water Risk Filter (https://riskfilter.org/water/home) Protected Planet (https://www.protectedplanet.net/en) MapX (https://unepgrid.ch/en/mapx) GRID-Geneva data platform (https://unepgrid.ch/en/platforms) EarthStat (http://www.earthstat.org/) SEDAC (https://sedac.ciesin.columbia.edu/) ➜ *For each platform, we screened all included datasets with global coverage for inclusion in our collection.*
Compatible global data reviews and commentaries	Bolognesi et al. (2018), Lindersson et al. (2020), Wang et al. (2022) ➜ *We screened all datasets reviewed or summarized in each study.*
Web of Science search	Search results were developed by filtering by "VAR" (see below) and "data," with the title filtered by the word "global", and results were filtered for those containing the tag of "associated data". This search was performed for "VAR" as "groundwater" (44), "socioeconomic" (95), "ecological" (255), "biophysical" (31), "governance" (36). Values in parentheses indicate the number of results for each query. ➜ *Each search result from the above queries was screened.*

3.4 Results

3.4.1 Data overview

In total, we identified and classified 131 datasets. All datasets, including metadata, persistent web-links, and citations are provided in Table III.1. A shortlist of relevant datasets excluded from our collection, including rationale for exclusion, is provided in Table III.2.

Table 3.2. Overview of data included in this review. See Table III.1 for all data citations.

System	List of variables
Atmosphere	**Zonal:** Köppen-Geiger climate zones, IPCC reference regions **Static**: Aridity index, Water table ratio **Time series**: Climate variables (e.g., precipitation, potential evapotranspiration) **Historical records:** -
Biosphere	**Zonal:** Freshwater and terrestrial ecoregions **Static**: Environmental flow groundwater head limit, Groundwater-dependent ecosystem extents, Amphibian and mammal species richness, Ecological conservation prioritization index, Global lakes and wetlands database, Ramsar Wetlands of International Importance, Ecohydrological classes of forest growth, Groundwater ecosystem biodiversity index, Root zone depth, Root zone water storage capacity, Groundwater-driven wetlands **Time series**: Vegetation indices (e.g., NDVI, EVI), Vegetation Index Phenology, Maximum rooting depth, Ecological vulnerability index, Vegetation health index, Wetland classification, Plant functional types **Historical records:** BioTIME biodiversity time series
Lithosphere	**Zonal**: - **Static**: Land subsidence, Permeability, Porosity, World soil database, Soil grids, Global lithological map **Time series**: **-** **Historical records:** -
Hydrosphere	**Zonal:** Watersheds (HydroBASINS), WHYMAP aquifers, Transboundary aquifers of the world **Static**: River width, Streamflow indices, Groundwater response time, Modern groundwater volume, River reach fragmentation, Lake bathymetry, Lake locations, River and stream intermittency, Groundwater recharge, Surface water extent, Groundwater vulnerability to floods and droughts **Time series**: Streamflow, Soil moisture, Water table depth, Runoff, Terrestrial water storage anomalies, Groundwater storage anomalies **Historical records:** River discharge, Groundwater monitoring wells, Dam locations and metadata

Food systems	**Zonal: -** **Static**: Crop allocation to end use, Gridded livestock systems, Crop harvested area, Crop production, Crop yield, Farm field size, Cropland area, Pasture area, Virtual water trade embedded in agriculture, Area equipped for groundwater irrigation **Time series**: Yield gap, Crop water footprints, Irrigation areas, Cropland extent **Historical records: -**
Governance, peace, management	**Zonal:** Administrative units, Indigenous territories, Indigenous treaties **Static**: - **Time series**: Varieties of democracy, Integrated water resources management implementation level indicators, Worldwide governance indicators, Environmental performance index **Historical records:** Water related intrastate conflict and cooperation, International river basin organizations, International water events, Water conflict map
Other socioeconomic systems	**Zonal:** Indigenous languages **Static**: Access to improved drinking water, Roads, Power plants, Accessibility to cities, Development pressure indices, Protected areas, Terrestrial human footprint, **Time series**: Groundwater withdrawal rates, **Historical records:** Managed aquifer recharge schemes, Electricity consumption, Population, Migration, Urban land fraction, Sectoral freshwater withdrawal, Gross domestic product, Human development index, Nighttime lights, Gender development inequality index, Human modification gradient index, Social adaptive capacity index, Gini index, Ambient population
Integrative	**Zonal:** Anthromes **Static**: Proximity to surface freshwater **Time series**: Land cover, River basin resilience **Historical records: -**

Of the 131 datasets (Table 3.2), we classified nine as direct groundwater data, 23 as explicitly linked to groundwater, and 99 as having an implicit or potential connection to groundwater (Figure 3.3a). All nine groundwater datasets (e.g., depth to water table, groundwater storage anomalies, etc.) are classified in the hydrosphere category, while the 23 datasets with explicit groundwater connections are distributed across all categories except the Integrative class. Amongst these 23 datasets documenting explicit system interactions with groundwater, biosphere data are most common (9) and include groundwater-dependent ecosystem maps, environmental flow thresholds, and rooting depth data. Yet, we find the most data with implicit or potential connections to groundwater within the socioeconomic systems classification (31), which includes data such as population count, gross domestic product, and gender development inequalities. Thus, we find a general trend where hydrosphere data dominate direct groundwater datasets, biosphere data dominate datasets with an explicit connection or treatment of groundwater, and socioeconomic data are most common among datasets with an implied or potential connection to groundwater. We reflect that this classification procedure is an inexact exercise and is principally motivated to ground a discussion on the data distribution across system types rather than present an authoritative assessment.

Socioeconomic, hydrosphere, biosphere, food systems, and governance data types all contain over 15 datasets, while lithosphere (6), integrative (5), and atmosphere (10) data are least represented. These distributions are partially reflective of methodological choices (e.g., as we focus on shallow groundwater systems) but could also reflect the potential for more explicit treatment of groundwater in these research disciplines. Of all the datasets reviewed, 12 were classified as zonal data, 57 as static datasets, 52 as time series, and 10 as historical records. We find that direct groundwater data are distributed across all four data types, while data that explicitly consider groundwater are predominantly static datasets (16 of 23).

We compared temporal ranges of all time series data (Figure 3.4), which reach as far back as the year 1900 and as far ahead as the year 2100. We find that the greatest overlap among time series data occurs during the 2000-2010 decade, peaking with over 40 time series available. A considerable spike in time series data occurs at the year 1990, where eight time series begin, and a consistent decline in time series availability is visible from the year ~2015 until present. We attribute this not to a decline in global time series data being generated, but rather to the time lag needed for research efforts to synthesize and publish data covering recent years. Only a small subset of time series data can be considered actively updated through on-going monitoring efforts. Among the time series data, we find very few direct (level 0) and explicit (level 1) datasets (Figure 3.4b) and thus these time series data largely represent systems with potential or implicit connections with groundwater. As discussed in section 3.5, moving many of these datasets from implicit to explicit grounds of groundwater consideration can be a major focus and priority for future global data efforts.

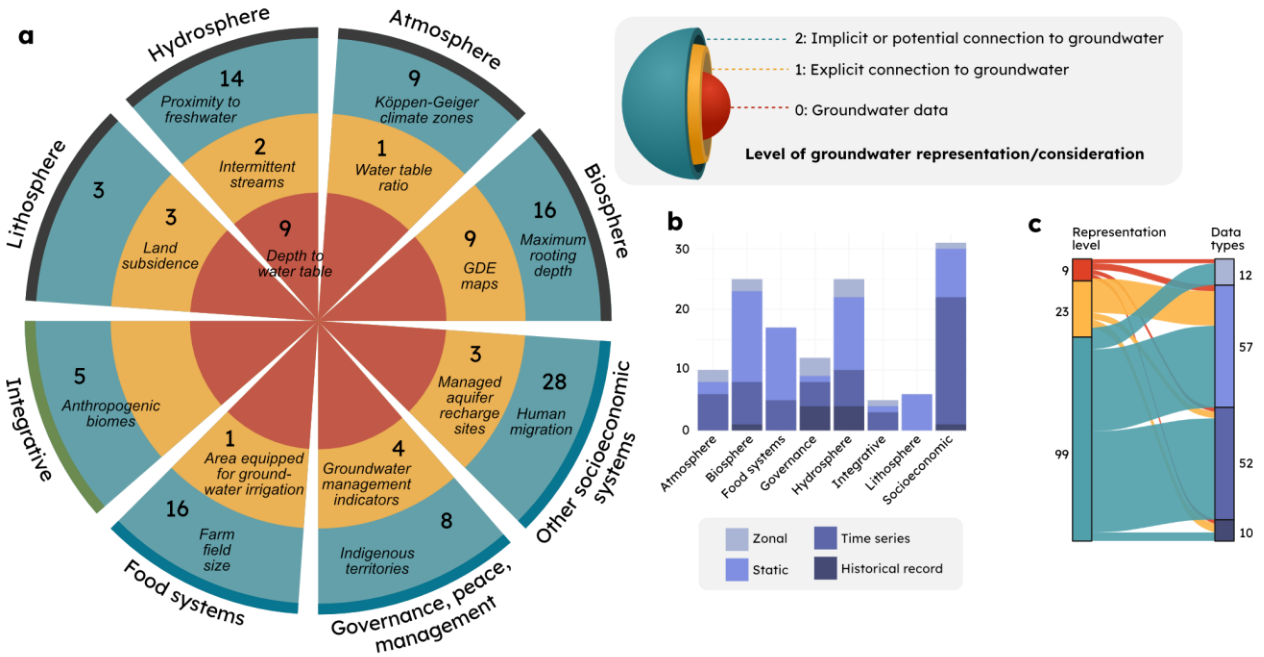


Figure 3.3. Underlying distribution of reviewed data across classification categories.

(a) Dataset count per system class and level of groundwater consideration. Example datasets for each classification are provided. (b) Distribution of data type per system type. (c) Relationship between level of groundwater consideration and data type.

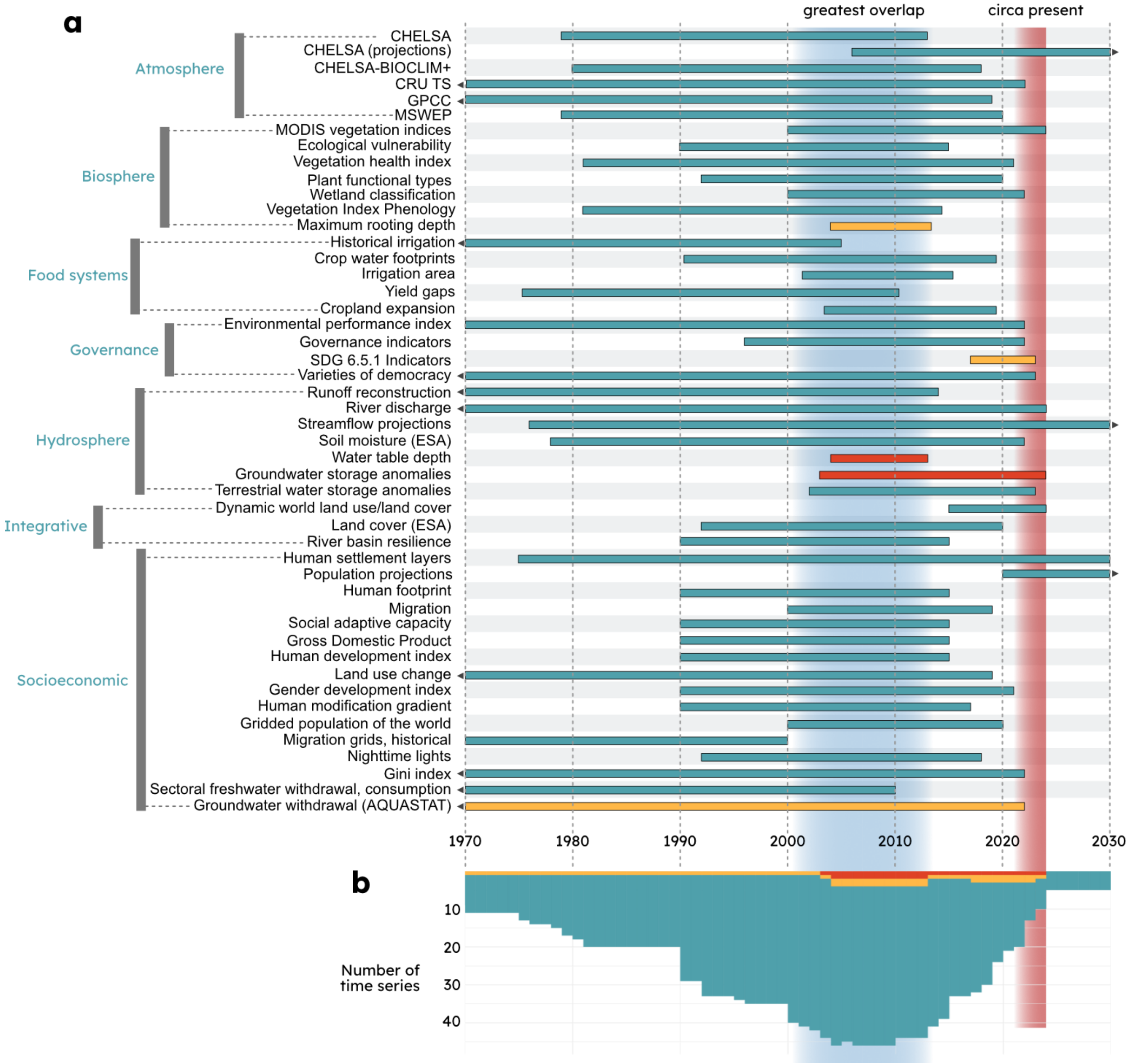


Figure 3.4. Temporal range of time series.

(a) Time range of individual datasets. (b) Histogram showing the number of time series available per year. Time series 'bars' are coloured according to the same legend as provided for Figure 3.3a.

We find a strong geographic bias in dataset authorship with the vast majority of datasets being generated from economically developed nations (Figure 3.5). The United States is the institutional home country generating the greatest volume of data, followed by the Netherlands, Canada, Finland, and Germany. Australia is the only country represented from the southern hemisphere, affirming these global datasets are overwhelmingly generated from the global North. We note the absence of several countries that are experiencing severe groundwater sustainability challenges including India, Iran, and Mexico. Additionally, South American and African nations are entirely absent from the authorship of these global data products, highlighting the representational bias in these data generation processes. These first-order observations prompt deeper questions of what research priorities are driving global scale research, whose interests these research priorities serve, the implications of potential mismatch between global scale research priorities with needs, knowledge systems, and worldviews in areas most acutely experiencing groundwater challenges worldwide. Fully unpacking these questions is beyond the scope of this work yet we hope these observations spur wider reflections on equitable and representative futures in global groundwater science.

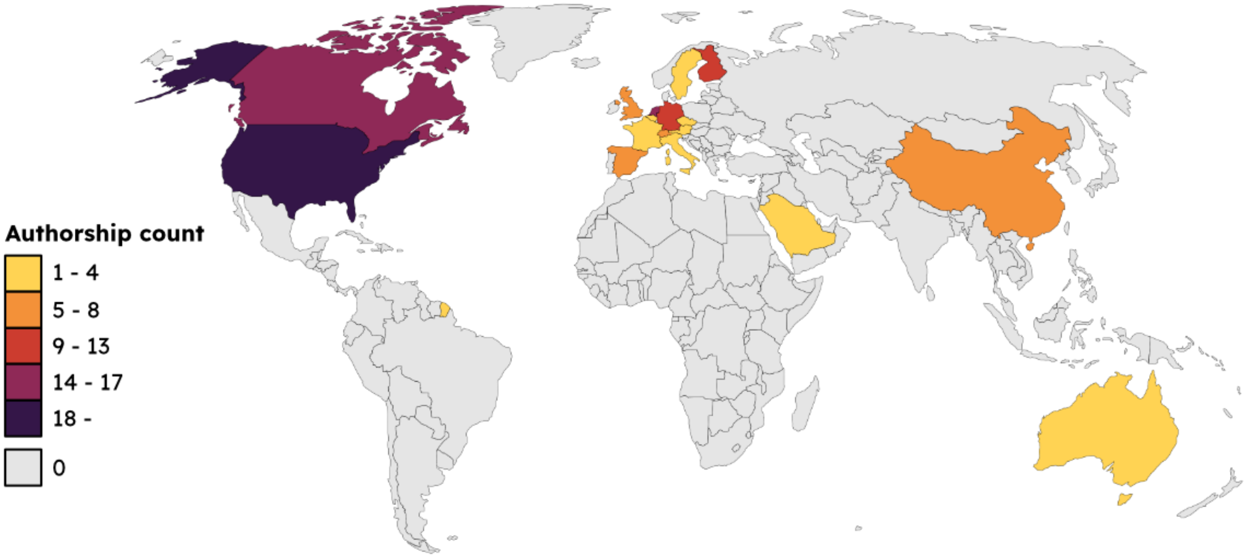


Figure 3.5. Institutional locations of dataset lead and corresponding authorship. Authorship counts do not equal the total number of reviewed datasets as two countries were counted when the lead author was affiliated with institutions in multiple countries and when the corresponding author's institution's country differed from that of the lead author.

Given a number of datasets are generated in large author studies, we focus only on the affiliations of lead and corresponding authorship affiliations. This approach, therefore, does not reflect the full diversity of researchers and institutions that participate in generating these global datasets but does reflect the institutions that are steering these efforts. We view this approach as a more representative and practical account of the dominant geographies and institutional contexts of these data generation activities.

3.5 Discussion

Our assembled data collection paints a portrait of the open data landscape available to study groundwater systems as social-ecological systems that is large and underutilized, yet also limited and biased. We find the volume of reviewed data to suggest that there exists a wide variety of global data that can be used to hypothesize, conceptualize, and model groundwater systems through a social-ecological systems framing. This points to a potentially fruitful and exciting future for such research directions in groundwater science. Yet, our finding that there is a much greater volume of data with potential or implicit (rather than explicit) connections with, or representation of, groundwater suggests that there is a need to revisit existing data to re-analyse and specify explicit relationships with groundwater. Until addressed, data-driven social-ecological assessments will necessarily be underpinned by substantial assumptions on dataset ability to represent interactions between groundwater and social-ecological systems. Yet, addressing these limitations is not trivial, and data developers must consider the role of existing data in the development of future datasets as dataset dependency can prevent or limit their use in system interaction studies (Lindersson et al., 2020).

As we find the vast majority of explicit (level 1) groundwater datasets to be temporally static, this review identifies an evident trade-off in between explicitness of groundwater treatment and temporal data availability (i.e., ‘explicit but static’ vs. ‘temporally dynamic but implicit’). This finding appears to constrain global studies to static analyses unless model-based approaches are used. Without sufficient temporal data availability, testing hypotheses on dynamic social-ecological system behaviour of groundwater systems, such as emergence, tipping points, context dependence, and system resilience (e.g., Preiser et al., 2018) will be limited to the conceptual and theoretical realm.

There are substantial omissions in the collection of reviewed data. These omissions include both system interactions with groundwater that have been studied but whose generated data is not openly available (such as data on hydro-political tensions), and interactions without treatment in the global-scale literature (such as ecosystem services). We view some of these omissions to be attributable to the relatively recent rise of open data practices or simply represent gaps in current data availability, while other omissions are enmeshed in larger, ethical discourses regarding data best-practices.

For instance, that global groundwater data are generated and studied in institutions often distant from the place- and land-based realities of acute groundwater sustainability challenges gestures to the potential implications of the global North bias identified above. Given that most of global land is contested (Meyfroidt et al., 2022), these are sweeping and critical principles to resolve (Carroll et al., 2021) yet have been absent in the global groundwater literature. How to reconcile global research agendas with the global mosaic of place-based values and needs is a question lacking guiding literature. Important social interactions with groundwater include sociocultural values such as groundwater's role in supporting senses of place among other cultural ecosystem services (i.e., recreational, spiritual, religious, aesthetic). Yet, there are no clear guidelines for the methodology to create such a global dataset on these services, nor ethical frameworks for the handling of this data. Even seemingly innocuous global data, such as the type and presence of groundwater-dependent ecosystems, may contradict Indigenous Peoples' authority to control data access that follow best practices established in recent data ethics frameworks (Carroll et al., 2021). Given an estimated 65% or more of land area is held under Indigenous Peoples' and local community customary systems (RRI, 2015), these are truly global ethical questions and priorities for groundwater science.

The proposition of "doing hydrology with dragons" (Wagener et al., 2021) has been introduced to encourage a greater reflection and embracing of process uncertainty in global hydrology. Might our above discussion add another dragon to this conversation, where spaces on maps (and in data) are left blank due to ethical ambiguity in addition to process uncertainty? We raise this question for extended consideration within the groundwater community. A possible direction which may support steps forward in this regard may be found in resolving conflicts between FAIR (findable, accessible, interoperable, reproducible) and CARE (collective benefit, ability to control, responsibility, and Indigenous ethics) data principles (Wilkinson et al., 2016; Carroll et al., 2020).

3.5.1 Addressing data limitations

Improving the global data landscape is a necessary endeavour to advance research agendas on groundwater systems as social-ecological systems. Through the classification schemes used in this study, we can identify a set of preferences for future data efforts that include: (1) observed over modelled data, (2) time series over static datasets, and (3) explicit consideration or representation of groundwater systems. Thus, future improvements of the existing data landscape can be readily identified by seeking to move data across the classes of this review (i.e., from static to time series, from explicit to implicit groundwater representation or consideration). For instance, existing data on groundwater-dependent ecosystem extents or gridded representations of area equipped for groundwater irrigation are currently only available in static forms yet would offer substantial insight into large-scale social-ecological system dynamics linking land use change, groundwater irrigation, and ecosystem functioning if available in temporally dynamic formats over consistent time ranges.

3.5.2 An aspirational future: essential variables and a global data cube

How can the global-scale groundwater data community organize to address these limitations and generate a more diverse global data landscape? Proposing such a strategy is necessarily a conversation and endeavour for the wider community to advance. Yet, to encourage the uptake of this conversation, we close by exploring one potential pathway through the linked concepts of essential variables and data cubes.

We view the concept of essential variables, such as pursued in climate (Bojinski et al., 2014), biodiversity (Pereira et al., 2013), ocean (Miloslavich et al., 2018), and ecosystem services (Balvanera et al., 2022), as one compelling way to organize and advance future data collection efforts. The essential variables concept is aimed at providing a minimum set of variables that provide sufficient monitoring to detect changes in the function and structure of a given system, and the word "essential" will have different interpretations for different systems and research communities. Groundwater storage change and water table levels are two of the 55 essential climate variables (GCOS, 2024), yet we view additional potential for a broader social-ecological configuration of groundwater essential variables.

Our small authorship team is inappropriate for setting these essential variables, which would necessarily be a community-wide and engaged process. Yet, to illustrate our envisioned potential for the concept, we provide a hypothetical set of essential variables for large-scale groundwater-connected systems (Figure 3.6). These variables span direct groundwater data (water table depth and storage anomalies) and data that span groundwater system functions across hydrological, ecological, and socioeconomic systems. While this set of variables does not contain data that explicitly represent social values, cultural practices, or meanings of groundwater, applying CARE and other ethical lenses remain a fundamental need as variables such as baseflow contributions and groundwater-dependent ecosystems are strongly linked with these considerations.

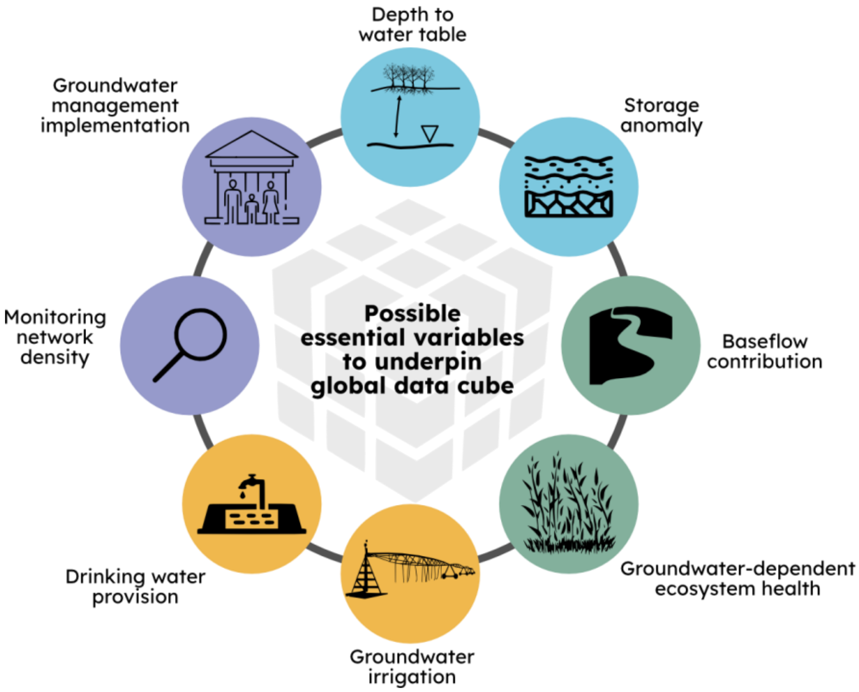


Figure 3.6. A possible set of essential variables for global groundwater systems.

All variables of this hypothetical set of essential variables have existing global-scale data. Developing such a set of groundwater essential variables could set in motion community-wide discussions on appropriate and practical data reporting frequencies that align with individual variable processes. For instance, it may be preferable to collect data on storage anomalies at the monthly time step but updating groundwater management implementation data may be more sensible at intervals that align with political cycles. Thus, implementing the groundwater essential variable concept would require pragmatic navigation across hydrologic, ecological, climate, and human timescales (Gleeson et al., 2012). Further justification of the potential set of essential variables is provided in Table III.3.

The development of a global groundwater data cube could complement this essential variables initiative, should it be pursued. Earth system data cubes are harmonized spatiotemporal datasets synthesized into a single "data stream" (Mahecha et al., 2020). Thus, data cubes directly facilitate the comparison and integration of multidimensional, spatially explicit data. Data cubes have been proposed for use in water security contexts (Bolognesi et al., 2018), yet to our knowledge the concept has yet to be specifically considered for global groundwater science and sustainability applications. We view the explicit linking of essential variables with the data cube concept (Giuliani et al., 2020) to represent a highly fruitful combination, which both provides a clarifying set of data collection objectives and creates an analysis-ready data product for use among the broader research community. Thus, in this way, the linking of essential variables with a data cube can serve interests of both data authors and data users better than the initiatives would in isolation. Furthermore, our view is that such an integrated effort could address several limitations and challenges in the existing global data landscape (see Table 3.3).

Table 3.3. An overview of how an essential variable and data cube initiative can address global groundwater data needs.

Global groundwater data needs	How an essential variable and data cube initiative can meet these needs
Establish priorities on global groundwater data collection	Identify a set of essential variables that balance data needs and observational capacities across research programs and management uses.
Address institutional bias in global data development	Develop an international network that balances institutions, geographies, and disciplines. Articulate and/or shift benefits of global groundwater data and science to encourage and facilitate participation of regions not represented in current global data authorship.
Navigate FAIR and CARE data principles	Co-develop and follow best practices on handling and sharing spatial data on land within Indigenous territories.
Generate more time series data with explicit consideration of groundwater systems	Develop observation capacities for all essential variables.
Synthesize and harmonize global data into analysis ready formats for groundwater scientists and managers	Integrate time series data of essential variables into an open-access data cube, and update at consistent frequency.

3.6 Conclusion

To support continental to global-scale research on groundwater systems as social-ecological systems, we identified and reviewed a large collection (n = 131) of open-access global datasets that directly, explicitly, or implicitly relate to shallow, terrestrial groundwater systems and their social-ecological system interactions. We reveal a rich variety of data are available for implementation in global studies which can serve as a reference resource for researchers to locate sources of interdisciplinary data. Yet, there are important limitations and biases in the existing data. At the forefront of these limitations is a lack of temporally dynamic datasets that explicitly incorporate groundwater considerations, undermining the ability of global groundwater science to generate a strong evidence base for social-ecological system dynamics. We also find the institutions producing these global datasets are overwhelmingly located in the global North, prompting questions about the potential mismatches in needs, interests, and incentives between groundwater data generation and groundwater data needs. We discuss the potential for a community-wide initiative to develop a set of essential variables underpinning a global data cube as one approach to organize future social-ecological groundwater data development. We argue that such an initiative, should it be explored, has the potential to address existing groundwater data limitations and biases while simultaneously providing an opportunity to engage with overarching technical and ethical challenges of collecting and sharing global data.

3.7 Open research

Environment	R project for statistical computing (R Core Team, 2023)
R packages	
tmap	Tennekes et al. (2023) https://cran.r-project.org/package=tmap
ggplot2	Wickham et al. (2024) https://cran.r-project.org/package=ggplot2
MetBrewer	Mills (2022) https://cran.r-project.org/package=MetBrewer
Script repository	https://github.com/XanderHuggins/groundwater-SES-data-review

3.8 Acknowledgements

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Chapter 4

GROUNDWATERSCAPES: A GLOBAL CLASSIFICATION AND MAPPING OF GROUNDWATER'S LARGE-SCALE SOCIOECONOMIC, ECOLOGICAL, AND EARTH SYSTEM FUNCTIONS

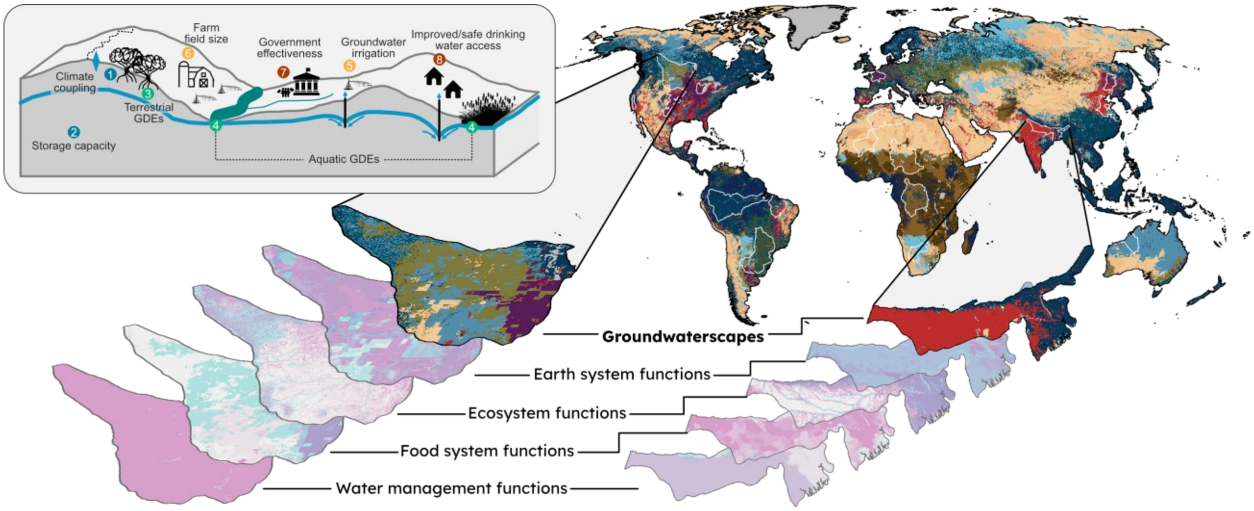
This chapter is in revision as a *research article* with *Water Resources Research*.

> Huggins, X., Gleeson, T., Villholth, K.G., Rocha, J.C. & Famiglietti, J. S. (2024). *Groundwaterscapes*: a global classification and mapping of groundwater's large-scale socioeconomic, ecological, and earth system functions. (In review.) Preprint: https://doi.org/10.31223/X5M382
> Submitted: 15 September 2023 – Revised 18 April 2024

Key points:

- Develops the concept of groundwaterscapes which are landscape units representing specific and broadly occurring configurations of groundwater's social-ecological and Earth system functions; and derives 15 groundwaterscapes at the global scale.
- All large aquifer systems of the world contain multiple groundwaterscapes, highlighting the functional diversity of groundwater in aquifers often treated as lumped systems in global groundwater assessments.
- Develops and applies a sequential and deeply iterative self-organizing map clustering methodology.

Figure 4.1. Graphical abstract.



4.1 Abstract

Groundwater is a dynamic component of the global water cycle with important social, economic, ecological, and Earth system functions. We present a new global classification and mapping of groundwater systems, which we call groundwaterscapes, that represent predominant configurations of large-scale groundwater system functions. We identify and map 15 groundwaterscapes which offer a new lens to conceptualize, study, model, and manage groundwater. Groundwaterscapes are derived using a novel application of sequenced self-organizing maps that capture patterns in groundwater system functions at the grid cell level (~10 km), including groundwater-dependent ecosystem type and density, storage capacity, irrigation, safe drinking water access, and national governance. All large aquifer systems of the world are characterized by multiple groundwaterscapes, highlighting the pitfalls of treating these groundwater bodies as lumped systems in global assessments. We evaluate the distribution of Global Groundwater Monitoring Network wells across groundwaterscapes and find that industrial agricultural regions are disproportionately monitored, while several groundwaterscapes have next to no monitoring wells. This disparity undermines the ability to understand system dynamics across the full range of settings that characterize groundwater systems globally. We argue that groundwaterscapes offer a conceptual and spatial tool to guide model development, hypothesis testing, and future data collection initiatives to better understand groundwater's embeddedness within social-ecological systems at the global scale.

4.2 Introduction

Conceptual models and classification schemes of groundwater systems traditionally focus on physical attributes and hydroclimatic setting (Margat & van der Gun, 2013; Winter, 2001) and primarily serve in support of fundamental hydrogeological investigations (e.g., as system boundaries for trend analyses in Richey et al., 2015; Shamsudduha & Taylor, 2020). Yet, recent years have witnessed a marked shift beyond traditional hydrogeology as interdisciplinary studies are increasingly conducted on global groundwater systems in response to the era of "human domination over the water cycle" (Abbott et al., 2019) and in recognition of groundwater system interlinkages with social, economic, ecological, and Earth systems (Gleeson et al., 2020; Huggins et al., 2023a). Yet, there is currently no set of guiding principles nor a globally consistent classification scheme through which to consider global groundwater systems as embedded within social-ecological systems (see Box 4.1 for key terminology). Here, we conduct a first attempt at filling this gap by producing a global, spatially explicit classification of groundwater systems on the basis of groundwater's large-scale socioeconomic, ecological, and Earth system functions.

The understanding of groundwater systems as dynamic components of social-ecological systems is propelled by the large and growing evidence-base documenting the functions the resource provides across social, economic, ecological, and Earth systems (Foster et al., 2013; Gleeson et al., 2020; Kuang et al., 2024; Scanlon et al., 2023). For instance, groundwater provides ~40% of global irrigation water (Siebert et al., 2010) and is an important, strategic buffer against increasing climate variability (Scanlon et al., 2023; Taylor et al., 2013). Groundwater supports ecosystems around the world in the form of groundwater-dependent ecosystems (Kløve et al., 2011; Link et al., 2023), which can take the form of aquatic, terrestrial, or subsurface ecosystems and that offer services of both ecological and cultural significance (Kreamer et al., 2015). Economically, groundwater is used in mining, manufacturing, energy generation, and agriculture, while simultaneously holding relational values such as through offering senses of place and identity in cultures around the world (Moggridge & Thompson, 2021; Griebler & Avramov, 2015). From an Earth system perspective, groundwater can be dynamically coupled to the atmosphere (Haitjema & Mitchell-Bruker, 2005), land-surface (Maxwell & Kollet, 2008), oceans (Luijendijk et al., 2020), and lithosphere (Konikow & Kendy, 2005).

Understanding how these diverse functions co-occur is an important first step in developing a more integrated, system-of-systems understanding of groundwater at the global scale. There are a handful of system-spanning global groundwater classifications, such as nation-scale groundwater economies (Shah et al., 2007), or classifications that map the mode of interaction between groundwater and the atmosphere (Cuthbert et al., 2019a). These existing studies focus on pairwise system interactions. Yet, to our knowledge, no study to date has developed a global groundwater system classification using a holistic framing that considers groundwater's socioeconomic and biophysical dimensions in equal depth or includes as wide a set of groundwater functions as we do here. As groundwater systems evolve under global change (Kuang et al., 2024), having such a baseline system classification can be useful as a reference with which to track changes between groundwater and its connected systems.

Outside the groundwater literature, a variety of global social-ecological system typologies have been developed in recent decades. These studies include the development of global anthromes (Ellis & Ramankutty, 2008), land system archetypes (Václavík et al., 2013), dryland vulnerability patterns (Kok et al., 2016; Sietz et al., 2011); and an even wider assortment of typologies at continental and regional scales (Beckmann et al., 2022; Rocha et al., 2020; Van Vliet et al., 2012; van der Zanden et al., 2016). Yet, these underlying concepts and methods have yet to be applied to groundwater systems.

The emerging field of archetype analysis is a central, driving force behind these social-ecological system characterisations (Eisenack et al., 2021). In this literature, an archetype is understood as "a mental representation of relationships between attributes and processes that characterize systems" (Eisenack et al., 2019). Archetype analysis is explicitly sustainability-oriented and seeks to identify "recurrent patterns of [a] phenomenon of interest at an intermediate level of abstraction to identify multiple models that explain the phenomenon under particular conditions" (Oberlack et al., 2019). While many methods have been used to perform archetype analysis (Sietz et al., 2019), a "full" analysis typically consists of a configuration of attributes, an underlying theory to explain these configurations, and empirical cases where this theory holds (Oberlack et al., 2019). Indeed, many of the social-ecological system typologies referenced above explicitly use an archetype analysis language and framing.

In this study, we apply the recently developed framing of groundwater-connected systems (Huggins et al., 2023a; Chapter 2) and implement a cluster analysis methodology consistent with spatial archetype analysis (Sietz et al., 2019) to develop a global typology of large-scale groundwater system function configurations. We focus on large-scale functions, which we understand as functions that broadly occur across regional extents (order ~10^4 km^2 and larger) and that are conducive to global, systematic pattern identification. We name the clusters that emerge from this process as groundwaterscapes (Box 4.1). These groundwaterscapes offer a first step towards characterizing the predominant configurations and spatial patterns within groundwater's socioeconomic, ecological, and Earth system functions, which we believe can offer widespread potential uses and benefits across groundwater science and management.

Box 4.1. Key terminology.

Social-ecological systems: Integrated systems formed by social and biophysical system interactions (Berkes & Folke, 1998). Investigations that use social-ecological system framings seek to understand how society and the environment are intertwined and co-evolved systems.

Groundwater-connected systems: Systems that are formed through interactions between social, ecological, and Earth systems with physical groundwater systems. Groundwater-connected systems are understood as specific forms of social-ecological systems (Huggins et al., 2023a).

Groundwaterscapes: A landscape unit with a specific and broadly occurring configuration of groundwater-connected system functions. In this work, we derive groundwaterscapes using global datasets representing the Earth system, ecosystems, food system, and water management system functions included in our conceptual model (Figure 4.2) but groundwaterscapes could be defined with other methods and data.

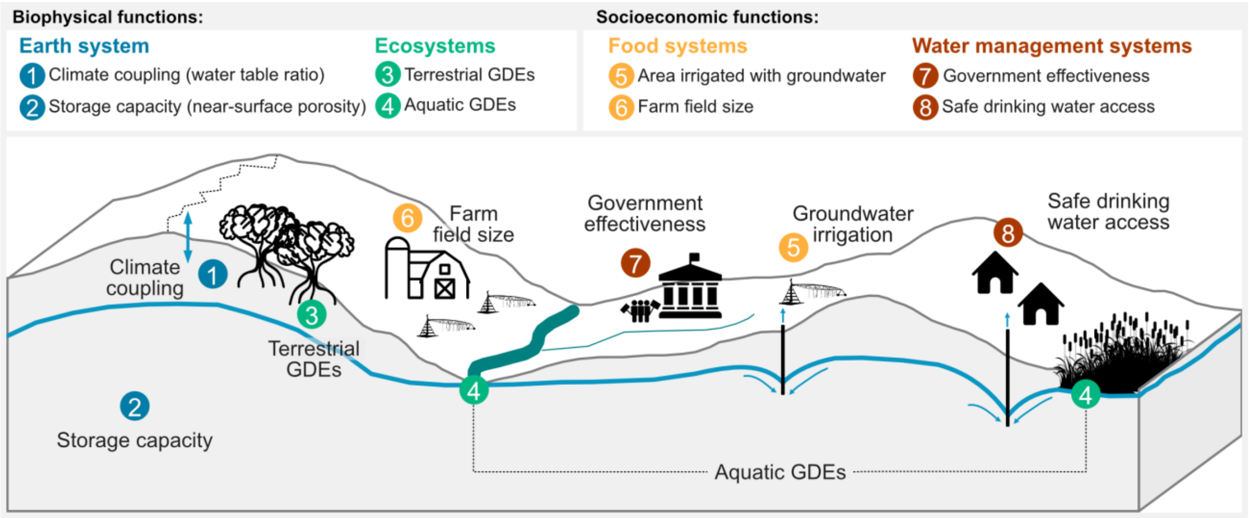


Figure 4.2. Groundwaterscape conceptual model, consisting of groundwater's large-scale Earth system, ecosystem, food system, and water management system functions.

Maps of the input data representing these functions are shown in Figure 4.3.

4.3 Materials and methods

4.3.1 Conceptual model

Drawing on recent reviews of global groundwater systems (Gleeson et al., 2020; Lall et al., 2020; Scanlon et al., 2023), we identified four core systems that groundwater interacts with at large spatial scales and that balance representation of biophysical and socioeconomic functions: Earth systems, ecosystems, food systems, and water management systems (Figure 4.2). We distinguish between biophysical and socioeconomic functions following the Social-Ecological Systems Framework (Ostrom, 2009), which argues for such a balanced approach when conceptualizing a social-ecological system (Binder et al., 2013). We included an equal number of functions (2) per system to ensure even representation in our analysis. In order to be included in our conceptual model, individual functions required a strong conceptual foundation in the large-scale groundwater literature and required global quantification in an existing dataset. This number of input datasets (8) is within the range of input layer counts commonly found in existing social-ecological system clustering studies. We found this number of input layers to include sufficient data to characterize our conceptual model while not being overly numerous to render the process of assessing and disentangling classification results intractable. Maps of the system functions included in our conceptual model are shown in Figure 4.3.

For groundwater's *Earth system* functions (Figure 4.3a), which represent groundwater's interactions with the atmosphere, land, lithosphere, and oceans (i.e., Earth system components), we focus on groundwater's climate and storage functions. Groundwater is increasingly studied through an Earth system lens (Gleeson et al., 2020), and is recognized as a critical resource that affects overall Earth system resilience (Rockström et al., 2023). Water table depth is an important control on the land-atmosphere energy balance (Maxwell & Kollet, 2008). In areas with shallow water tables, groundwater is tightly coupled with land surface and energy processes (i.e., a bidirectional mode of interaction occurs with both groundwater recharge and evapotranspiration fluxes), and this coupling dissipates with deeper water tables and becomes recharge-dominated (i.e., a unidirectional mode). We use the water table ratio, a dimensionless criterion that classifies the mode of groundwater-climate interactions as bidirectional or unidirectional (Haitjema & Mitchell-Bruker, 2005) to represent groundwater's hydroclimatic function (Cuthbert et al., 2019a). Secondly, as the largest store of unfrozen freshwater globally, groundwater provides important storage functions (Gleeson et al., 2020). Net groundwater storage loss is a secondary contributor to global sea level rise (Konikow, 2011) while groundwater's large storage capacity also provides important retention and attenuation functions in the water cycle (Opie et al., 2020). Thus, groundwater naturally serves as an important control on hydrological processes such as drought (Van Lanen et al., 2013). As groundwater storage, particularly at depths that are dynamically connected to the Earth system, is challenging to quantify (Condon et al., 2020; Ferguson et al., 2021), we use shallow subsurface porosity (representative for depths on the order of 100m) as a proxy representation of groundwater storage capacity (Gleeson et al., 2014).

To represent groundwater's *ecosystem* functions (Figure 4.3b), we consider the type and density of groundwater-dependent ecosystems (GDEs). GDEs are terrestrial, aquatic, or subterranean ecosystems that rely on groundwater for some or all of their freshwater needs (Kløve et al., 2011). We focus on terrestrial and aquatic GDEs as these ecosystems are more closely coupled to land-surface processes, are better understood relative to subterranean GDEs, dominate conservation and management dialogues (Rohde et al., 2017; Saito et al., 2021), and have benefitted from recent global mapping efforts (Huggins et al., 2023b; Link et al., 2023). Terrestrial GDEs exist where root systems source groundwater and thus rely on the subsurface presence of groundwater while aquatic GDEs rely on surface expressions of groundwater and include rivers, streams, and wetlands.

Groundwater is a critical resource for a wide array of economic functions, including uses in mining, manufacturing, energy generation, and agriculture sectors. In this study, we focus exclusively on agriculture which is the primary sector driving groundwater consumption globally (Giordano & Villholth, 2007; Wada et al., 2012). Thus, to reflect groundwater's *food system* functions (Figure 4.3c), we consider the extent of areas irrigated with groundwater and farm field size. Including groundwater irrigation patterns enables this analysis to differentiate regions based on agricultural reliance on groundwater. Secondly, though not often incorporated in groundwater studies, field size is a key attribute of agricultural systems that is associated with many functional differences in groundwater interactions, livelihoods, agricultural practices, and productivity (Meyfroidt, 2017). For instance, small scale farms, especially in developing countries, are less likely to have access to basic services, infrastructure, and mechanization (Meyfroidt et al., 2022), whereas large, irrigated farms are generally associated with greater productivity and higher levels of economic development (Meyfroidt, 2017). Field size, which is related to farm size (Graesser & Ramankutty, 2017; Lesiv et al., 2019), is additionally important in relation to land tenure and water and land management dynamics. For instance, a management area will have considerably more actors, a greater mosaic of land ownership, and thus a more complex management setting in regions with smaller farms in comparison to if the same area were covered by larger farms. Case studies have also identified that farm size is associated with participation rates and dynamics in collaborative management processes (Amblard et al., 2023; Dobbin, 2020). Thus, incorporating field size is a pragmatic, coarse approach to represent qualitative differences in industrial versus smallholder agricultural systems.

Our inclusion of *water management system functions* (Figure 4.3d) is an effort to represent what actions are taken “within governance [frameworks] related to the development and protection of groundwater” (Villholth & Conti, 2018). Our included water management system functions aim to represent societal forms of interaction with groundwater resources expressed through policy measures, collective action, priority setting, and service provision. Inversely, societal interactions with groundwater systems form values and worldviews that in turn can shape water management practices. In the absence of a global dataset on groundwater or water governance and management, we use a general indicator of nation-scale government effectiveness as proxy representation of national water governance. Thus, we consider general government effectiveness to be a foundation of good water governance, which reflects the quality of public services, the degree of independence from political pressures, the quality of policy formulation and implementation, and the credibility of government commitment to these policies (Kaufmann and Kraay, 2023). In incorporating this government effectiveness layer, we follow Varis et al. (2019) who used this same dataset to characterize the governance dimension of social adaptive capacity within river basins in a global analysis. We note that initial steps are being taken to monitor groundwater governance directly (e.g., https://iwrmdataportal.unepdhi.org/), however the process of comprehensively generating and validating these data globally remains an on-going challenge that prevents their inclusion in this study. More broadly, it is not straightforward to quantify governance and management dimensions and the process of doing so is often contested (Thomas, 2010). Furthermore, water governance frameworks, organizations, and actions will vary sub-nationally depending on local and regional hydrological and political context. Thus, our inclusion of national government effectiveness informs on the general governance setting but sub-national variations in governance are invisible to this study. Despite these limitations, we view the inclusion of this governance dimension as a crucial component of our analysis that ensures this first assessment of groundwaterscapes reflects the broad scope of the groundwater-connected systems framing and creates a baseline that can be refined in future studies.

Secondly, to consider the role of groundwater management in relation to safe water access, equity, and the domestic services of groundwater, we integrate fundamental data on the percentage of people that collect or use unimproved drinking water. This unimproved drinking water can come from many sources, including an unprotected dug well or spring, or alternatively from surface water sources such as a river, pond, or canal. Data that disaggregate these sources of unimproved drinking water do not exist to the best of our knowledge. We view this indicator as a useful representation of groundwater's utilisation, or lack thereof, in supporting domestic activities and water security.

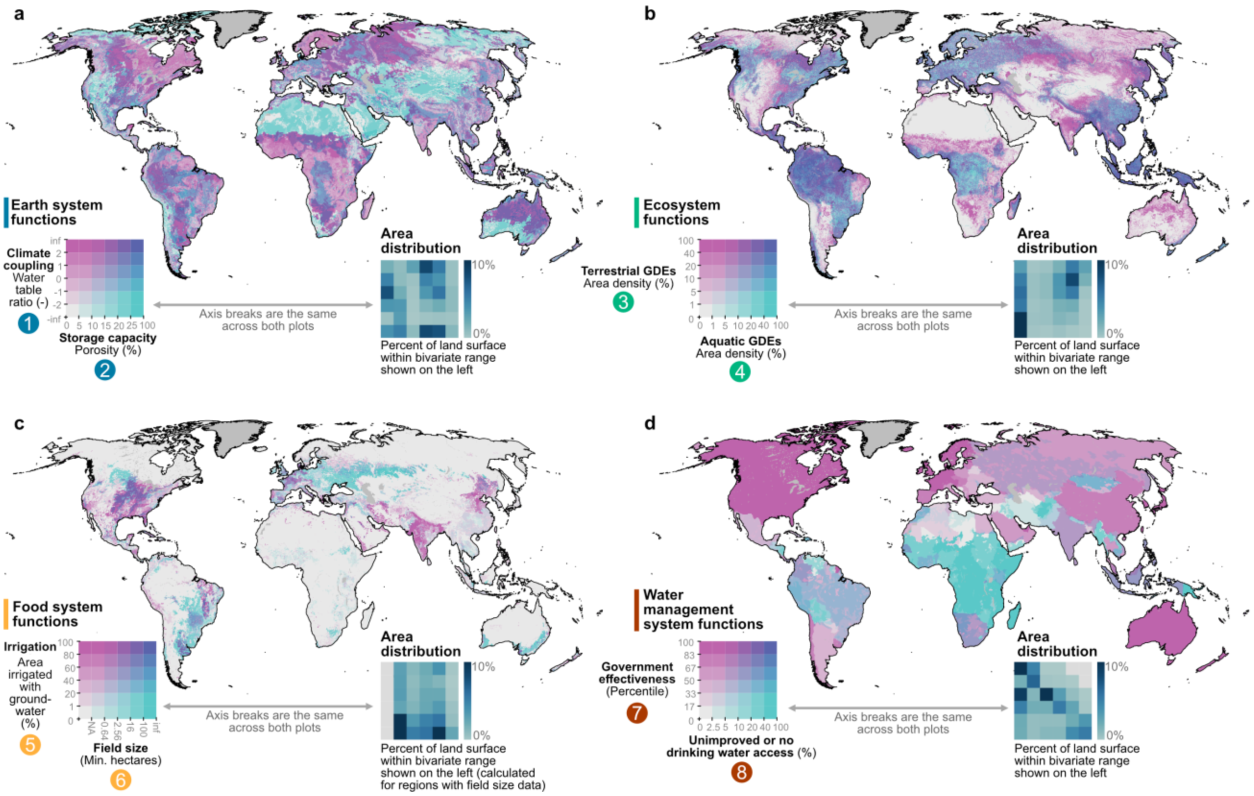


Figure 4.3. Exploratory mapping of groundwater's large-scale (a) Earth system, (b) ecosystem, (c) food system, and (d) water management system functions.

Bivariate legends are numbered accordingly with conceptual model elements show in Figure 4.2. The area distribution of each mapped bivariate relationship is shown by inset heatmaps which have the same axis breaks shown in each map's bivariate legend.

4.3.2 Spatial resolution and preprocessing

We conduct all analyses at 5 arcminute resolution (~10 km grids near the equator). This produces a moderate-resolution global groundwaterscape map that balances the base resolutions of input datasets (Table 4.1) and is compatible with a wide array of global hydrological models (e.g., Burek et al., 2020; Sutanudjaja et al., 2018) and freshwater-focused social-ecological studies (e.g., Varis et al., 2019, Gain et al., 2016). Secondly, operating at the unit of 5 arcminute grid cells rather than aquifers, basins, or administrative units enables analysis of groundwaterscape heterogeneity in these systems (see 4.3.4 Post hoc analysis).

All input datasets were preprocessed to generate a spatially harmonized raster stack at 5 arcminute resolution. Each raster layer was subsequently normalized such that grid cell distributions held the properties of zero mean and unit variance. Two exceptions were made for the water management system data which were normalized at the nation and watershed scale, to match the scale at which they were respectively derived, before rasterization was conducted. We subsequently applied feature clipping by setting minimum and maximum values at +/-2 standard deviations away from the mean to ensure that extreme outliers within individual data layers did not exert an outsized impact on groundwaterscape results. The study domain was defined by a common global earth mask (Wessel et al., 2019; Wessel & Smith, 1996) and further excluded Greenland and Antarctica given low data coverage across these regions. Sources, descriptions, and summaries of preprocessing steps for each dataset are provided in Table 4.1.

Table 4.1. Input datasets. Maps and histograms of each dataset are shown in Figure IV.1.

Dataset	**Data source, information, and preprocessing**
Water table ratio	**Data source**: Cuthbert et al. (2019b) **Persistent web-link**: https://doi.org/10.6084/m9.figshare.7393304.v8 **Spatial resolution**: 1 km **Temporal range**: Ca. 2000 **Harmonisation:** Bilinear resampling to 5 arcminute resolution. **Additional preprocessing:** Regions with recharge <5 mm yr-1 were set to the minimum normalised value following Cuthbert et al. (2019a) who removed these regions given the variable's sensitivity to low recharge rates. We adopted this approach to reflect how arid regions typically have deep water tables with minimal evapotranspiration fluxes from groundwater. We used the same recharge dataset (Döll & Fiedler, 2008) as used in Cuthbert et al. (2019a) to apply this mask.

Near-surface porosity	**Data source**: Gleeson et al. (2018) **Persistent web-link**: https://doi.org/10.5683/SP2/DLGXYO **Spatial resolution**: Polygons with average size of ~14,000 km^2 **Temporal range**: N/A **Harmonisation:** Vector polygon rasterization to 5 arcminute resolution.
Groundwater-dependent ecosystem types (both aquatic and terrestrial)	**Data source**: Huggins et al. (2023c) **Persistent web-link**: https://doi.org/10.5683/SP3/P3OU3A **Spatial resolution**: 30 arcsecond **Temporal range**: ca. 2015 **Harmonisation:** Area density calculated per 5-arcminute grid cell.
Area irrigated with groundwater	**Data source**: Siebert et al. (2013) **Persistent web-link**: https://www.fao.org/aquastat/en/geospatial-information/global-maps-irrigated-areas/latest-version/ **Spatial resolution**: 5 arcminute **Temporal range**: 2005 **Harmonisation:** None
Farm field size	**Data source**: Lesiv et al. (2018) **Persistent web-link**: https://pure.iiasa.ac.at/id/eprint/15526/ **Spatial resolution**: ~1 km **Temporal range**: ca. 2010-2016 **Harmonisation:** Modal resampling to 5 arcminute resolution.
Government effectiveness	**Data source**: Worldwide governance indicators (Kaufmann & Kraay, 2023). **Persistent web-link**: www.govindicators.org. **Spatial resolution**: Nation scale **Temporal range**: 2020 **Harmonisation:** Vector polygon rasterization to 5 arcminute grids.
Unimproved drinking Water	**Data source**: World Resources Institute's Aqueduct Water Risk Atlas (Kuzma et al., 2023) **Persistent web-link**: https://www.wri.org/data/aqueduct-global-maps-40-data **Spatial resolution**: HydroBASIN Level 6 **Temporal range**: 2015 **Harmonisation:** Vector polygon rasterization to 5 arcminute resolution.

Before performing the groundwaterscape derivation, we first evaluated the collinearity of the eight normalised input datasets by calculating Pearson correlation coefficients on a random sample of 40,000 grid cells (~2% of all grid cells within study domain) to avoid impacts of spatial autocorrelation (cf. Beckmann et al., 2022; Václavík et al., 2013). There are moderate levels of collinearity ($r^2 \approx 0.5$) between certain inputs, such as between aquatic and terrestrial GDE density and between government effectiveness and safe drinking water access (Figure IV.2), but no correlation values were sufficiently high to require further modification when using common thresholds to evaluate detrimental levels of collinearity ($r^2 > 0.7$) (Dormann et al., 2013).

4.3.3 Iterative self-organizing maps to derive groundwaterscapes

Social-ecological system classification has no consensus methodology (Sietz et al., 2019) and can be approached from either top-down or bottom-up perspectives. Bottom-up classification begins with individual case studies and groups cases together based on similarity in system composition or behaviour. These approaches are contextually rich but can be geographically or contextually limited based on spatial extent or case study count and diversity. Conversely, top-down approaches begin with spatially distributed datasets and derive recurring patterns using a variety of approaches such rule-based classification or cluster analysis. Top-down approaches provide a wider and more consistent spatial coverage in comparison to bottom-up approaches but can be limited by the quality of data used to represent system attributes and by bias in the data selection process. Thus, top-down approaches are more common among regional to global scale assessments. However, the two methodologies may support each other in mixed-method processes (Sietz & Neudert, 2022), where bottom-up approaches can aid in ground-truthing insights derived from top-down methods (Eisenack et al., 2021).

Here, we use an iterative and sequential self-organizing map (SOM) methodology to derive groundwaterscapes. SOMs are a form of unsupervised artificial neural network that perform a unique type of data quantization (Kohonen, 2013). SOMs work by projecting an n-dimensional input data space onto a low dimensional (typically two-dimensional) grid of nodes, where each node contains an n-dimensional “codebook” vector representing a contiguous region in the input data space. Nodes with similar codebook vectors are located closer to each other in this low dimensional grid and dissimilar codebook vectors further apart. SOMs are thus a particularly powerful method for data exploration and visualization as the low-dimensional grid of nodes preserve the topology of the input data and as so have been widely used to address clustering problems (Flexer, 2001; Kohonen, 2013; Vesanto & Alhoniemi, 2000), including the classification of social-ecological systems (Beckmann et al., 2022; Jung et al., 2024; Levers et al., 2018; Václavík et al., 2013; van der Zanden et al., 2016). SOMs are further advantageous for clustering applications as they are less prone to identifying local optima relative to other approaches (Bação et al., 2005). As the method does not require the specification of any parameter thresholds to determine clusters, it is considered as a clustering method less prone (but not immune) to researcher bias (Sietz et al., 2019).

A common strategy to conduct SOM-based clustering is to perform cluster analysis on the generated set of codebook vectors as this approach has the additional benefit of identifying complex cluster structures (Taşdemir et al., 2012; Delgado et al., 2017). We implement a similar methodology in this study by following Delgado et al. (2017) and perform a two-staged clustering methodology that implements SOMs at both stages of the clustering process (Figure 4.4). The first stage of this methodology develops a two-dimensional SOM to generate a vector quantization of the input data space that is substantially smaller but topologically similar to the original input data. The second stage of this method uses the codebook vectors of the first-stage SOM as input data and develops a one-dimensional SOM whose vector quantization derives the clusters we present as groundwaterscapes. In each stage of this methodology, we iterate across a wide range of SOM grid sizes and select the best performing size based on a set of performance metrics (see below). In recognition of the stochastic property of SOMs, we develop a set of alternative of SOMs at each grid size and filter-out performance outliers to improve reproducibility (see below).

First-stage SOM methods: For the first-stage SOM iterations, we follow Delgado et al. (2017) and set the minimum SOM grid size ($S \text{ x } S$) as: $S_{min} = \sqrt{2N^{0.4}}$, where *N* is the number of patterns in the input data, and set the maximum SOM grid size as $S_{max} = \sqrt{0.15N}$. We iterate from: S_{min} to S_{max} in increments of 2. In determining *N*, which was originally intended to represent the number of unique input data points (as in Delgado et al., 2017) to be infeasible at our spatial resolution (>2 million grid cells, thus 2 million input features) as the approach suggests grid sizes far greater than are commonly found in similar SOM applications in the literature. Thus, to pragmatically estimate *N*, we iteratively performed k-means clustering on our input data until 99% of the input data variation (within cluster sum of squares relative to total sum of squares) is represented by these clusters. This criterion was met at k = 12,000 clusters, and thus this *k* was used to estimate the number of patterns in the input data (i.e., *N*) which then set the range of first-stage SOM grid sizes tested (S_{min}= 10, S_{max}= 42). We generated a set of 60 alternative SOM models for each S from S_{min}, $S_{min} + 2$, $S_{min} + 4$, …, S_{max} (1,020 SOM models across all grid sizes). As this procedure was designed to guide identification of the optimal first-stage SOM grid size, we deemed it unnecessary to develop these SOMs on the full input data (>2 million data points) and instead conducted this step using the synthetic representation of the data space generated by our k-means cluster centers. This process identified the SOM grid size S = 22 best balanced SOM-specific and general clustering performance metrics (see below). With this optimal grid size identified, we then developed a set of 60 alternative SOMs at S = 22 using the full set of input features and selected the best performing model using performance metrics as described below. The codebook vectors from his best performing model yield a set of 484 features that reflect the underlying structure of the input data (Figure IV.3) and offer an intermediate classification level.

Second-stage SOM methods: The codebook vectors from the selected first-stage SOM became the input features for the second-stage SOM models. During this second stage, we followed Delgado et al. (2017) and iterated across one-dimensional SOM grid sizes so that models that determine prime numbers of clusters can be evaluated. For these second-stage SOMs, we set a minimum size (1 x S) of S_{min}= 2, and a maximum size of S_{max}= 30 following the upper limit of classes to identify as recommended for archetype analysis (Eisenack et al. 2019). As the input feature space is considerably smaller in this second stage, we generate a set of 120 alternative SOM models for each grid size S from S_{min}, $S_{min} + 1, S_{min} + 2,$ …, S_{max} (3,480 SOM models across all grid sizes). The best-performing SOM model from this set produces the groundwaterscapes presented in this study. The crisp (e.g. mutually exclusive) classification provided by our method (where each grid cell is associated with a single node in the selected first-stage SOM model, and each of these first-stage SOM nodes is associated with a single node in the selected second-stage SOM model) enables a simple reclassification of geospatial grid cells to their respective groundwaterscape.

SOM performance metrics: For the first-stage SOM models, we calculated performance using the SOM-specific Kaski-Lagus error function (Kaski & Lagus, 1996) and the clustering-specific Davies–Bouldin index (Davies & Bouldin, 1979). The Kaski-Lagus error function combines aspects of quantization error (average squared distance between input features and their assigned codebook vector) and topographic error (an indicator of how well the input data's topography is preserved in the SOM based on the share of total input features whose assigned and second-closest SOM node codebook vectors are neighbours within the SOM node grid). Conversely, the Davies-Bouldin index is a measure of both the compactness of individual clusters and the separation between clusters. To compare these performance metrics across SOM iterations, we min-max normalized each metric so that each had an equal influence on the performance evaluation. The SOM model with the minimum combined performance score is selected as the best-performing model.

For the second-stage SOM, we continued to use the same Kaski-Lagus error function and Davies-Bouldin Index and additionally included two more metrics. The first is the percentage of unexplained variation, which we were drawn to include based on our observation that there was significantly lower range of explained variance in the second-stage SOMs at small grid sizes that were not captured by the Kaski-Lagus error function due to topographic performance trade-offs. This variation-based performance metric was thus equally weighted with the Kaski-Lagus error function when deriving the second-stage SOM performance scores.

The second additional performance metric is a size preference metric that was included to quantitatively reflect our preference of identifying a manageable number of system classes (i.e., preferencing fewer clusters should performance metrics otherwise be similar). Our inclusion of this size preference metric stems from our observation that SOM results can show similar performance across a wide range of SOM grid sizes and thus could benefit from additional discrimination by explicitly embedding this size preference in our derivation methodology. To accomplish this, we superimpose a trapezoidal function (set to preference cluster counts that are equal to and greater than an a priori estimate of the best number of partitions in the data) and the logarithm of the number of clusters (set to preference a lower number of clusters, based on Varshney & Sun, 2013). This a priori best estimate of cluster partitions is determined by taking the median value across 30 different clustering indexes that estimate the optimal number of clusters in a dataset (Charrad et al., 2014) and is an approach that has been used to inform previous social-ecological system clustering (Rocha et al., 2020). The result is a curve resembling a piecewise function with its minimum located at this a priori estimate (Figure IV.4). We do not use this size preference function with equal weighting to the SOM- and cluster-specific performance metrics, but rather as an additional consideration in a sensitivity analysis to assist our decision-making process (see below). While other SOM-based studies take simpler approaches to identify the optimal number of clusters, such as visually identifying the “elbow” in the within-cluster sum of squares (Beckmann et al., 2022), we view our method as a more elaborate but reflective approach consistent with our underpinning values and objectives for this study.

Reproducibility and sensitivity analyses: To increase the reproducibility of this approach given the stochastic nature of SOMs, we filter and remove performance outliers within alternative SOM models at each grid size. The threshold to detect outliers per SOM size is established using the median absolute deviation (MAD) of individual and combined performance metrics. We thus removed outlier models for each grid size if any of the SOM's individual performance metrics or integrated performance metric was outside the respective MAD from the size-specific median performance value. We found this approach to lead to highly reproducible results across successive runs of our clustering scripts.

In our second-stage SOM performance evaluation, we perform a bivariate sensitivity analysis to better understand possible trade-offs between study reproducibility, clustering performance, and cluster count preferences. To do this, we identify the best-performing SOM across the set of alternative models while varying (i) the allowable limit of performance deviation and (ii) the weight given to the size preference function relative to the other performance metrics. The resulting matrix reveals the trade-offs embedded in this clustering process and enables a transparent selection across alternative local optimal models that best fit the needs of the study. As 15 clusters are proposed across the majority of sensitivity analysis combinations (Figure 4.4, panel 6), a second-stage SOM model of grid size 1x15 was selected as the optimal solution to this clustering problem.

4.3.4 Post hoc analysis

We calculated several landscape metrics to evaluate the spatial distribution of the groundwaterscapes within the large aquifer systems of the world (Richts et al., 2011). These metrics include the area distribution, Simpson's evenness index (Simpson, 1949), the contagion index (Riitters et al., 1996), marginal entropy, and relative mutual information (Nowosad & Stepinski, 2019) of groundwaterscapes. Simpson's evenness index is a diversity metric that represents if groundwaterscapes are evenly distributed within the aquifer (index is high) or if a few groundwaterscapes dominate the area (index is low). The contagion index is an aggregated metric that represents the likelihood that two adjacent grid cells belong to the same groundwaterscape. Marginal entropy measures the thematic complexity of groundwaterscapes within an aquifer, while relative mutual information has been shown as a useful approach to differentiate landscape patterns that otherwise show similar levels of complexity (Nowosad & Stepinski, 2019). Calculating these metrics within the large aquifer systems of the world facilitates the exploration of spatial patterns of groundwaterscapes in these aquifer systems and can enable aquifer grouping based on their groundwaterscape composition.

Lastly, we compared the groundwaterscape map with the location of monitoring wells in the Global Groundwater Monitoring Network (GGMN) (IGRAC, 2024). While the GGMN is a participative initiative and thus does not reflect all monitoring wells worldwide, it is the best-available open dataset of global groundwater monitoring well locations. To assess the coverage of monitoring wells across groundwaterscapes, we calculate both the number of monitoring wells found within each groundwaterscape as well as the monitoring well area density per groundwaterscape.

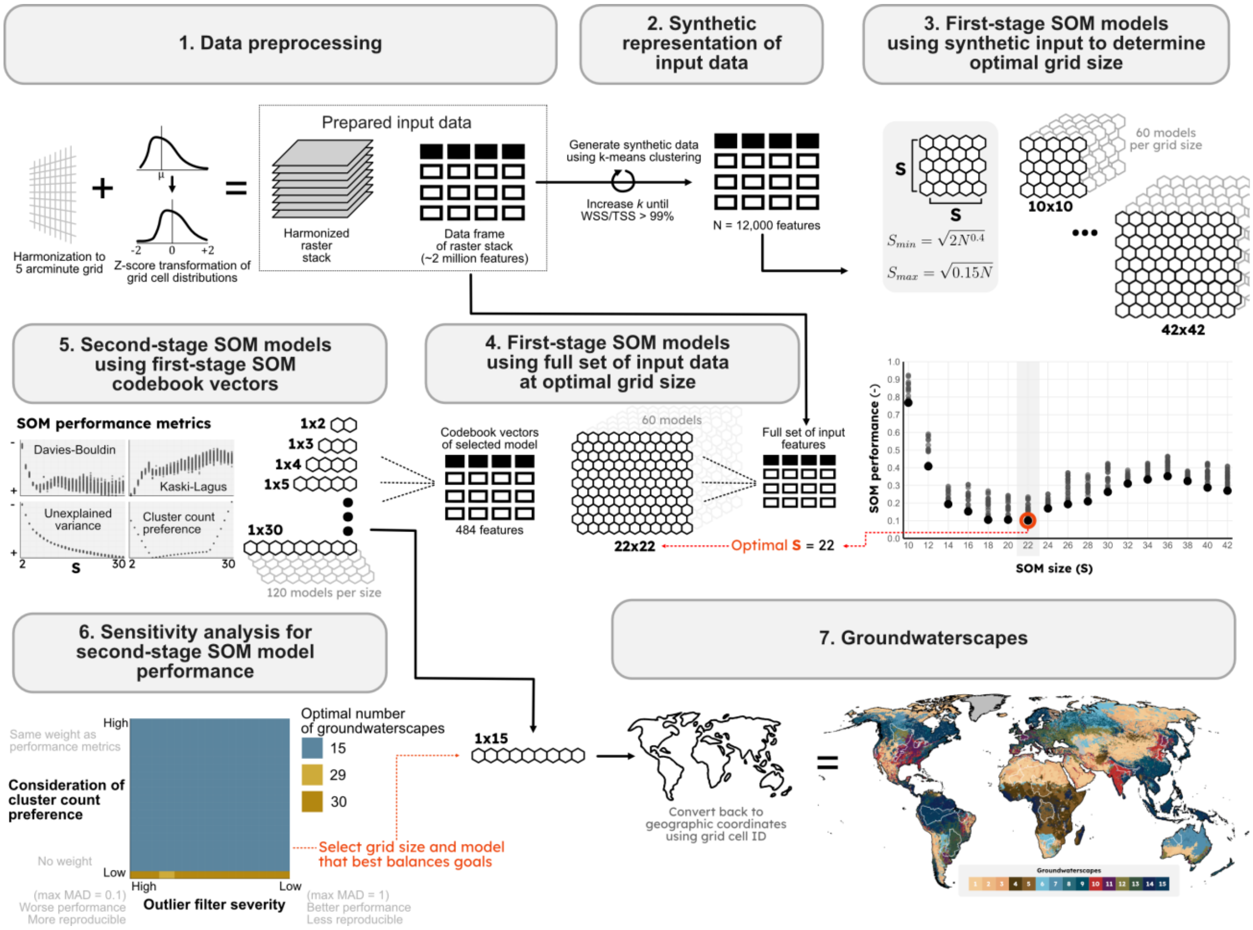


Figure 4.4. Groundwaterscape derivation.

Numbered panels denote individual components of the method.

4.4 Results and discussion

4.4.1 Global groundwaterscapes

Our classification method identifies and maps a set of 15 groundwaterscapes (Figure 4.5). Each groundwaterscape represents a unique configuration of the Earth system, ecosystem, food system, and water management system functions included in our conceptual model. These characteristic configurations of function magnitudes are visualized for each groundwaterscape using radial plots (Figure 4.5d,e).

We find that groundwaterscapes span spatially contiguous regions and capture broad patterns visible in the individual and underlying groundwater functions (e.g., as shown in Figure 4.3). The largest groundwaterscape by surface area (GS1) represents arid and desert environments such as the Central Basin (USA) and the Gobi Desert (China) which have large storage capacities amid minimal other functions and covers 18% of the land surface assessed in this study (Figure 4.5b). By contrast, the smallest groundwaterscape (GS11) represents industrial agricultural regions dependent on groundwater and is found in the American Midwest, California’s Central Valley, and Argentinian Pampas, the Paris Basin, and in Northeastern China and covers 3% of the land surface. All of the remaining groundwaterscapes cover between 5 to 9% of the land surface (Figure 4.5b). Groundwaterscape are described individually in Table 4.2 and the extent of individual groundwaterscapes are mapped in Figure IV.5. Figure IV.6 shows the interquartile range of function magnitudes within each groundwaterscape to supplement the radial plots shown in Figure 4.5.

We group the 15 groundwaterscapes into seven groups (Figure 4.5e), as guided by agglomerative clustering (Figure 4.5c), to provide a simplified version of this classification scheme and to enable easier interpretation of groundwaterscape differences. Groundwaterscape groups share overarching similarities but differ on a subset of functions. For instance, GS14 and GS15 (group VII) are identified as landscapes that have extensive aquatic and terrestrial GDEs and have limited agricultural functions and generally ineffective national governance but differ in storage capacity. Similarly, GS10 and GS11 (group V) are characterized as agricultural regions dependent on groundwater irrigation, yet GS10 is characterized by smallholder farms whereas GS11 is characterized by large-scale, industrialized agriculture. Of the 15 groundwaterscapes, 11 describe non-agricultural regions (groups I-IV, VII), while agricultural areas are described by four groundwaterscapes (groups V and VI).

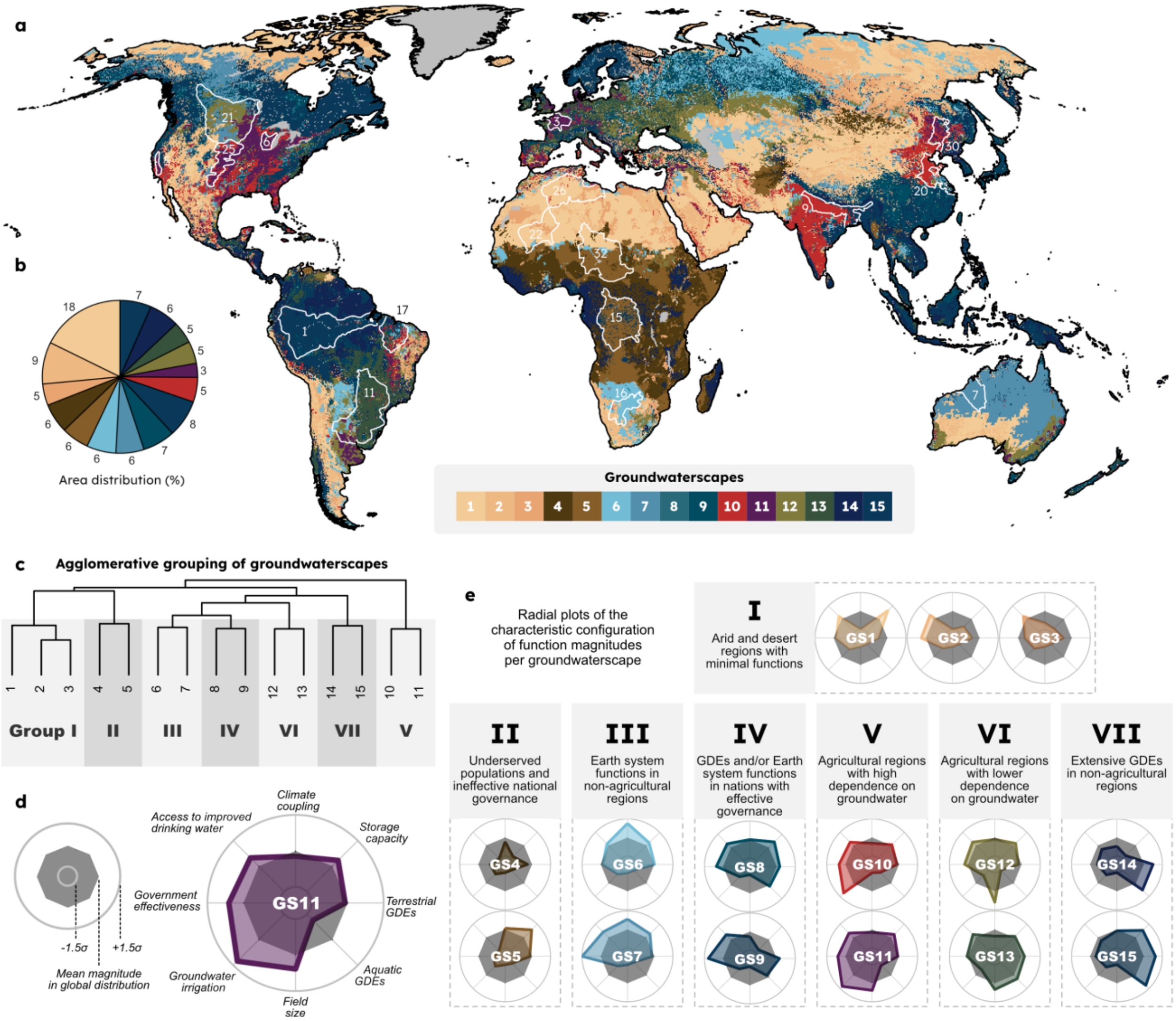


Figure 4.5. Global groundwaterscapes.

(a) Map of the 15 derived groundwaterscapes. White polygon outlines and annotated numbers represent aquifer systems and aquifer IDs that are shown in subsequent figures. (b) Area distribution of groundwaterscapes. (c) Agglomerative grouping of groundwaterscapes. (d) Radial plot legend. (e) Radial plot of function magnitudes per groundwaterscape. Figure IV.6 shows the interquartile range of function magnitudes for each groundwaterscape.

We find that any grid cell of a given groundwaterscape is most likely to neighbour with grid cells of the same groundwaterscape (Figure IV.7). Given that geographic location was not considered in our derivation methodology yet groundwaterscapes are found in contiguous patches suggests that our classification approach successfully identifies and reflects broad and contiguous patterns in the groundwater functions included in our conceptual model. Yet, not every grid cell is represented in equal fidelity by this classification scheme as some grid cells have function configurations that more closely mirror their groundwaterscape model than others. To represent this 'fit' of groundwaterscape classification at the grid cell level, we plot the Z-score of grid cell residual magnitudes per groundwaterscape (Figure IV.8). We find some regions to correspond tightly with their groundwaterscape representation such as the Amazon, central USA, and the Greater Sahel region. Other regions, such as the Congo basin have functional configurations with relatively large residuals from their associated groundwaterscape model and could benefit from an investigation of "nested" groundwaterscapes (cf. Sietz et al., 2017) to further differentiate and describe groundwater systems in these regions. Using the intermediary codebook vectors produced through the first-stage SOM (and as included in our data deposition) provide a sub-groundwaterscape classification that could be used for this purpose. However, we leave such recursive groundwaterscape derivations and investigations for future study.

4.4.2 Groundwaterscapes to facilitate systems thinking on global groundwater

To illustrate how groundwaterscapes capture patterns across the underlying functions considered in our conceptual model, we look to five large aquifer systems and visualize the distribution of groundwaterscapes side-by-side with Earth system, ecosystem, food system, and water management system functions (Figure 4.6). For instance, we can observe how the Northern Great Plains Aquifer (Figure 4.6b) contains a mosaic of groundwaterscapes with GS12 (industrial agriculture with low-moderate groundwater use) characterizing the central and western extents of the aquifer and GS11 (industrial agriculture with high groundwater use) found across its southeastern regions. In addition to reflecting the gradient in agricultural reliance on groundwater within the aquifer, the groundwaterscapes also capture the aquatic and terrestrial GDEs in the northeastern reaches of the aquifer through assignment to GS9 (moderate GDEs with small storage capacity). We similarly illustrate how this overlaying of system functions can visually confirm and clarify groundwaterscape maps for the Guarani Aquifer System, Northwestern Sahara Aquifer System, Ganges-Brahmaputra Basin, and North China Aquifer System (see in-figure annotations in Figure 4.6). Below, we further explore and discuss the fidelity of the groundwaterscape maps by briefly placing global results in relation to recent regional descriptions of the greater Sahel region (Rohde et al., 2024) and California's Central Valley (USA) (Huggins et al., 2023a).

The Greater Sahel region is characterized by a challenging intersection of food and water insecurity, cultural diversity, social instability, and weak governance. In these drylands, groundwater crucially sustains terrestrial GDEs that support biodiversity and offer refuge for pastoralists during drought (Rohde et al., 2024). In our mapping, the region is predominantly characterized by groundwaterscape group II (GS4 and GS5: underserved populations and ineffective national governance). In both of these groundwaterscapes, moderate terrestrial GDE densities are represented, a result that is consistent with the prevalence of these ecosystems in the Sahel relative to more extensive GDE landscapes in humid climates, such as across the Amazon and Indonesia. The groundwaterscapes thus broadly reflect the underutilized role of groundwater in the Sahel to support rural water and food security and the challenges of accomplishing development goals in a region with fragile institutions. Yet, local-scale dynamics such as the linkages between GDEs and pastoral livelihoods are invisible to our global groundwaterscapes.

California's Central Valley is one of the most productive agricultural regions in the USA and worldwide. Here, industrial agriculture is highly reliant on groundwater for irrigation while thousands of domestic wells underpin rural water security (Pauloo et al., 2020). Through the Sustainable Groundwater Management Act, there is a strong governance framework for groundwater in the state, which established local Groundwater Sustainability Agencies (GSAs) responsible for developing and implementing Groundwater Sustainability Plans (GSPs). Yet, through insufficient and uncoordinated stakeholder integration, the majority of GSPs currently fail to protect the majority of their agricultural wells, domestic wells, and ecosystems (Perrone et al., 2023).

In our mapping, the majority of the Central Valley is classified as Groundwaterscape 11 (industrial, large farms highly dependent on groundwater with effective national governance). Here, like in the Sahel example, groundwaterscapes broadly capture the dominant characteristics of these groundwater systems when evaluated in a global context. Yet, while the state-wide groundwater governance framework appears to be represented by the effective governance dimension of the groundwaterscape, this is in-fact a coincidental occurrence of state governance happening to correspond with the national governance indicator. Indeed, there exists a wide range of approaches to groundwater governance across U.S. states (for instance, 80 percent of land in neighbouring Arizona has no groundwater regulation; Jacobs, 2009). While comparative reviews on groundwater governance have been conducted for specific regions or policy goals, such as the U.S. southwest (Nelson and Perrone, 2016) and in the context of managing groundwater-dependent ecosystems (Rohde et al., 2017), there may be no good way to summarize and quantify sub-national levels of groundwater governance for use in quantitative global-scale studies.

These two examples point to the potential and limitations of these global groundwaterscapes. We show that the groundwaterscapes offer a capable tool to facilitate first-order differentiation of groundwater systems as social-ecological systems at the regional scale. Yet, simultaneously, we find that the methods we implement and our reliance on quantifiable processes and quantitative data simplify and 'flatten' the place-based complexity of ecohydrological processes, governance, and on the interactions between water and food systems, which may be better expressed and represented at sub-global scales and/or in non-quantitative formats.

Characterizing groundwater systems as groundwaterscapes can facilitate science on the interlinkages between these diverse groundwater functions. While hypothesis testing is beyond the scope of this study, we pose hypothetical lines of inquiry to exemplify this potential. For instance, how might the expansion of irrigated agriculture across the Northwestern Sahara Aquifer System (UNECE, 2020) alter the dominant mode of groundwater-climate interactions and impact ecosystems in these landscapes? Alternatively, how might regional differences in storage capacity within the Guarani Aquifer contribute to different realities regarding climate resilience across groundwater irrigating regions in the north and south of the aquifer? This thinking can also be facilitated at groundwaterscape level. For instance, we find a point of interest in the co-occurrences of landscapes with extensive GDEs in nations with generally ineffective governance (i.e., GS14 and GS15). Might these groundwaterscapes simply be a product of an independent intersection of climate zones and national development trajectories? Else, might effective governance play a role in agricultural industrialization (cf. Thirtle & Piesse, 2007) that may in turn drive land use change and lead to degraded, fragmented, and more spare GDEs? These are the lines of inquiry and types of hypotheses that we envision the groundwaterscapes concept to help facilitate.

Table 4.2. Groundwaterscape descriptions.

Groundwaterscape group	Additional descriptions specific to individual groundwaterscape	Example region
I: Arid and desert regions with minimal functions		
GS1	Large storage capacity.	Sahara
GS2	Small storage capacity and moderately effective national governance.	Arabian Peninsula
GS3	Small storage capacity and ineffective national governance.	Northern Libya
II: Underserved populations and ineffective national governance		
GS4	Some terrestrial GDEs amid generally limited functions.	Great Rift Valley
GS5	Large storage capacity, moderate climate coupling, and some terrestrial GDEs.	Congo Basin
III: Earth system functions in non-agricultural regions		
GS6	Some terrestrial GDEs.	Siberia
GS7	Effective national governance and some terrestrial GDEs.	Northern Australia, Northern Canada.
IV: Moderate GDEs and/or Earth system functions in nations with effective governance		
GS8	Large storage capacity and moderate climate coupling.	Eastern China
GS9	Small storage capacity and very effective national governance.	Scandinavia
V: Agricultural regions with high groundwater dependence		
GS10	Smallholder farming and moderately effective national governance.	Ganges River Basin
GS11	Large farms with effective national governance.	California Central Valley
VI: Agricultural regions with lower dependence on groundwater		
GS12	Moderate climate interactions, few aquatic GDEs, large farms.	Canadian Prairie
GS13	Large farms situated among GDEs.	Southeastern Brazil
VII: Extensive GDEs in non-agricultural regions		
GS14	Small storage capacity, underserved populations, and ineffective national governance.	Eastern Madagascar
GS15	Large storage capacity, large range in underserved populations, and ineffective national governance.	Amazon Basin

Groundwaterscapes on their own cannot answer these questions. Yet, the groundwaterscapes provide a spatial template of comparable units to evaluate particular system behaviours across a variety of system conditions. Given that generalising relationships in complex freshwater systems, such as biodiversity responses to environmental flow transgressions, has proven analytically challenging (Mohan et al., 2022), we suggest that integrating groundwaterscapes and their derivatives in similar investigations can provide an alternative zonal template for analysis of these complex, interlinked systems.

4.4.3 Multiple groundwaterscapes in all large aquifers

All of the 37 large aquifer systems of the world contain multiple groundwaterscapes (Figure 4.7). The Amazon Basin (Brazil) and Canning Basin (Australia) are the least diverse of these large aquifer systems, with only 2 and 1 groundwaterscapes, respectively mapped across more than at least 1% of the aquifer's surface area (6 and 5 groundwaterscapes, respectively, when using a 0.1% area threshold). For the remainder of this section, we will discuss groundwaterscape counts per aquifer in correspondence to the number of groundwaterscapes meeting a 1% area threshold.

In contrast to these relatively homogenous aquifers, the Karoo Basin (South Africa) and Maranhão Basin (Brazil) both contain 9 groundwaterscapes. That 9 of the 15 groundwaterscapes (60%) are found within these aquifer's boundaries highlight their respective region's exceptional groundwater system heterogeneity. That the Maranhão Basin and Amazon Basin exist at opposite ends of this spectrum of groundwaterscape diversity yet are separated by less than 100 km at their nearest points highlights how groundwaterscape compositions can vary considerably over relatively short distances.

We perceive the finding that every large aquifer system is characterized by multiple groundwaterscapes to be a fundamental insight that could have important implications for groundwater science. Treating these systems as homogeneous, lumped units, as is often the case in global groundwater assessments, severely underrepresents the functional heterogeneity that exists within each aquifer. Yet, as aquifer and groundwaterscape mapping are based on vastly different conceptual models, we foresee the potential to use these resources in tandem. It is possible for groundwaterscapes to span aquifers (as aquifers do not consider their overlying social-ecological and Earth system functions) and for aquifers to span groundwaterscapes (as groundwaterscapes do not account for lateral flow or the specific geology of the region and are derived uniquely per grid cell).

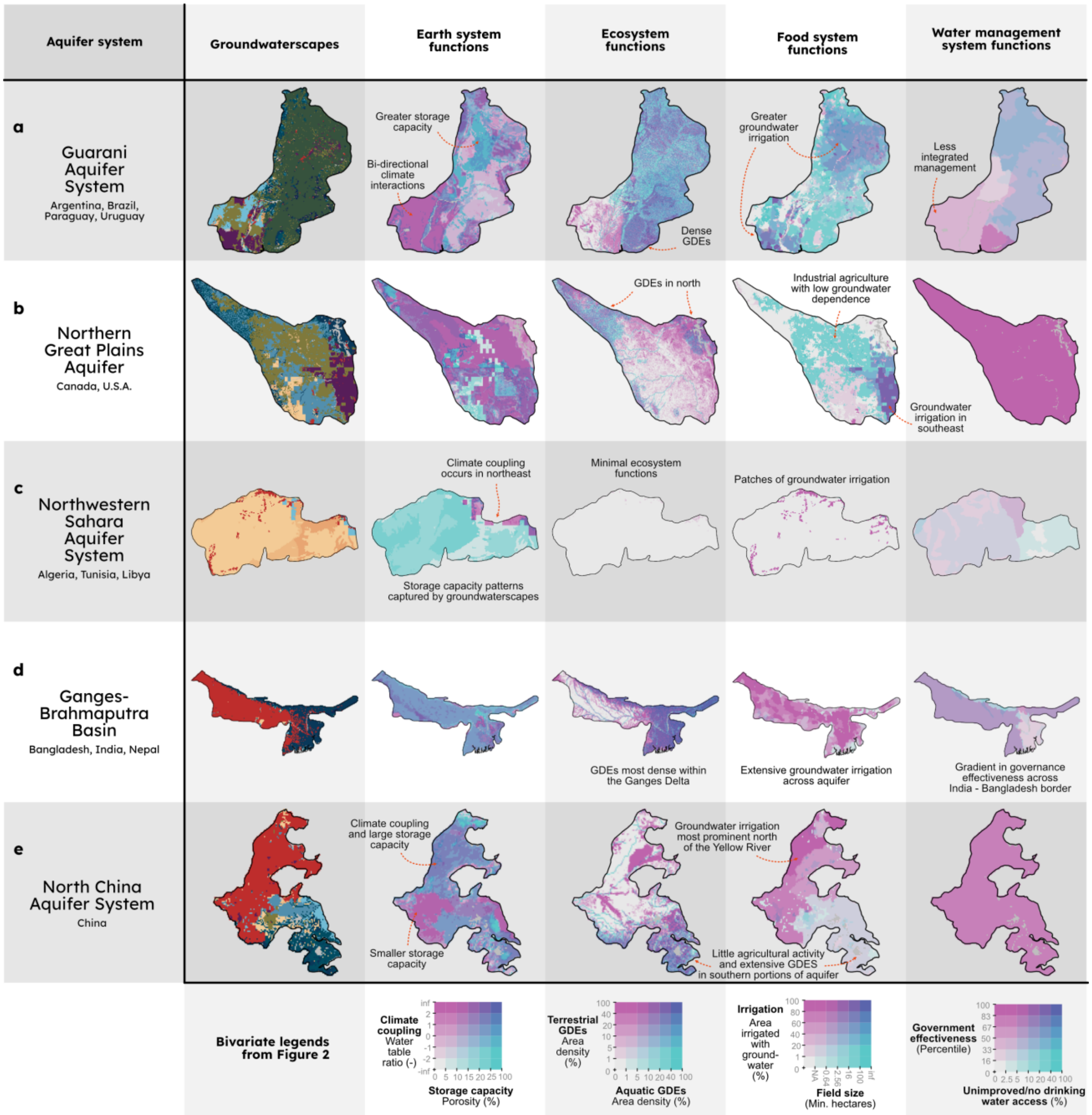


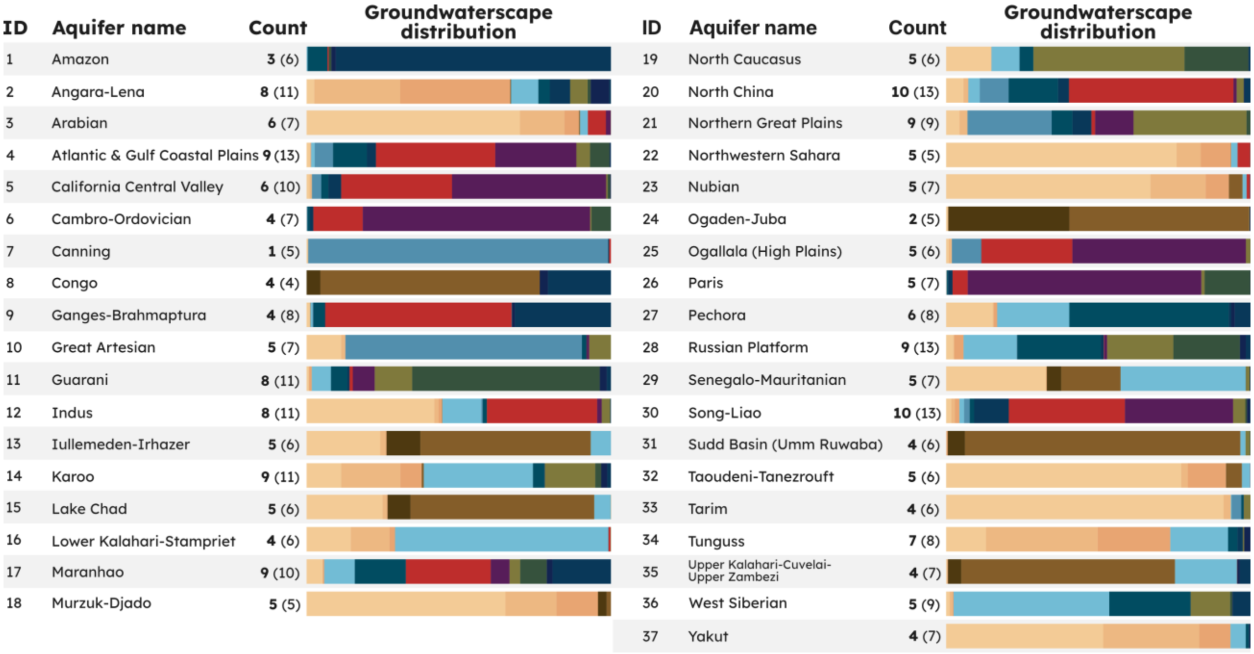
Figure 4.6. The composition of groundwaterscapes.
Columns represent spatial patterns in groundwaterscape distributions, Earth system functions, ecosystem functions, food system functions, and water management system functions for five case study aquifers: (a) the Guarani Aquifer System (Argentina, Brazil, Paraguay, Uruguay), (b) The Northern Great Plains Aquifer (USA, Canada), (c) the Northwestern Sahara Aquifer System (Algeria, Tunisia, Libya), (d) the Ganges-

Brahmaputra Basin (Bangladesh, India, Nepal), and (e) the North China Aquifer System (China). We provide a similar mapping of all 37 large aquifer systems of the world in Figures IV.9-IV.16

For example, understanding groundwater storage trends in the major aquifer systems of the world (e.g., as in Richey et al., 2015) could be strengthened by further specifying storage trends at the groundwaterscape unit within aquifers. It is well established that there are divergent groundwater storage trends within the High Plains (Ogallala) Aquifer, with pronounced depletion in its central and southern regions but groundwater storage gain in its northern regions (McGuire, 2017), yet taking a lumped-system approach moderates groundwater storage trend results across the entire aquifer. In contrast, evaluating the groundwater storage trends within contiguous groundwaterscapes patches could support a more disaggregated specification of storage trends within aquifers while simultaneously facilitating contextualized thinking about the potential socioeconomic, ecological, and Earth system functions at risk due to hydrological change.

Simply counting the number of groundwaterscapes within an aquifer provides an introductory but insufficient description of the groundwaterscape distribution within aquifer systems. For instance, although the Guarani Aquifer System and Karoo Basin (South Africa) contain a similar number of groundwaterscapes within their boundaries (8 and 9, respectively), it can be observed that one groundwaterscape is relatively dominant and covers a considerable area fraction of the Guarani while the 9 groundwaterscapes within the Karoo Basin are more evenly distributed by area (Figure 4.7). Thus, we supplemented this analysis by calculating several additional landscape metrics to further describe the spatial patterns of groundwaterscapes within aquifers (Figure 4.8). While similar analyses could be conducted across other zonal templates (e.g., country borders, water management administrative regions, protected areas, ecological biomes, etc.), we continue our focus on the large aquifer systems as they represent a primary, well-known, and widely used global groundwater system classification.

There is a strong relationship between the Simpson's evenness index and the contagion index of groundwaterscapes within aquifers (Figure 4.8a). These metrics identify aquifers such as the Amazon Basin and Canning Basin as among the least diverse and most contiguous in their groundwaterscape make-up, whereas the Song-Liao Basin (China) and Maranhão Basin are among aquifers with the greatest heterogeneity and diversity of groundwaterscapes. Given landscape indices such as the Simpson's evenness index and the contagion index are often correlated, plotting marginal entropy against relative mutual information is one approach that has been used to differentiate and classify landscape patterns through indices with weaker correlation (Nowosad & Stepinski, 2019). When applying this approach (Figure 4.8b), groundwaterscape patterns between aquifers that contain similar levels of evenness and contiguity can be differentiated. For instance, the Paris Basin (France) and Taoudeni-Tanezrouft Basin (Mali, Mauritania, and Algeria) show similar levels of evenness and contiguity (Figure 4.8a) yet the two basins can be differentiated on the basis of relative mutual information, with the Paris Basin having considerably less relative mutual information (Figure 4.8b). Such analytical approaches could be useful for future applications of the groundwaterscapes that would benefit from grouping aquifers based on similarity in their groundwaterscape composition and complexity.



ID	Aquifer name	Count	ID	Aquifer name	Count
1	Amazon	**3** (6)	19	North Caucasus	**5** (6)
2	Angara-Lena	**8** (11)	20	North China	**10** (13)
3	Arabian	**6** (7)	21	Northern Great Plains	**9** (9)
4	Atlantic & Gulf Coastal Plains	**9** (13)	22	Northwestern Sahara	**5** (5)
5	California Central Valley	**6** (10)	23	Nubian	**5** (7)
6	Cambro-Ordovician	**4** (7)	24	Ogaden-Juba	**2** (5)
7	Canning	**1** (5)	25	Ogallala (High Plains)	**5** (6)
8	Congo	**4** (4)	26	Paris	**5** (7)
9	Ganges-Brahmaptura	**4** (8)	27	Pechora	**6** (8)
10	Great Artesian	**5** (7)	28	Russian Platform	**9** (13)
11	Guarani	**8** (11)	29	Senegalo-Mauritanian	**5** (7)
12	Indus	**8** (11)	30	Song-Liao	**10** (13)
13	Iullemeden-Irhazer	**5** (6)	31	Sudd Basin (Umm Ruwaba)	**4** (6)
14	Karoo	**9** (11)	32	Taoudeni-Tanezrouft	**5** (6)
15	Lake Chad	**5** (6)	33	Tarim	**4** (6)
16	Lower Kalahari-Stampriet	**4** (6)	34	Tunguss	**7** (8)
17	Maranhao	**9** (10)	35	Upper Kalahari-Cuvelai-Upper Zambezi	**4** (7)
18	Murzuk-Djado	**5** (5)	36	West Siberian	**5** (9)
			37	Yakut	**4** (7)

Figure 4.7. Groundwaterscape area distributions in the large aquifer systems of the world. Groundwaterscape counts are calculated based on those that cover a minimum threshold of 1% (and 0.1%) of the aquifer area.

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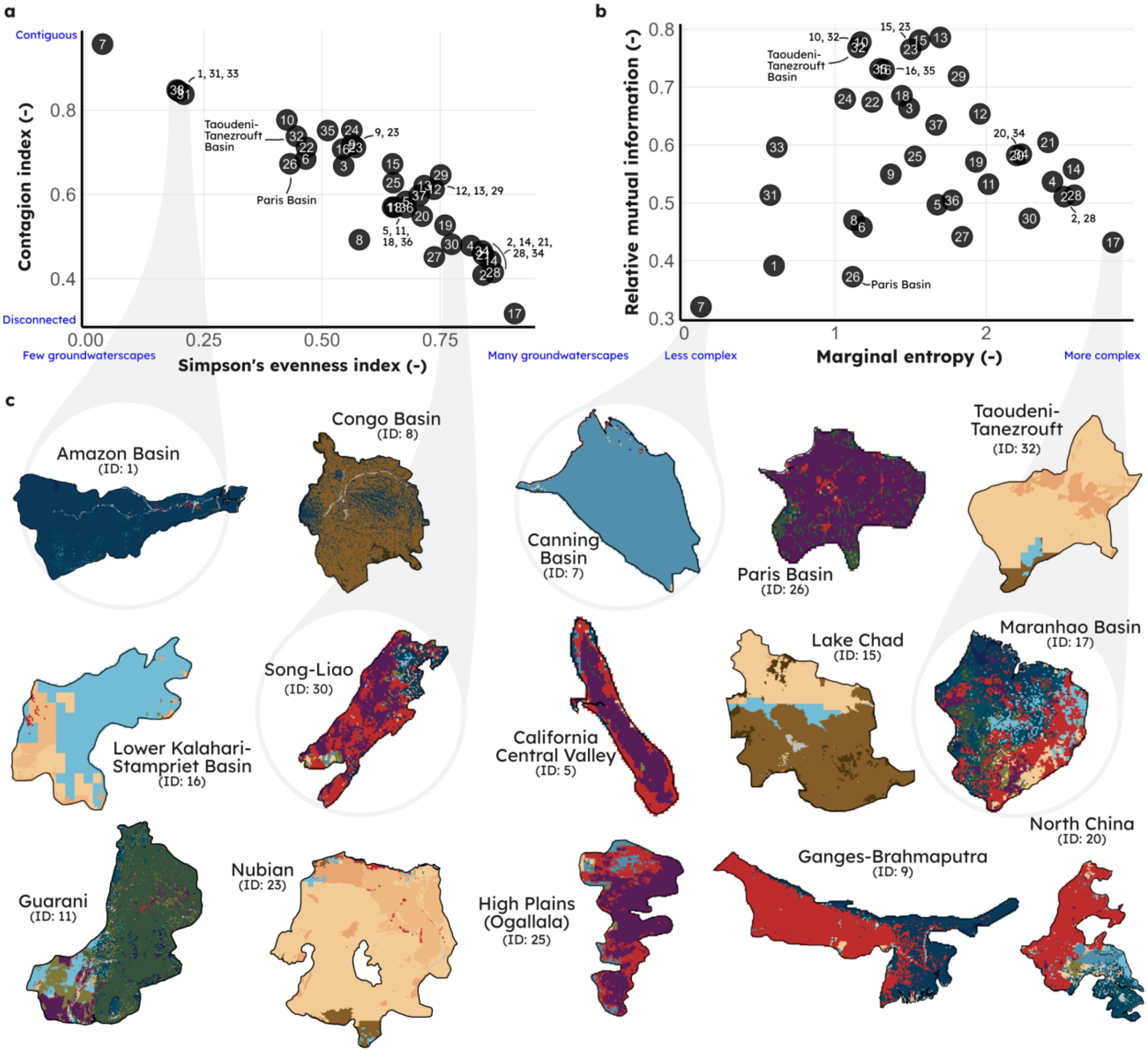

Figure 4.8. Landscape metrics of groundwaterscapes within the large aquifer systems of the world.

(a) Plot of Simpson's evenness index (x-axis) and the contagion index (y-axis). (b) Plot of marginal entropy (x-axis) and relative mutual information (y-axis). (c) Groundwaterscape distributions within highlighted aquifers. Aquifer IDs correspond to the points labels in panels (a) and (b) and also correspond to the aquifer borders mapped in Figure 4.5. Inset maps are sized for visualisation and are not shown at a consistent scale.

4.4.4 Groundwaterscapes are not equally monitored

These groundwaterscapes offer an alternative conceptual model to understand, study, and manage global groundwater systems. To juxtapose this study with the influential concept of hydrologic landscapes, which hypothesize that hydrological systems behave as a function of land-surface form, geology, and climatic setting (Winter, 2001), we present groundwaterscapes as systems whose behaviour is a function of interacting Earth systems, ecosystem, agricultural system, and water management system processes. On this basis, groundwaterscapes are different and distinct systems to measure and study in comparison to physical groundwater systems.

We find a striking imbalance in the global groundwater monitoring well network distribution across groundwaterscapes (Figure 4.9a). Groundwaterscapes GS11-GS13 (characterized by industrial agriculture) benefit from >60% of all monitoring wells despite covering a combined 13% of the land surface (Figure 4.9b). Conversely, some groundwaterscapes, such as GS4-GS6, have almost very little representation in the observation network. These groundwaterscapes cumulatively contain less than 1% of all monitoring wells yet cover over 18% of the land surface. These monitoring disparities widen when normalizing by surface area (Figure 4.9c). While GS12 contains the most monitoring wells, GS11 has a higher monitoring well density. As economic factors and governance capacity influence the ability of jurisdictions to monitor their groundwater resources, it is not surprising that groundwaterscapes characterizing industrial agriculture dominate the monitoring network distribution. Yet, even within agricultural regions we see imbalances in monitoring. For instance, GS10 (groundwater-reliant smallholder agriculture) has about one-third of the monitoring well density of GS11.

The biases we observe in the well network may not be entirely independent from our derivation method as we may expect that groundwaterscapes characterized with more effective national governance would benefit from more observation wells. However, it remains that effective groundwater management depends on representative data (Curran et al., 2023), and therefore the biases and blind spots in global groundwater data collection undermine the ability to manage groundwaterscapes on a data-driven basis. In this way, the groundwaterscape concept can be a used as a tool to identify data collection priorities, and moreover as a deliberation device to re-imagine what effective groundwater data collection entails in order to assemble more representative and capable sets of observations to understand change in groundwaterscapes.

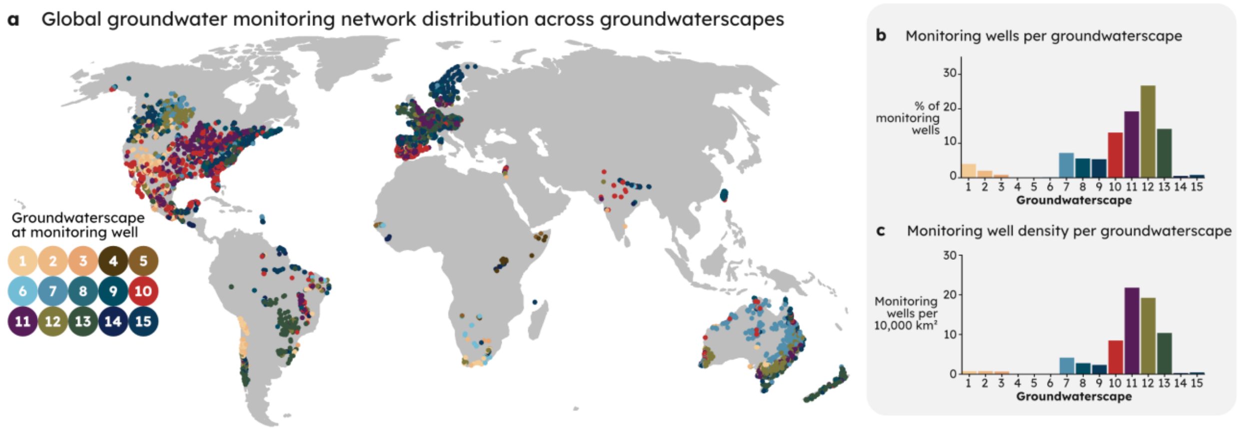

Figure 4.9. Distribution of the Global Groundwater Monitoring well network (GGMN) (IGRAC, 2024) across groundwaterscapes.

(a) Map of GGMN wells coloured according to their groundwaterscape. (b) Proportion of GGMN wells found within each groundwaterscape. (c) GGMN well density per groundwaterscape.

4.4.5 Groundwaterscapes as a starting point

We present these groundwaterscapes as a plausible classification of global groundwater systems built on a function-oriented understanding of groundwater systems as social-ecological systems. Yet moreover, these groundwaterscapes represent a global mapping of the alternative conceptual model presented by the groundwater-connected systems framing (Huggins et al., 2023a) and thus support an overarching ambition to characterize, understand, and manage groundwater systems on the basis of the resource's role in social-ecological systems. Our perception is that debate on effective ways to proceed in this regard is far from settled and we expand on this reflection in a number of ways below.

In a practical sense, the groundwaterscapes are challenging to validate. This is not unique to this study and rather is a general problem in archetype analysis (Piemontese et al., 2022). This stems from the fact that social-ecological system typologies are conceptual constructs rather than physical entities (Oberlack et al., 2019) and thus cannot be directly measured. In the archetype analysis literature, a comprehensive validation procedure is proposed to consist of six dimensions (Piemontese et al., 2022) that span qualitative evaluations on the strength of conceptual framing, data fidelity, methodological robustness, the explicitness of study scope, empirical justification, and an evaluation of the potential application. As this study does not conduct a 'full' archetype analysis and rather presents the groundwaterscapes as possible archetype 'candidates' for evaluation and future refinement, we do not foresee the need for the full set of proposed validation components to be incorporated here.

We perceive our study to follow ‘strong’ validation guidelines by using a theory-grounded conceptual model to underpin our study, sourcing global datasets that correspond closely with our conceptual model, and in implementing a robust and reproducible derivation method. We bound our study by acknowledging that the groundwaterscapes only represent the groundwater functions included in our conceptual model, and thus omit important functions that occur in coastal environments, small islands, permafrost regions, and urban settings. We additionally do not consider non-agricultural economic uses of groundwater such as mining, manufacturing, and energy generation, nor do we consider groundwater quality or geochemical functions. We foresee the potential for adapted groundwaterscapes to address these conceptual limitations and readily welcome the pluralisation of the groundwaterscape concept.

There are important data limitations that provide further basis to view the groundwaterscapes through a critical lens. While we used the best-available, analysis ready, and open-access data to represent each function in our conceptual model, several datasets would benefit from further refinement. We used data layers for their most-recent year available, but some layers are now considerably dated such as groundwater irrigation areas which correspond to the year 2005. Additional challenges to individual datasets include a simple, inference-based approach used to map groundwater-dependent ecosystems, the lack of a specific groundwater governance dataset, and a reliance on a drinking water services dataset that does not separate groundwater from other sources. Yet, we view these data limitations as opportunities for future groundwaterscape improvement. We note that our reproducible methods and script repository enable the update of our groundwaterscape map following the release of new datasets.

We perceive this groundwaterscape mapping study as a potential catalyst for wider application of social-ecological system concepts within the global-scale groundwater domain. For instance, global hydrological models, which are arcing towards visions of “physically-based continental Earth system models” (Bierkens, 2015), could benefit from parameterization and conceptual model development facilitated through groundwaterscapes. The groundwaterscape concept can also be applied to support data collection strategies and as a spatial template to identify diverse case study locations for modelling or field work studies.

Groundwaterscapes can more generally be used to test hypotheses on groundwater-connected system behaviour. Thus, groundwaterscapes can support the application and development of middle range theories of change to groundwater science, which represent "contextual generalisations that describe chains of causal mechanisms explaining a well-bounded range of phenomena, as well as the conditions that trigger, enable, or prevent these causal chains" (Meyfroidt et al., 2018). Thus, an overarching potential of the groundwaterscape concept is to serve as a conceptual and analytical tool to facilitate investigations on causal processes connecting these complex and intertwined hydrological, social, ecological, and Earth systems.

4.5 Conclusion

We developed the concept of groundwaterscapes which are landscape units with common configurations of groundwater system functions. We classified and mapped groundwaterscapes globally based on eight large-scale groundwater functions across Earth systems, ecosystems, food systems, and water management systems using a two-stage, deeply iterative self-organizing map method. The 15 groundwaterscapes characterize landscapes such as arid and desert regions with minimal functions, underserved populations with ineffective governance, Earth system functions in non-agricultural regions, moderate GDEs or Earth system functions with effective governance, agricultural regions with both high and low dependence on groundwater, and non-agricultural regions with extensive GDEs. All large aquifer systems of the world contain multiple groundwaterscapes, highlighting the functional heterogeneity that is overlooked when these systems are treated as homogenous units in global assessments. We found a striking imbalance in global monitoring wells across groundwaterscapes with only three groundwaterscapes benefiting from 60% of all monitoring wells while other groundwaterscapes contain next to no monitoring capacity. The groundwaterscapes can serve as a conceptual and spatial tool for the large-scale groundwater research community to engage more fully with the complex realities of groundwater system dynamics in social-ecological systems. Important steps are being taken in this direction by multiple research groups, mainly oriented around developing understanding of pairwise system interactions with groundwater (e.g., groundwater-climate processes, groundwater-streamflow processes, groundwater-terrestrial ecosystem processes). This study is our attempt to begin the process of bringing together these research streams and make initial progress towards developing a more holistic, system-of-systems understanding of groundwater at the global scale.

4.6 Code availability

Environment	R project for statistical computing (R Core Team, 2023)
R packages	
terra	Hijmans et al. (2024) https://cran.r-project.org/package=terra
kohonen	Wehrens & Kruisselbrink (2018) https://cran.r-project.org/package=kohonen
aweSOM	Boelaert et al. (2022) https://cran.r-project.org/package=aweSOM
clusterSIM	Walesiak & Dudek (2020) https://cran.r-project.org/package=clusterSIM
landscapemetrics	Hesselbarth et al. (2019) https://cran.r-project.org/package=landscapemetrics
tmap	Tennekes et al. (2018) https://cran.r-project.org/package=tmap
ggplot2	Wickham (2016) https://cran.r-project.org/package=ggplot2
MetBrewer	Mills (2022) https://cran.r-project.org/package=MetBrewer
Script repository	https://github.com/XanderHuggins/groundwaterscapes
Data repository	Borealis (https://borealisdata.ca/), pending manuscript acceptance.

4.7 Acknowledgements

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Chapter 5

GROUNDWATERSCAPE RISKS COMPREHENSIVELY MAP GLOBAL GROUNDWATER SUSTAINABILITY CHALLENGES

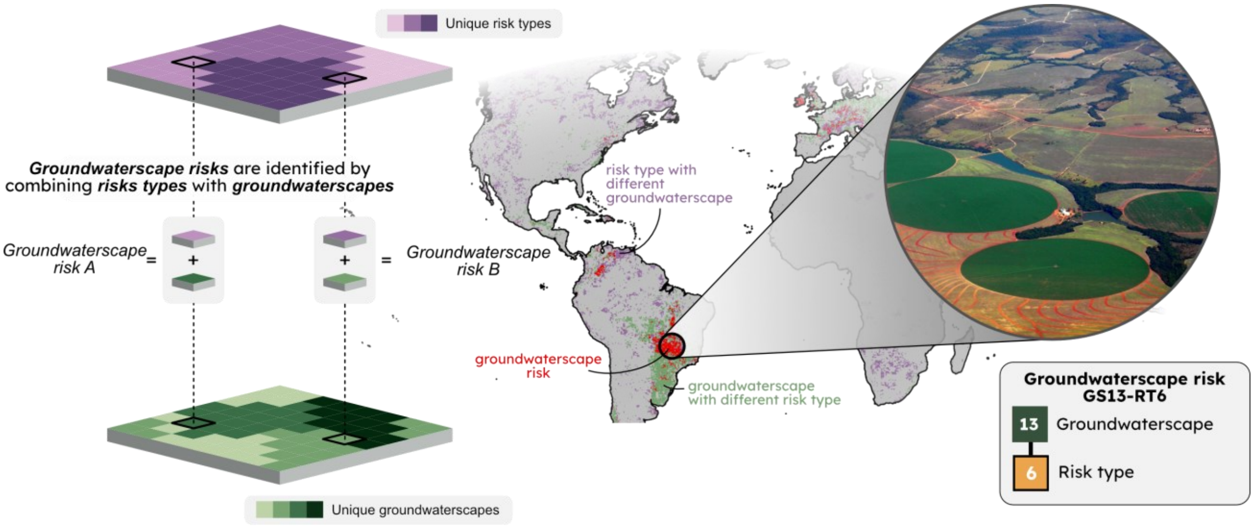
This chapter will be formatted for submission.

Huggins, X., Gleeson, T., Moore, M.-L., & Famiglietti, J. S. (2024). Groundwaterscape risks comprehensively map global groundwater sustainability challenges. Manuscript.

Key points:

- Uses an Anthropocene risk framing to identify groundwater risks across themes of Earth system change and inequality emergence.
- Identifies a set of 270 groundwaterscape risks, representing specific and unique configurations of groundwater system functions and groundwater system risks.
- Groundwaterscape risks present the most comprehensive synthesis of global groundwater challenges to date and offer a conceptual and spatial tool for solution development and transfer.

Figure 5.1. Graphical abstract.



5.1 Abstract

Understanding where groundwater system functions and risks co-occur is an important first step in developing context-appropriate groundwater sustainability strategies, yet this perspective and supporting analysis is missing in the current global groundwater crisis discourse. To address this need, we present a spatially explicit and comprehensive characterization of global groundwater sustainability challenges given data availability. Our approach includes but extends beyond physical risks such as groundwater depletion and embeds a wider set of biophysical and socioeconomic system risks. We develop a global mapping of groundwaterscape risks which identify specific and broadly occurring configurations of groundwater system functions (such as storage capacity and groundwater-dependent ecosystems) and groundwater system risks (such as groundwater storage loss, land use change, land subsidence, and gender inequality). In total, 270 groundwaterscape risks characterize the global groundwater sustainability problem space. Just 38 groundwaterscape risks cover half of the land surface and only 19 groundwaterscape risks encompass half of the global population. We present these groundwaterscape risks as a concept that can embed regionalized social-ecological context and narratives into the global groundwater sustainability discourse and can act as a tool to facilitate solution transfer on the basis of groundwaterscape risk similarity.

5.2 Introduction

Groundwater sustainability is a place-based concept and process, mirroring the richness of the physical geographies, ecologies, and societies that groundwater is situated within and connected to. Yet, sustainability-oriented groundwater assessments at the global scale are dominated by a prevailing interest in quantitative, physical assessments such as on groundwater storage and water table trends (Bierkens & Wada, 2019; Jasechko et al., 2024; Richey et al., 2015; Thomas et al., 2017; Wada & Heinrich, 2013). These important efforts continue to provide fundamental insights to understand changing hydrogeological conditions (Kuang et al., 2024), and crucially have drawn attention to the global groundwater crisis (Famiglietti, 2014). Yet, approaches that focus exclusively on physical groundwater systems mask over the complex realities facing groundwater that stem from the resource's embeddedness within social-ecological systems (Huggins et al., 2023a) that materialize through ecological, agricultural, sociocultural, and economic system interactions with groundwater resources. We contend that approaches to assess global groundwater challenges which center the resource's social-ecological setting are necessary to more extensively represent the problem space presented in groundwater sustainability (Gleeson et al., 2020) and to better align global groundwater science with the place-based nature of groundwater systems (Foster et al., 2013; Zwarteveen et al., 2021).

We build on and contribute to a handful of global studies that have investigated social-ecological dimensions of groundwater sustainability, including groundwater consumption in relation to environmental flow needs (Gleeson et al., 2012; de Graaf et al., 2019), groundwater depletion and human activity in relation to groundwater-dependent ecosystems (Huggins et al., 2023b; Link et al., 2023; Rohde et al., 2024), water table depths relative to well depths (Jasechko & Perrone, 2021), and groundwater depletion embedded in international food trade (Dalin et al., 2017). Yet, no work to date has taken an explicit social-ecological system-wide approach to evaluate the breadth of sustainability challenges facing groundwater across the resource's various interactions with social, ecological, and Earth systems. Taking a groundwater-connected systems approach (Huggins et al., 2023), where groundwater systems are conceptualized not on the basis of underlying hydrogeological properties but through system functions such as storage capacity, supporting groundwater-dependent ecosystems, and groundwater irrigation, can be useful for this purpose.

Here, we make initial progress in this direction by producing a global map of groundwaterscape risks (see Box 5.1 for key terminology). These risks represent specific and broadly occurring spatial patterns across multiple groundwater system functions and groundwater system risks. We build directly on the recent global mapping of groundwaterscapes (Huggins et al., 2024; Chapter 4), which characterize global patterns in groundwater's large-scale Earth system, ecosystem, food system, and water management system functions. We complement the groundwaterscapes with an analogous mapping of global groundwater system risk types, which characterize global patterns in risks such as land use change, hydro-political tension, gender development inequalities, and groundwater storage loss (further described below).

Groundwaterscape risks are mapped by overlaying groundwaterscapes with groundwater risk types (Figure 5.2). Each groundwaterscape risk is defined by a unique pairing of groundwaterscapes and groundwater risk types. Linking groundwater system functions and risks in this way is crucial to begin understanding the functional impacts, interactions, and complexity of the global groundwater crisis. We argue that advancing this perspective can serve both scientific needs to better understand and model the dynamics that connect groundwater with social-ecological systems, and sustainability needs to better manage groundwater on context-specific bases informed by groundwater system functions and social-ecological settings.

These classifications are developed by performing spatially explicit social-ecological system cluster analysis (Sietz et al., 2019). Social-ecological cluster analysis is an approach to identify predominant patterns in attributes and behaviours of social-ecological systems and has been applied from sub-national (Rocha et al., 2020) to regional (Levers et al., 2018; Sietz et al., 2017) to global (Jung et al., 2024; Václavík et al., 2013) scales for various systems. This classification procedure was performed using a self-organizing map-based derivation methodology that provides a robust, reproducible, multi-layered, and narrative-oriented classification of predominant patterns across multi-dimensional, spatially explicit datasets (see Methods, Section 5.3).

Our conceptual approach to identify groundwater risks is described in “Conceptualizing groundwater system risks” (Section 5.3.1). We interpret the resulting risk types in “Groundwater risk types synthesize Earth system change and potential for inequality emergence in groundwater-connected systems” (Section 5.4.1). We do not focus on describing groundwaterscapes in equal depth as they are described in Huggins et al. (2024) (see Box 5.2). Groundwaterscape risks are presented in “Groundwaterscape risks characterize the groundwater sustainability problem space” (Section 5.4.2). Case study groundwaterscape risks are discussed in “Groundwaterscape risks: a guiding tool for solution development and transfer” (Section 5.5.1).

All analyses in this study are conducted at 5 arcminute resolution (~10 km grids at the equator), providing a medium-resolution global dataset for use by the wider groundwater science and sustainability research community (see Data availability, Section 5.6). All data used in this study come from previously published studies and data depositions, ensuring we can focus our attention on the integration of these data to derive groundwaterscape risks.

Box 5.1. Key terminology.

Groundwater-connected systems: A conceptual framing to view groundwater systems as social-ecological systems by focusing on groundwater interactions with socioeconomic, ecological, and Earth systems (Huggins et al., 2023; Chapter 3). ***Groundwaterscapes***, which are based on the groundwater-connected system framing (Huggins et al., 2024; Chapter 4), are landscape units that represent specific and broadly occurring configurations of groundwater functions across Earth systems, ecosystems, food systems, and water management systems.

Anthropocene risks: A conceptual risk framing to represent the complex and emergent risks in social-ecological systems that characterize the Anthropocene (Keys et al., 2019). Anthropocene risk relates to traditional risk components of hazard, exposure, and vulnerability, but focuses on anthropogenic changes to Earth system functions, social-ecological system behaviors such as inequality and injustice emergence, and cross-scale interactions that occur over potentially long time horizons and can involve Earth system tipping elements. We apply the Anthropocene risks framing to determine the set of groundwater system risks we include in our derivation of ***groundwater risk types***.

Social-ecological system clustering: Predominant configurations of social-ecological system attributes, identified by combining cluster analysis with a social-ecological system conceptual model (Rocha et al., 2020; Sietz et al., 2019; Václavík et al., 2013). Social-ecological system clustering is a common element of archetype analysis in sustainability science (Eisenack et al., 2021) which supports the identification of plausible hypotheses on configurations of causal factors that lead to distinct social-ecological system behaviours. Social-ecological system clusters are not physical entities but represent conceptual constructs aimed at addressing challenges of overgeneralization by identifying system types at intermediate levels of abstraction that bridge global and local realities (Oberlack et al. 2019). We apply social-ecological system clustering to transform our conceptual models of groundwater system functions and groundwater system risks into groundwaterscape and groundwater risk type maps, respectively.

Groundwaterscape risks: Combinations of groundwaterscapes and groundwater risk types (Figure 5.2). Thus, each groundwaterscape risk has a unique configuration of groundwater system functions and groundwater system risks, which together present specific challenges for groundwater sustainability. Groundwaterscape risks are a data-driven, narrative-oriented tool to facilitate consideration of social-ecological context in global groundwater sustainability efforts across science and management.

Summary of global classification maps included in this study:

Groundwaterscapes (GS): classification map of system functions	Groundwater ***risk types*** (RT): classification map of system risks	***Groundwaterscape risks***: classification map of both system functions and risks

5.3 Methods

Note: section order has been rearranged from the manuscript version of this chapter so that the methods section is located before the results.

5.3.1 Conceptualizing groundwater system risks

We draw on an Anthropocene risk framing (see Box 5.1) to ensure our conceptualization of groundwater system risks corresponds with leading theory on risk in social-ecological systems. Taking this approach resulted in the inclusion of often omitted, underrepresented, and emergent risks such as hydropolitical tension, gender development inequality, and ecological conservation needs in addition to commonly centered and highlighted groundwater risks such as groundwater storage trends, land subsidence, agricultural intensification, and land use change (e.g., urbanization and cropland expansion).

In total, our conceptual model of groundwater risks consists of nine risks (Figure 5.2) that span the hydrosphere, lithosphere, atmosphere, biosphere, and anthroposphere. Our inclusion of land subsidence, groundwater storage trends, change in heavy rainfall frequency, land use change, and agricultural intensification all correspond to human-mediated changes in the Earth system. Alternatively, our inclusion of ecological conservation needs, hydro-political tension, gender development inequality, and water crowding relate to potential inequality emergence in social-ecological systems connected to groundwater. Cross-scale dynamics are core properties of these groundwater-connected systems (Huggins et al., 2023a), and thus our considered set of groundwater risks directly correspond to all core components of Anthropocene risk (Keys et al., 2019).

These groundwater risks can be intertwined with one another. For instance, agricultural intensification and land use change can reduce groundwater recharge and lead to groundwater depletion (Scanlon et al., 2005; Wada et al., 2012), land use change can undermine ecological conservation priorities (Jones et al., 2018), water crowding can lead to increasing competition and potential for hydro-political tensions or conflict (Almer et al., 2017), and groundwater depletion can exacerbate gender inequalities in regions where responsibilities for collecting drinking water disproportionately fall on women (Swaraj & Maheshwari, 2022). Understanding such interactions and co-occurrences between groundwater risks is crucial to tailor effective groundwater management strategies and leverage groundwater sustainability as a tool to support wider social, economic, and ecological goals.

The groundwater risks we identify have implications beyond physical groundwater systems and can impact broader groundwater system functions. For instance, land use change (risk) can threaten the integrity or presence of groundwater-dependent ecosystems (function) (Kløve et al., 2011), groundwater storage loss (risk) can alter the directional mode of climate-groundwater interactions (function) (Cuthbert et al., 2019), and land subsidence (risk) reduces aquifer storage capacity (function) (Konikow & Kendy, 2005). Interactions between risks and functions occur in the opposite direction as well. For instance, ineffective governance (function) can create conditions that foster hydropolitical tensions (risk) (Gorelick & Zheng, 2015), and groundwater irrigation (function) is the dominant global driver of groundwater depletion (risk) (Wada et al., 2012). This understanding of dynamic and complex function-risk relationships is what underpins our approach to derive groundwaterscape risks through the overlaying of groundwaterscapes with groundwater risk types.

Our set of risks consist of both snapshot (i.e., single timestamp data such as gender development inequality for the year 2020), and trend data (such relative change in water crowding projected between the year 2020 and 2050). Our use of either snapshot or trend data for each risk depends on whether the principal risk is found in the magnitude of the risk itself (snapshot) or in a change in the magnitude (trend). We further reflect on the composition of our groundwater system risks in the Section 5.5. In Table 5.1, we describe and justify the inclusion of each risk. For static (snapshot) data, we use data as closely aligned to the year 2020 as possible. For trend data, time ranges are best selected under data availability constraints. For a full list of data sources and preprocessing steps, see Table V.1.

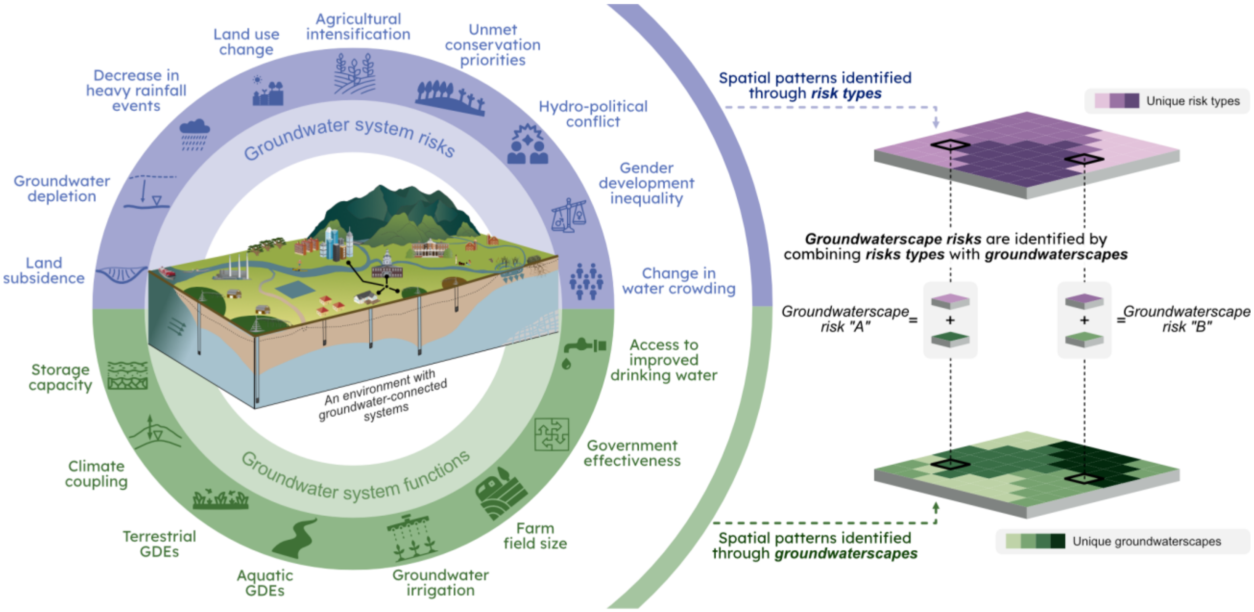


Figure 5.2. Groundwaterscape risk conceptual model, based on both groundwater system functions (bottom) and groundwater system risks (top).

Spatial patterns in groundwater system functions are identified through *groundwaterscapes*, and spatial patterns in groundwater system risks are identified through *groundwater risk types*. Groundwaterscape risks are derived by overlaying groundwaterscapes with risk types.

Table 5.1. Groundwater Anthropocene risks and rationale for inclusion.

Anthropocene risk	
Land subsidence	**Rationale:** Can cause irreversible loss of aquifer storage capacity. **Indicator:** Global potential subsidence. **Source:** Herrera-García et al. (2021).
Groundwater storage trends	**Rationale:** Change in groundwater availability, which can be caused by human use (pumping), land use change, climate change, and natural variability; and experienced through lower water tables, detrimental groundwater-dependent ecosystem states, and altered groundwater-climate interactions. **Indicator:** Trends in groundwater storage over 2003-2022. **Source:** Li et al. (2019).
Change in extreme rainfall	**Rationale:** Importance of episodic groundwater recharge during extreme rainfall events (Taylor et al. 2013; Owor, et al. 2009). **Indicator:** Relative change in the annual frequency of 10mm d^{-1} precipitation events between 1980-1989 and 2010-2019. **Source:** MSWEP; Beck et al. (2019)

Land use change	**Rationale:** Land use change is a direct driver of change in groundwater recharge (Scanlon et al. 2005), and modern groundwater, the groundwater most vulnerable to global change, is evaluated as groundwater that has recharged within 50 years (Gleeson et al. 2016). **Indicator:** Relative change in urban, cropland, and pasture classes of land use from 1960-1969 to 2010-2019. **Source:** Winkler et al. (2021)
Agricultural intensification	**Rationale:** Agriculture represents the largest sectoral user of groundwater, and agricultural intensification to close yield gaps poses a parallel threat to cropland expansion to drive increased agricultural water use. **Indicator:** Yield gaps averaged across 10 major crops for the year 2010 **Source:** Gerber et al. (2024)
Unmet conservation needs	**Rationale:** Groundwater supports groundwater-dependent ecosystems around the world (Saccò et al., 2024), and groundwater management is crucial to ensure the function of these ecosystems and their services (Saito et al., 2021). **Indicator:** Prioritization index for conserving terrestrial biodiversity, carbon, and water considering current extents of protected areas, circa 2020. **Source:** Jung et al. (2021)
Hydropolitical conflict	**Rationale:** Groundwater depletion is linked with social conflicts, from tensions within communities to acting as a contributing trigger for conflict, underscoring the need to manage groundwater as a tool for peace and cooperation. **Indicator:** Indicator of hydropolitical risk over 1997-2012 **Source:** Farinosi et al. (2018)
Gender development inequalities	**Rationale:** Improved water access can act as a tool to reduce gender inequalities in regions where women spend disproportionately more time collecting water than men (United Nations, 2016), and has cascading impacts on education and women empowerment (Kookana et al., 2016). **Indicator:** Subnational Gender Development Index for the year 2020 **Source:** Global Data Lab
Change in water crowding	**Rationale:** Population growth corresponds with greater domestic and agricultural demand for freshwater resources and creates opportunities for further inequalities following greater competition for freshwater access. **Indicator:** Relative population change from 2020 to 2050 under SSP3 **Source:** Wang et al. (2022)

5.3.2 Data preprocessing

All data used in this study are sourced from previously published datasets. We summarize these input datasets in Table V.1. This study is performed at 5 arcminute resolution (~10 km grids near the equator), and all input data were harmonized to this resolution using methods as outlined in Table V.1.

5.3.3 Classification of groundwater system risk types

Our approach to derive groundwater system risk types follows the sequential self-organizing map (SOM)-based classification procedure used to derive groundwaterscapes (Huggins et al., 2024; Chapter 4). Thus, as we follow the same procedure, we provide a more parsimonious description of the methods here and defer to Huggins et al. (2024) for a comprehensive account. In short, this method uses SOMs at two stages to derive clusters within a multidimensional dataset. SOMs are a form of unsupervised machine learning that project a multi-dimensional input feature space onto a low-dimensional grid of nodes, with each node containing a 'codebook' vector in the full dimensionality of the input data. Each codebook vector represents the modal value of the input data represented by that node. In this two-stage methodology, the first-stage, two-dimensional (i.e., square) SOM reduces the overall size of input feature space while preserving the general topology of the input feature space, while the second-stage, one-dimensional SOM uses the codebook vectors of the first-stage SOM as the input feature space to identify clusters. This nested structure allows for greater traceability of input feature classification in comparison to single-step methods, which we view as advantageous in global applications where 'unpacking' cluster results by using intermediary levels of clustering may be useful for subsequent investigations. This methodology is deeply iterative, and SOMs at both stages are tested across a wide range of grid sizes to enable the best-performing SOM size to emerge from the iterations rather than requiring preceding specification. A visualization of the complete groundwater system risk type derivation methodology is shown in Figure V.4.

After all input data are harmonized to 5 arcminute resolution, we normalize each raster layer by converting to Z-scores by subtracting by the mean layer value and dividing by the layer's standard deviation. To limit extreme outlier cells from exerting an overpowering influence on our classification, we set upper and lower limits of +2 and -2 for each layer. Normalized raster layers are shown in Figure V.1.

To estimate the number of patterns embedded in the initial input feature space, we perform k-means clustering at increasing k until the between cluster sum of squares exceeded 98% of the total sum of squares. We found that k=6,000 reached this threshold, which we then used to set the grid size ranges for the first-stage SOM models (Delgado et al., 2017). We then developed 60 first-stage SOM models per grid size, ranging from S_{min} = 8 to S_{max}= 30, increasing in increments of 2 (i.e., 8x8, 10x10, …, 30x30). We then evaluated the performance of each model using the Kaski-Lagus error function (Kaski & Lagus, 1996) and the Davies-Bouldin index (Davies & Bouldin, 1979). To reduce the influence of the stochastic property of SOMs and improve the reproducibility of the study, we removed outlier models whose performance for individual or combined metrics were outside a median absolute deviation-based range around each grid size's median performance. This process yielded a best-performing first-stage SOM at the grid size of 30x30 (Figure V.2). With this optimal grid size identified, we then developed a set of 60 alternative SOMs at S = 30 using the full set of input data and similarly screened for performance outliers before selecting the best-performing model. The codebook vectors from this model yield a set of 900 codebook vectors that are subsequently used as the input feature space for the second-stage SOM.

The second-stage, one-dimensional SOM models were developed across a range of grid sizes from 2 to 30, increasing in increments of 1 (i.e., 1x2, 1x3, …, 1x30). For these second-stage SOMs, 120 alternative models were developed per grid size. We evaluated these second-stage models using the same performance metrics as in the first-stage SOM, in addition to the percentage of explained variance and a cluster size preference function aimed at selecting fewer clusters, but no fewer than an a priori estimate of the number of clusters in the dataset, should all other performance metrics be relatively equal (see Huggins et al., 2024; Chapter 4). To select the best-performing second-stage SOM model and grid size, we perform a sensitivity analysis adjusting both the stringency of the performance outlier filtering process and the weight allocated to the cluster count preference function. The resulting matrix (step 6 in Figure V.2) provides a landscape of optimal clustering results depending on preferences of SOM reproducibility and cluster count. From this landscape of alternative clustering outcomes, we selected the modal result (i.e., the SOM size suggested across the most combinations in our sensitivity analysis) of 18 clusters. As individual grid cells are quantized to nodes in the first-stage SOM, and codebook vectors of this first-stage SOM are quantized to nodes in the second-stage SOM, the resulting clusters of this second-stage SOM can be converted back to geographic space for plotting. As performed in other spatial SOM-based clustering studies to reduce minor speckling (noise) in clustering outputs (Jung et al., 2024), we apply a 3x3 modal filter to the final classification map.

5.3.4 Groundwaterscape risks

Groundwaterscape risks are identified by combining groundwaterscapes with the groundwater systems risk types generated above. To generate a unique coding of groundwaterscape risks, we developed a 4-digit system of XXYY, where values XX correspond to the specific groundwaterscape (with 15 classes, from 1 to 15), and where values for YY correspond to the risk type (also with 18 classes, from 1 to 18). Thus, the groundwaterscape risk code of 1118 corresponds to groundwaterscape 11 (GS11) and risk type 18 (RT18); code 108 to GS1-RT8, code 118 to GS1-RT18, and so on.

5.4 Results

5.4.1 Groundwater risk types synthesize Earth system change and potential for inequality emergence in groundwater-connected systems

We first assess the spatial distribution of individual groundwater system risks (Figure 5.3a). Some risks show constrained extents, such as for agricultural intensification which are bounded by existing cropping patterns while others are globally prevalent, such as the intensification of extreme precipitation. Overlaying these risks reveals regions where multiple risk hotspots co-occur. Regions facing the greatest number of co-occurring risks include the Ganges-Brahmaputra Basin, the Tigris-Euphrates river system, and across central Sub-Saharan Africa. Mapping multiple risks in this way highlights one aspect of the ‘wickedness’ (*sensu* Rittel & Webber, 1973) of groundwater management which must navigate multiple, diverse, and conflicting challenges in these regions.

Eighteen (18) groundwater risk types (RT) emerge from the underlying global risk data (Figure 5.3b) and present a portrait of the multiple, interdisciplinary, and co-occurring global groundwater sustainability challenges. The risk types are distributed heterogeneously around the world in large contiguous patches. For example, RT17 and RT18 (extensive risks) are prevalent throughout the Indian Subcontinent, RT2 (political tensions with land subsidence threat) across much of the

American Midwest, RT14 (gender inequality, and water crowding) across the Sahel, RT6 and RT7 (land subsidence and conservation needs) across northern Europe. RT6 (unmet conservation needs) is the most prevalent risk type, covering 13% of global land area, while RT15 (gender inequality, water crowding, and hydropolitical conflict) is the smallest at 1%.

The risk types are not simply based on the presence of a certain set of risks but rather the complete set of all nine risks and their respective magnitudes (Figure 5.3c). For instance, RT4 and RT5 can be grouped together (group III) as regions facing similar magnitudes of land use change and land subsidence threat, yet RT4 faces considerably greater hydropolitical tension while RT5 has greater conservation needs. In this way, we have similarly grouped all groundwaterscape risks into six groupings as guided by agglomerative clustering (Figure 5.3c). RT1-RT3 share risk profiles (group I) that have severe hydropolitical tension and potential land subsidence, RT6-RT10 (group II) represent areas characterized by a single risk or have generally moderate risk magnitudes, RT11-RT13 (group IV) are characterized by agricultural intensification potential and unmet conservation needs, RT14-RT16 (group V) have acute gender inequality and increased water crowding, and RT17-RT18 (group VI) are facing extensive risks (each with the majority of all risks exceeding their respective global mean magnitude).

The risk types reveal that not all risks consistently co-occur, such as how the threat of land subsidence is found alongside hydro-political tension in RTs 1-3 (group I), and RT17 and RT18 (group VI), but not in RT7 or RT8 (group II). Furthermore, the risk types identify risk co-occurrences whose underlying interactions are not yet explored in the global scale literature. These potential research directions could provide a global perspective to match local-scale research programs on, for instance, gender inequality and groundwater depletion (Kookana et al., 2016; Swaraj & Maheshwari, 2022; Varua et al., 2018) (as found in RT1), tradeoffs and impacts of conflicting land use change and conservation needs (as in RT5), or large-scale dynamics linking land subsidence threat with political tension (as in RT7).

Contrary to conventional sustainability assessments, risk types are not intended to be interpreted along a spectrum of severity, and we make no assessment on the need to prioritize groundwater sustainability or specific actions within one risk type versus another. Rather, the risk types present a set of risk configurations that describe specific challenges and opportunities to groundwater management and sustainability.

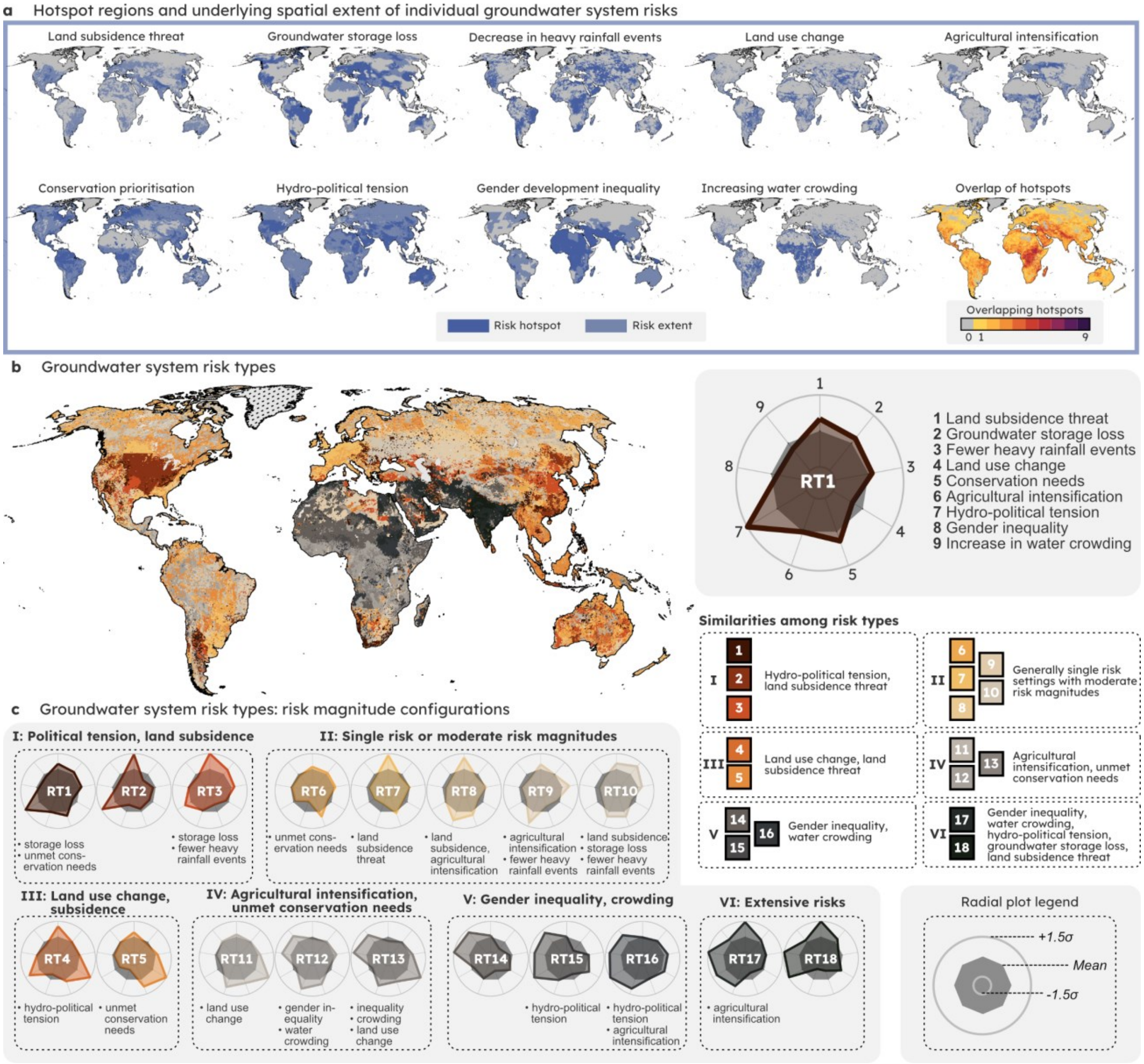


Figure 5.3. Groundwater system risks and risk types.

(a) Spatial extent of individual groundwater system risks and individual risk hotspot regions. 70th percentiles of each risk's global distribution were used as thresholds to determine hotspot extents. Each map in panel (a) is reproduced at a larger size in Figures V.6 and V.7. (b) Groundwater system risk types. (c) Radial plots representing characteristic configurations of risk magnitudes per risk type. See inset legend (bottom right in figure) for radial plot interpretation.

5.4.2 Groundwaterscape risks characterize the groundwater sustainability problem space

Groundwaterscape risks (Figure 5.4) offer a spatial representation of our conceptualization of the global groundwater sustainability problem space (Figure 5.2). Groundwaterscape risks are derived by overlaying groundwaterscapes (see Box 5.2) with groundwater system risk types (Figure 5.3b) and thus capture predominant patterns across both groundwater system functions and groundwater system risks. A total of 270 groundwaterscape risks are identified, representing a first attempt to define the diversity of settings that comprise the global challenge of groundwater sustainability.

Box 5.2. Overview of global groundwaterscapes.

As this study focuses on the integration of groundwaterscapes with groundwater risk types to derive groundwaterscape risks, we defer to Huggins et al. (2024; Chapter 4) for their complete description. Yet, as a baseline understanding is necessary to understand groundwaterscape risks, we provide a brief overview here. Groundwaterscapes (Figure 5.4b) are landscape units that describe global patterns in groundwater's large-scale Earth system, ecosystem, food system, and water management system functions. The 15 groundwaterscapes can be sorted into seven groups based on similar function configurations and are listed here in place of a full description of each groundwaterscape. These groups consist of: arid and desert regions with minimal functions (n = 3), underserved populations with ineffective governance (n = 2), Earth system functions in non-agricultural regions (n = 2), GDEs or moderate Earth system functions in countries with effective governance (n = 2), agricultural regions with high (n = 2) and low (n = 2) groundwater use, and extensive GDEs in non-agricultural regions (n = 2).

Unique groundwaterscape risks characterize the spatial extent of groundwater systems that function in similar ways and face similar risks. To demonstrate this, we plot the distribution of one groundwaterscape risk (GS4-RT14, Figure 5.4c) which represents the overlapping extent of groundwaterscape 4 (underserved populations with ineffective national governance and some terrestrial GDEs) with risk type 14 (gender inequality and increasing water crowding). We find GS4-RT14 to cover broad and spatially contiguous patches across the Sub-Saharan Afrotropics bioregion, particularly within the central Sahel and Sudanian savanna and in bushlands across the Horn of Africa. This approach to mapping groundwaterscape risk enables the identification of regions where the same groundwaterscape exists but with a different risk type (green regions in

Figure 5.4c) and regions where the same risk type exists but with a different groundwaterscape (purple regions). In this way, the groundwaterscape risks are a tool to conceptualize and visualize the similarity and regional differences in the groundwater sustainability problem space.

The 270 groundwaterscape risks are not distributed evenly across the global domain. Half of the terrestrial surface area is covered by just 38 groundwaterscape risks, while 75% coverage is reached at 81 groundwaterscape risks (Figure V.3a). This tail-heavy distribution is even greater when evaluated by population count as half of the global population is found within just 19 groundwaterscape risks, while 75% of the population resides within 55 groundwaterscape risks (Figure V.3b). Nearly one-tenth (9%) of the global population live within groundwaterscape risk GS10-RT18 (Figure V.3c), which corresponds to smallholder farming highly reliant on groundwater irrigation with extensive risks including land subsidence, groundwater storage loss, political tensions, gender inequality, and water crowding (Figure V3d). Thus, while the global set of groundwaterscape risks already constrains the global problem space to a finite number of settings, a much smaller set of groundwaterscape risks characterize the majority of the global land surface and population.

This reality stems from certain groundwaterscapes co-occurring more frequently with certain risk types that others (Figure V.4). For instance, 40% of GS11 (industrial agriculture reliant on groundwater in nations with effective governance) overlaps with RT2 (political tension and land subsidence). Similarly, 33% of GS7 (large storage capacity and climate coupling in non-agricultural regions and nations with effective governance) co-occurs with RT6 (unmet conservation needs and moderate threat of land subsidence). There are also groundwaterscapes with no dominant risk type (such as GS6, GS8 and GS10, for which no single risk type covers over 20% of their extent).

To further understand the relationship between groundwaterscapes and risk types, we assess the underlying correlations between input data within and across the two system classifications (Figure V.5). For instance, global land subsidence is correlated with groundwater irrigation, field size, storage capacity, and government effectiveness; conservation needs are correlated with aquatic groundwater-dependent ecosystem density and negatively with gender inequality; and water crowding is negatively correlated with drinking water access, government effectiveness, field size, and positively correlated with gender development inequality. These statistical relationships additionally point to the interwoven and compounding challenges of groundwater management and sustainability in social-ecological systems.

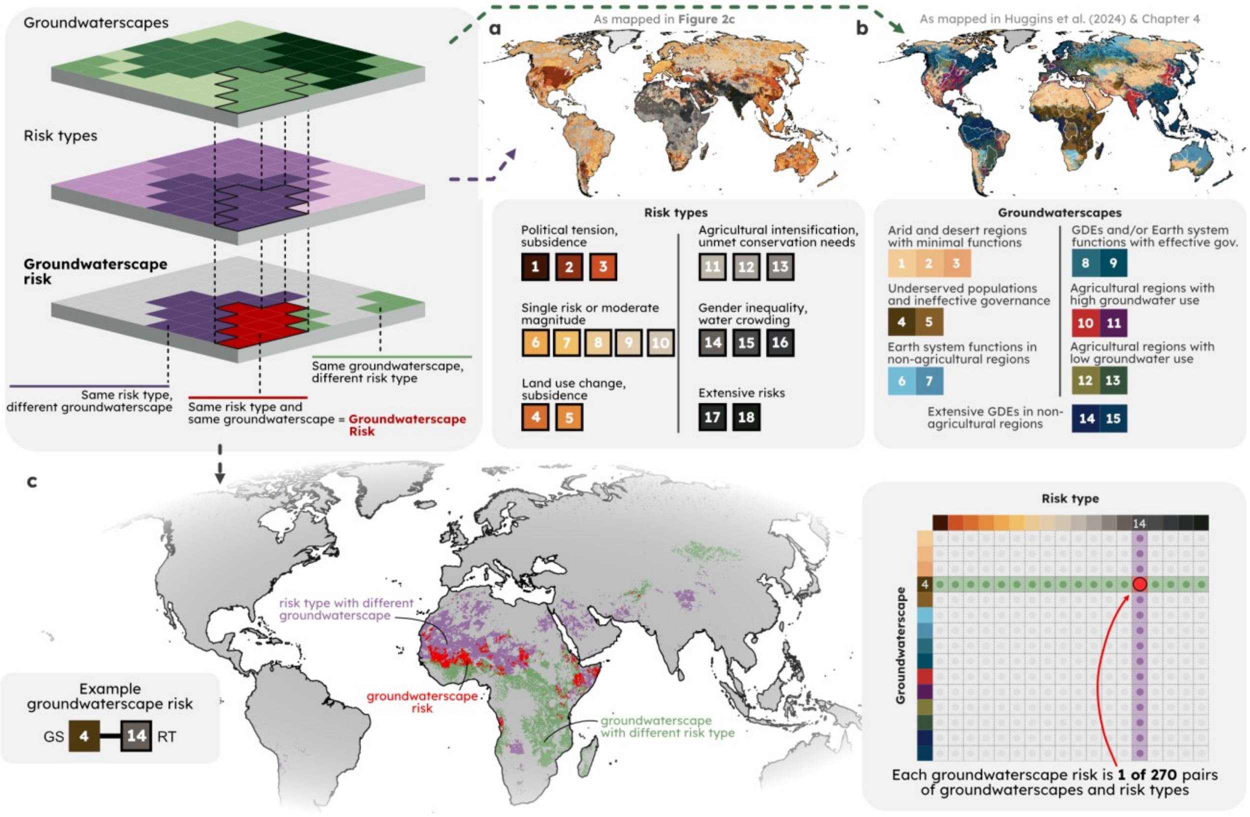


Figure 5.4. Groundwaterscape risk derivation and example.

(a) Groundwater risk types. (b) Groundwaterscapes. (c) Example groundwaterscape risk (GS4-RT14), representing one of the 270 groundwaterscape risks covering the global domain.

5.5 Discussion

5.5.1 Groundwaterscape risks: a guiding tool for solution development and transfer

There is general agreement that groundwater management and governance should account for the complex, multidimensional nature of the resource (Closas & Villholth, 2020; Curran et al., 2023; Nelson, 2022; Ostrom, 1993; Villholth et al., 2018). Yet, we observe there to be a mismatch between how data are often collected, analysed and used and the ideals of these integrated approaches. We find this to be particularly the case at large scales, where existing global support tools for water resource decision makers predominantly focus on the physical aspects of the resource and insufficiently capture the social-ecological complexity and context of groundwater systems. Thus, we foresee the potential for the concept of groundwaterscape risk to be an advance in itself that can enrich groundwater sustainability support tools by moving away from a normative, overgeneralized model of "the groundwater crisis" and towards a contextually-rich understanding of the many and varied groundwater sustainability challenges, enabling a more descriptive and holistic characterization of the multidimensional challenges facing governance bodies and sustainability organizations.

Explicitly linking groundwater to social-ecological system risks imagines the pursuit of groundwater sustainability as a transformative tool to support social well-being, ecological integrity, social and environmental justice, and Earth system stability. Thus, despite being framed around risks, we present this analysis to support context-specific groundwater sustainability strategies. These groundwaterscape risks can support management and governance efforts by providing a wide set of functions and risks to consider in solution design and evaluation processes.

Groundwaterscape risks provide data-driven descriptions of regional groundwater sustainability challenges that help bridge "global narratives with local realities" (Oberlack et al., 2019). For instance, while global groundwater storage analyses highlight aquifers in California's Central Valley (USA) and across Iran as facing similar challenges in dropping water tables (Jasechko et al., 2024), the approach and analysis conducted here reveals how the scope of system risks and set of groundwater system functions at-risk vary substantially between these regions (e.g., smallholder vs. industrial agriculture, presence vs. absence of groundwater-dependent ecosystems, and different magnitudes of agricultural intensification potential, gender inequality, and ecological conservation needs).

Thus, the groundwaterscape risks generate data-driven regional problem descriptions. For instance, a groundwaterscape risk analysis can facilitate considerations on how groundwater sustainability should be defined and operationalized in regions characterized by the co-occurrence of unimproved drinking water in regions with some terrestrial GDEs and in nations with ineffective governance, while navigating risks such as gender inequality and water crowding (e.g., as found in in Burkina Faso, Figure 5.5a). Similar narratives can be generated for agricultural economies in Goiás (Brazil) situated among groundwater-dependent ecosystems undergoing land use change (Figure 5.5b), the political tensions and threat of continued land subsidence in California (USA) within industrial, groundwater-irrigating agricultural regions (Figure 5.5c), and for the intersection of smallholder farming highly reliant on groundwater irrigation in Andhra Pradesh (India) tasked with navigating agricultural intensification, gender inequality, water crowding, and political tensions (Figure 5.5d).

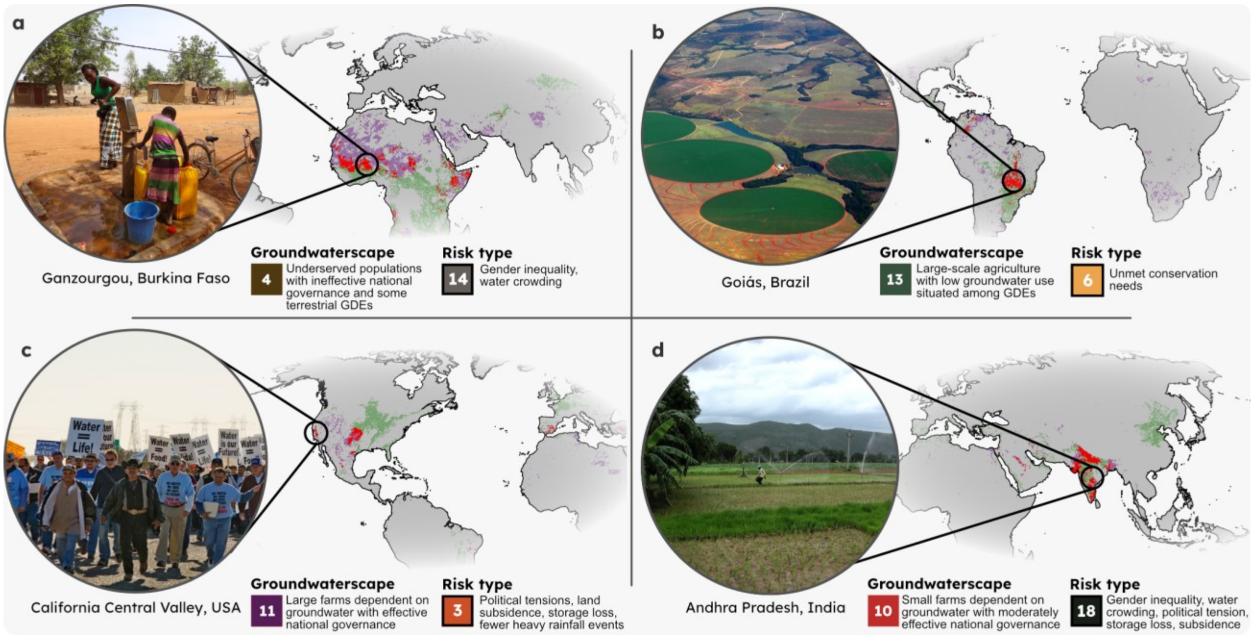


Figure 5.5. Case studies highlighting four groundwaterscape risks.

(a) Ganzourgou, Burkina Faso. (b) Goiás, Brazil. (c) California Central Valley, USA. (d) Andhra Pradesh, India.

As this global mapping highlights the systematic similarities and differences facing groundwater management across distal regions, the maps can be used as tools to consider the suitability of solution transfer across regions. For instance, none of the highlighted regions in Figure 5.5 share a groundwaterscape nor a risk type. These differences may suggest that sustainability strategies pursued in one of these regions could have different outcomes or unforeseen consequences in other regions if contextual differences in groundwaterscape risk are not accounted for. Thus, groundwaterscape risks can support context-specific governance and management and can combat misconceptions of groundwater sustainability "panaceas" (Loring, 2020; Molle & Closas, 2021; Pahl-Wostl et al., 2012).

Alternatively, where groundwaterscape risks are the same across multiple regions, the concept can be used as a tool for solution transfer. While some groundwaterscape risks are regionally constrained (e.g., GS10-RT18, which is found extensively and nearly exclusively throughout the Indian subcontinent) other groundwaterscape risks are more globally distributed. For instance, groundwaterscape risk GS11-RT03 (Figure 5.6a), which characterizes California's Central Valley (USA), is also found in a subset of regions in the Pampas (Argentina), the southern Iberian Peninsula (Spain and Portugal), and the Songliao Plain (China). While it may be surprising that certain distal regions share a groundwaterscape risk, we note that the prevailing approaches to consider groundwater systems are based on fewer considerations in comparison to the broad, multidimensional approach conducted here. We argue that this new lens to evaluate global groundwater systems can be used as a tool to open new perspectives about potential ways to advance large-scale groundwater management. For instance, one use of the groundwaterscape risks could be to use their mapping as a basis for developing solution networks among water managers, researchers, and water professionals to share best-practices, lessons, successes and overarching strategies.

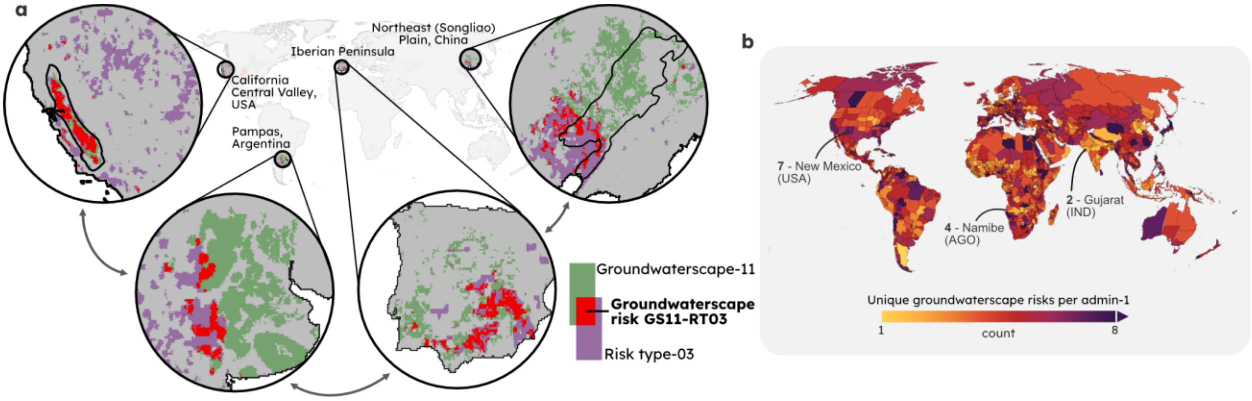


Figure 5.6. Groundwaterscape risks as a tool to facilitate solution transfer and to characterize groundwater management wickedness.

(a) Global distribution of groundwaterscape risk GS11-RT3, with regional inset maps. (b) The number of unique groundwaterscape risks within first-level administrative divisions.

Given groundwater management variously falls across national, subnational, and local levels of governance, decision-making, and collective action, it can be instructive to frame the groundwaterscape risk results relative to administrative boundaries (Figure 5.6b). Some jurisdictions contain multiple groundwaterscape risks (e.g., New Mexico, USA), while others have a more homogeneous composition (e.g., Gujarat, India). In complement to our analysis on the number of overlapping risk hotspots (Figure 5.3), this exercise offers an alternative way to consider the 'wickedness' of groundwater management as jurisdictions with multiple groundwaterscape risks require greater administrative flexibility to manage groundwater across a greater diversity of system functions and risks.

5.5.2 A diverse future for groundwaterscape risks

This study does not intend to frame global groundwater sustainability challenges as a purely technical pursuit. There are myriad groundwater system risks and functions that are not represented in global data, either based on data limitations, and technical or ethical incompatibilities (e.g., groundwater's role in supporting senses of place and ceremonial practices). On this basis, we frame these groundwaterscape risks as an initial step with the intention of shaping the scope of global groundwater sustainability towards increasingly inter- and transdisciplinary directions.

While our analysis integrates an unprecedentedly wide range of groundwater risks and groundwater system functions, the underlying conceptual model does not explicitly consider coastal or cold region dynamics, and centers shallow groundwater systems (i.e., within the upper ~100 meters of Earth's crust) in hydrologic landscapes such as plateaus, high plains, riverine valleys, hummocky terrains, and playa (cf. Winter, 2001). There are also risks we do not include (e.g., virtual water trade) due to overly coarse data or lack of data availability. This study also focuses on the functions and risks to groundwater quantity but overlooks functions and risks to groundwater quality. We view these omissions to overview the multitude of ways the groundwaterscape risk concept can be modified or adapted for application to wider domains.

5.5.3 Groundwaterscape risks as a bridge across disciplines and scales

The larger potential for this analysis is to act as a bridge to connect disciplines and local to global scale communities working on groundwater science and sustainability topics. While we understand and appreciate the tension between the local and global scale, we hold the view that the two scales and their respective research communities are not irreconcilable and that such bridging is necessary for ethical and grounded stewardship of groundwater amongst hydrologists, water managers, and communities (Schmidt, 2023; Schmidt & Peppard, 2014).

Groundwaterscape risks can serve as a support tool for analysis and discussion in inter- and transdisciplinary contexts by facilitating thinking on core uncertainties in such open systems, challenges to discipline-specific theory, approach for data collection, and to identify new research questions. At the same time, the perspectives crystallized in the groundwaterscape risk mapping have the potential to challenge prevailing mental models among hydrogeologists by articulating an explicitly social-ecological system characterization of groundwater systems and risks.

We envision the concept of groundwaterscape risks to hold potential as a 'meeting ground' for scientists, stakeholders, practitioners, citizens, and decision makers, working across disciplines, scales, and holding diverse worldviews, coming together to refine understandings of the many goals embedded in global groundwater sustainability pursuits.

5.6 Data availability

This study uses data from multiple open-access datasets. Source data are documented in Table V.1 and can be downloaded from the persistent web-links provided. The groundwaterscape risks generated in this study will be deposited on Borealis, the Canadian Dataverse Repository, upon manuscript acceptance.

5.7 Code availability

Environment	R project for statistical computing (R Core Team, 2023)
R packages	
terra	Hijmans et al. (2024) https://cran.r-project.org/package=terra
kohonen	Wehrens & Kruisselbrink (2018) https://cran.r-project.org/package=kohonen
aweSOM	Boelaert et al. (2022) https://cran.r-project.org/package=aweSOM
clusterSIM	Walesiak & Dudek (2020) https://cran.r-project.org/package=clusterSIM
landscapemetrics	Hesselbarth et al. (2019) https://cran.r-project.org/package=landscapemetrics

tmap	Tennekes et al. (2023) https://cran.r-project.org/package=tmap
ggplot2	Wickham et al. (2024) https://cran.r-project.org/package=ggplot2
MetBrewer	Mills (2022) https://cran.r-project.org/package=MetBrewer
Script repository	https://github.com/XanderHuggins/gwscape-risks
Data repository	Borealis (https://borealisdata.ca/), pending manuscript acceptance.

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Image inserts used in Figure 5.5 are sourced from the Water Alternatives photo collection under a CC BY-NC license. Individual photographers include Dominique Mercier, Jake Burke, California Department of Water Resources, and Hervé Théry. Composite figures were assembled in Affinity Designer (https://affinity.serif.com/en-us/designer/).

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Chapter 6

HOTSPOTS FOR SOCIAL AND ECOLOGICAL IMPACTS FROM FRESHWATER STRESS AND STORAGE LOSS

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Huggins, X., Gleeson, T., Kummu, M., Zipper, S.C,. Wada, Y., Troy, T.J. & J.S. Famiglietti. (2022). Hotspots for social and ecological impacts from freshwater stress and storage loss. *Nature Communications*, *13*, 439. https://doi.org/10.1038/s41467-022-28029-w

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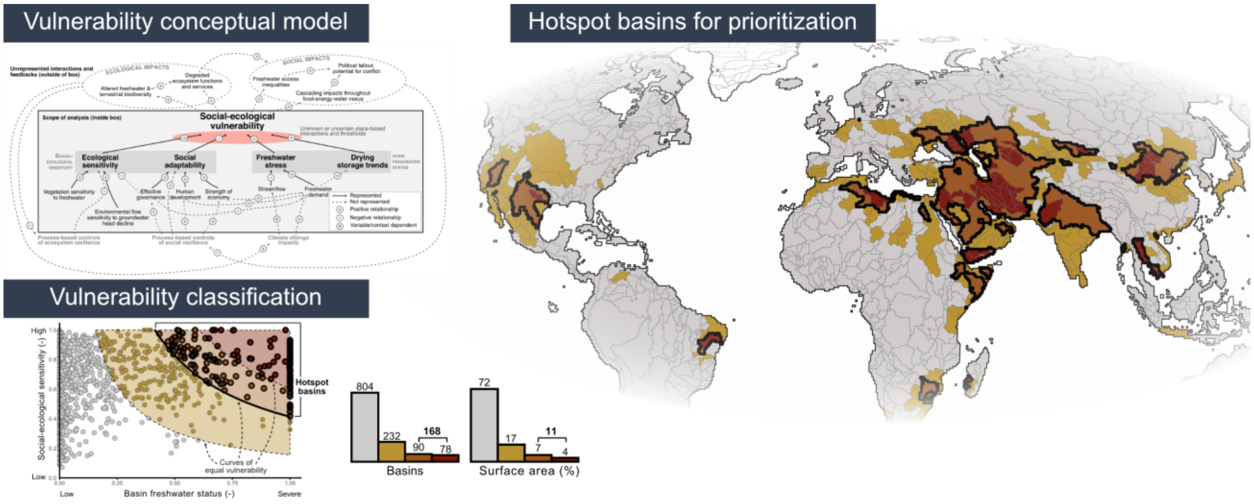
Key points:

- Derives social-ecological vulnerability at the river basin scale by integrating freshwater stress and freshwater storage loss with indicators of ecological sensitivity and social adaptive capacity.
- Identifies 168 basins as vulnerability hotspots for global prioritization.
- Compares vulnerability results with levels of integrated water resources management and finds that transboundary hotspot basins have lower levels of integrated management than non-transboundary hotspot basins.

Figure 6.1. Graphical abstract.



6.1 Abstract

Humans and ecosystems are deeply connected to, and through, the hydrological cycle. However, impacts of hydrological change on social and ecological systems are infrequently evaluated together at the global scale. Here, we focus on the potential for social and ecological impacts from freshwater stress and storage loss. We find basins with existing freshwater stress are drying (losing storage) disproportionately, exacerbating the challenges facing the water stressed versus non-stressed basins of the world. We map the global gradient in social-ecological vulnerability to freshwater stress and storage loss and identify hotspot basins for prioritization (n = 168). These most-vulnerable basins encompass over 1.5 billion people, 17% of global food crop production, 13% of global gross domestic product, and hundreds of significant wetlands. There are thus substantial social and ecological benefits to reducing vulnerability in hotspot basins, which can be achieved through hydro-diplomacy, social adaptive capacity building, and integrated water resources management practices.

6.2 Introduction

Humans and ecosystems, alongside hydrological and biogeochemical cycles, are a deeply coupled and global social-ecological system (Falkenmark, 1977; Falkenmark et al., 2019; Falkenmark et al., 2021). Social-ecological systems are complex adaptive systems formed by interactions and feedbacks between biophysical and social processes (Folke et al., 2016; Ostrom, 2009), and freshwater is fundamental for flourishing and resilient ecosystems, societies, and the larger Earth System (Falkenmark et al., 2021; Steffen et al., 2021; Gleeson et al., 2020; Rockström et al., 2014). Global freshwater storage and flows are dynamic with oscillations and persistent trends driven by the combined influence of human activity, climate change, and natural variability on sub-seasonal to multi-decadal timescales (Rodell et al., 2018; Haddeland et al., 2014). Human dominance over the water cycle is increasingly recognized (Abbott et al., 2019) and the continued development of global hydrological models enables higher fidelity representations of human and climate change impacts on water resources (Bierkens et al., 2015). However, the science of understanding the reciprocal impacts of freshwater stress and storage trends on humans and ecosystems remains in its infancy at the global scale. We argue that studying both directions of this coupled social-ecological system (i.e., social-ecological activity impacts on freshwater and freshwater impacts on social-ecological activity) is crucial to confronting global freshwater challenges, yet the latter has received considerably less attention. In this paper, we consider the potential for freshwater stress and storage loss to impact humans and ecosystems. We do this by synthesizing a subset of the few but critical global ecohydrological and sociohydrological datasets with freshwater storage, freshwater withdrawal, and streamflow datasets (see Table VI.1).

We seek to build on the existing literature on global freshwater scarcity and security topics, which broadly address social and ecological impacts of freshwater-related stresses and hazards. We refer here to freshwater scarcity studies as those which evaluate the ratios of water use to streamflow and streamflow per capita, typically at the basin scale, (e.g., Vörösmarty et al., 2000; Kummu et al., 2010; Kummu et al., 2016; Wada et al., 2014) and to freshwater security studies as those which integrate multidimensional indicators of physical, chemical, socioeconomic, and institutional factors and aggregate using grid-based, basin, or administrative discretization schemes (e.g., Padowski et al., 2015; Gain et al., 2016; Vörösmarty et al., 2010). While water scarcity assessments exclusively focus on freshwater stress, it is only one element of physical water security and thus only one component of water security assessments. Both approaches, however, have important limitations that constrain their ability to support specific conclusions and drive policy implementation regarding the impacts of freshwater-related hazards such as stress and storage loss.

For instance, freshwater scarcity assessments typically apply globally consistent classification schemes which do not represent important spatial variations in social and ecological sensitivities and responses. A few holistic derivatives of freshwater scarcity assessments, such as the social water stress (Ohlsson et al., 2000) and water poverty (Sullivan et al., 2002) indices, have only been evaluated at the national level. Alternatively, water security assessments consider water scarcity as just one of many input variables. These assessments typically aggregate multidimensional indicators of different aspects of water security, which can lead to similar water security outcomes with different input indicator combinations. As a result, water scarcity impacts become challenging to isolate from final water security assessment results. Furthermore, this aggregation approach does not consider interactions or relationships between elements of water security which are critical determinants of social-ecological system behavior (Chapin et al., 2009).

In this paper, we combine the strengths of water scarcity and water security research, and address their limitations by integrating concepts from social-ecological systems research. We combine concepts from these fields to address the following core objectives of this study: (1) Asses the global co-occurrence of freshwater stress and freshwater storage trends at the basin scale. (2) Analyze the relationship between social adaptive capacity and ecological sensitivity indicators with freshwater stress and storage trends. (3) Derive the global gradient in social-ecological vulnerability to freshwater stress and storage trends by considering all indicators listed above, and identify hotspot basins as those with high vulnerability values with respect to the global distribution. (4) Evaluate current levels of integrated water resources management within hotspot basins. Basins, at various scales, are an increasingly used and particularly suitable geospatial unit of analysis for hydrologically-based social-ecological systems analysis (Varis et al., 2019a). In this study, all analyses are performed at a large basin scale (n = 1,204; median area ~70,000 km^2). Input data align to the year 2015 as best as possible, and data are summarized to the basin scale by computing the area-weighted basin average or within-basin sum, depending on the intensive or extensive nature of each dataset (see Section 6.3 Methods and Appendix VI). See Box 6.1 for definitions of key terminology used in this paper.

Box 6.1. Key terminology.

Freshwater stress: The ratio of annual freshwater withdrawal (W) to annual streamflow (Q). We refer to basins with W/Q ≥ 10% as stressed basins and those with W/Q ≥ 40% as highly stressed basins.

Freshwater storage trends: Year-over-year trends in total freshwater storage based on satellite observations over the 2002–2016 time period. Total freshwater storage is a vertically aggregated measure of water storage that includes groundwater, soil water, surface water, canopy water, and ice and snow water equivalents where present. For simplicity, we refer to negative freshwater storage trends as drying trends or storage loss and positive trends as wetting trends or storage gain.

Basin freshwater status: An integrated indicator that combines normalized freshwater stress and normalized freshwater storage trends at the basin scale. High indicator scores are assigned to basins with co-occurring freshwater stress and drying trends. We refer to high freshwater status scores through status severity.

Vulnerability: The likelihood of society and ecosystems to experience harms due to exposure to freshwater stress and storage loss when considered together as a basin's freshwater status. This vulnerability definition is an application of Turner et al.'s (2003) generic definition. Vulnerability is quantified using social adaptability, ecological sensitivity, and basin freshwater status indicators. Social adaptability and ecological sensitivity indicators are described in the text and Methods (Section 6.3).

Hotspot basin: Highlighted basins that possess the greatest vulnerability scores. We identify hotspot basins to support their prioritization in global water resources and integrated management initiatives. Basins are considered hotspots if sorted into ‘high’ and ‘very high’ vulnerability classes following a categorical classification of the numerical vulnerability results.

6.3 Methods

Note: section order has been rearranged from the manuscript version of this chapter so that the methods section is located before the results.

6.3.1 Study approach

The overall study approach is summarized and illustrated in Section VI.1 and Figures VI.1 and VI.2. Below, we focus on the specific methods performed in our analysis. A flow chart of our methodology is shown in Figure VI.3.

6.3.2 Data selection

All underlying data to this study were retrieved from pre-existing, published, and open datasets. We used 12 geospatial datasets and one basin scheme. We aligned our input data to the year 2015 as best as possible. We summarize input dataset selection, justification, and relevant preprocessing in Table VI.1 (Section VI.2).

6.3.3 Geospatial methods

We performed all analyses using the basin scheme of HydroBASINS level 4 (Lehner & Grill, 2013) and at the spatial resolution of 0.5 degrees using the World Geodetic Reference System 1984 ellipsoid (WGS 84). All raw input datasets were harmonized to 0.5 degrees using methods outlined in Table VI.1. All data were summarized to the basin scale by (i) calculating the area-weighted average value of intensive properties within each basin using an algorithm to calculate cell area on a geographic grid (Santini et al., 2010), or (ii) calculating the within-basin sum of extensive properties. We masked basins in Greenland, northern Canada, and several small islands from our analysis due to inconsistent data coverage. This masking reduced the HydroBASINS level 4 discretization scheme to 1204 basins from an original set of 1341 (90% retention by count, 97% by surface area).

6.3.4 Hotspot basins

For a more extensive summary of the theory and justification of our approach, see Appendix VI, Section VI.1. In brief, we base our analysis on the vulnerability definition of Turner et al. (2003) although there is considerable consensus around the general principles of vulnerability (Lei et al., 2014). Our approach is similar to that of Varis et al. (2019a), a recent global assessment of river basin resilience using social-ecological principles. However, our study is of a narrower, more specific scope directed at the potential for social and ecological impacts from freshwater stress and storage loss rather than evaluating broad basin resilience to a wide range of ecological vulnerabilities.

We analyze social-ecological system vulnerability as the product of (i) exposure to freshwater stress and storage loss, represented by the basin freshwater status indicator, and (ii) the combination of ecological sensitivity and social adaptive capacity, which we represent using derived indicators.

Basin freshwater status is derived to represent exposure to the spatial co-occurrence and severity of freshwater stress and freshwater storage trends. This approach enables trends in freshwater storage to differentiate basins of equal freshwater stress as storage trends can aggravate or offset existing stress levels depending on the direction of the storage trend and the existing stress level. The derivation of this indicator is summarized in “Basin freshwater status” (Section 6.3.5).

The social-ecological sensitivity indicator represents the ecological ability to either adapt or absorb freshwater stress or storage perturbations and social ability to adapt proactively and reactively to generic stresses (Varis et al., 2019a). The derivation of this social-ecological sensitivity indicator is summarized below in “Ecosystem sensitivity and social adaptive capacity” (Section 6.3.6). We note that other common quantitative vulnerability approaches incorporate additional considerations such as scaling the sensitivity term by the proximity of the system state to a critical threshold, and undertaking a probabilistic approach to exposure (Luers et al., 2003). Such thresholds, however, are highly uncertain, spatially variable, or unknown in most complex adaptive systems (Scheffer et al., 2009), including human-water systems at the global scale (Xu et al., 2021). Thus, we did not perform this threshold proximity scaling of the sensitivity term. Furthermore, as we evaluated the current state of freshwater stress and the existing trend in freshwater storage, there was no probabilistic component to our analysis. Vulnerability was thus represented in our analysis through Eq. (6.1).

$$V_i = S_i B_i \tag{6.1}$$

where V represents vulnerability of the social-ecological system, S represents the combined social-ecological sensitivity indicator, and B represents the basin freshwater status, per basin i.

6.3.5 Basin freshwater status

We derived basin freshwater status to compress the bivariate relationship of freshwater stress and storage trends into a single indicator. The indicator is a composite of both freshwater stress and storage trend inputs which we individually normalized using the value of 0.4 times annual streamflow (Q) per basin. The composite indicator (Eq. 6.4) is the arithmetic mean of the normalized freshwater stress indicator (Eq. 6.2) and the normalized storage trend indicator (Eq. 6.3). Our freshwater stress calculation differs from other freshwater stress studies which consider water sharing rules between upstream and downstream basins (Kummu et al., 2016) and treat arid regions separately (Hofste et al., 2019). As these existing freshwater stress dataset are not available at our operating scale of HydroBASINS level 4, we calculated the ratio of withdrawal to streamflow within each basin and interpret this ratio as an approximate, relative measure of freshwater demand to within-basin generated renewable freshwater. We normalized both freshwater stress and freshwater storage trends by 0.4Q as this threshold is used throughout the freshwater stress literature to denote high basin stress levels (Vörösmarty et al., 2000; Kummu et al., 2016; Falkenmark & Lundqvist, 1998; Smakhtin et al., 2004). Following common approaches in other global indicator-based assessments, we bound both normalized indicators to a maximum magnitude of 1, i.e., we set an upper limit of 1 for the normalized freshwater stress results by setting all values >1 to 1, and set upper and lower limits of 1 and −1 for the normalized storage trends by setting all vales <−1 to −1 and >1 to 1. We also ‘flipped’ the normalized storage trend indicator (i.e., multiplied the indicator by −1) so that drying trends drying trends correspond to positive indicator scores for consistency with the freshwater stress indicator for which greater (more positive) values correspond with greater levels of stress. Basin freshwater status was calculated as the arithmetic mean of these two indicators, with a minimum value set to 0 as negative values are possible where wetting trends offset existing freshwater stress. Where large earthquakes interfered with storage trend observations (i.e., the 2011 Tohoku earthquake and the 2004 Sumatra-Andaman earthquake) (Rodell et al., 2018), basin freshwater status was set to the independent normalized freshwater stress indicator alone. Input data for these indicators are shown in Figure VI.4.

$$F_i = min\left(\frac{W_i}{0.4Q_i}, \qquad 1\right) \tag{6.2}$$

$$T_i = max\left(min\left(\frac{\frac{dTWS_i}{dt}}{0.4Q_i}, 1\right), \qquad -1\right) \times -1 \tag{6.3}$$

$$B_i = max\left(\frac{F_i + T_i}{2}, \qquad 0\right) \tag{6.4}$$

where F is the freshwater stress indicator, T is the normalized freshwater storage trend indicator, and B is basin freshwater status. W represents annual freshwater withdrawals (mm year^{-1}), Q represents annual streamflow (mm year^{-1}), and dTWS/dt represents year-over-year trends in freshwater storage (mm year^{-1}), per basin i.

6.3.6 Ecosystem sensitivity and social adaptive capacity

We derived an indicator to represent general ecological sensitivity to freshwater stress and storage loss as no existing dataset fit this use. We combined data products from de Graaf et al. (2019) and Seddon et al. (2016), which represent the most relevant global ecohydrological studies. de Graaf et al. (2019) used a global surface water-groundwater model to estimate the groundwater head decline at which environmental flow limits are transgressed for all basins in which there is currently groundwater pumping. Seddon et al. (2016) developed the Vegetation Sensitivity Index to quantify the sensitivity in vegetation productivity to anomalies in three climate variables: water, temperature, and cloudiness, where water anomalies were represented by the ratio of actual evapotranspiration to potential transpiration. We incorporated only the water-specific component of the Vegetation Sensitivity Index. In brief, the de Graaf et al. dataset represents the sensitivity of environmental flows to changes in groundwater storage in basins where groundwater is currently being withdrawn, while the Seddon et al. dataset represents vegetation sensitivity to anomalies in soil moisture and shallow groundwater storage.

To combine these ecohydrological datasets of different dimensions which simultaneously contribute to our overall understanding of ecological sensitivity to changes in freshwater storage, we performed a purely statistical, percentile-based approach. At the gridded resolution of 0.5 degrees, we converted both input datasets into area-weighted percentile datasets, where a value of 1 represents the grid cells with the global maximum values (99th–100th percentile) per dataset and values of 0.01 represent the grid cells with the global minimum values (0th–1st percentile) per dataset and where all values in-between apply to an equal proportion of the land surface. We then (i) computed the average value of both percentile-transformed datasets within each basin, (ii) averaged the two independent averages from (i) to combine the two datasets, and (iii) then normalized all basins by the global maximum basin value. This approach ensured that the most-sensitive basin received an ecosystem sensitivity indicator score of 1 and the least-sensitive basins received scores near 0. It is readily acknowledged, however, that a process-based derivation of ecosystem sensitivity that integrates groundwater, streamflow, and soil moisture considerations would be a superior alternative however none exist to our knowledge. The input data and derivation of the ecosystem sensitivity indicator are shown in Figure VI.5.

To reconcile the ecological sensitivity indicator (where higher values correspond with more sensitive basins) with the concept of social adaptive capacity, we inverted the dataset of social adaptive capacity so that greater values corresponded with lower adaptability. We sourced the adaptive capacity dataset from Varis et al. (2019b), as outlined above. Social adaptive capacity is shown in Figure VI.6.

We combined ecosystem sensitivity and adaptive capacity indicators using the fuzzy sum operation (Eq. 6.4). The fuzzy sum is an increasing linear combination operator that ensures the sum is no less than the largest input value, yet the contribution of subsequent inputs decrease as inputs overlap. If all inputs are normalized [0, 1], the fuzzy sum converges on the upper limit (i.e., 1). The fuzzy sum is increasingly used in geospatial applications, such as in landscape integrity mapping (Kennedy et al., 2019) and we used the fuzzy sum as it eliminates the need to weight the ecosystem sensitivity and adaptive capacity inputs, which would introduce an additional degree of subjectivity. The fuzzy sum operation to derive the social-ecological sensitivity indicator is shown in Equation 6.5. The resulting social-ecological sensitivity indicator and its input datasets are shown in Figure VI.7.

$$S_i = 1 - (1 - E_i)\big(1 - (1 - A_i)\big), \tag{6.5}$$

which reduces to:

$$S_i = 1 - (1 - E_i)(A_i)$$

where S represents the social-ecological sensitivity indicator, E represents ecological sensitivity, and A represents social adaptive capacity, all per basin i. Note that the (1–A) term represents the inversion of the adaptive capacity dataset.

6.3.7 Hotspot identification

As our vulnerability analysis yields a relative gradient in global social-ecological vulnerability, classifying individual basins into vulnerability categories presents a particular challenge as no process-based thresholds are identified in the literature to differentiate the results. For example, while other prominent hotspot mapping initiatives such as the biodiversity hotspots are based on strict criteria (e.g., the biodiversity hotspots must contain 0.5% or 1500 of the world's plant species and must have lost 70% or more of its primary vegetation) (Myers et al., 2000), our social-ecological vulnerability hotspots are deeply interdisciplinary and multivariate, and thus make such explicit criteria challenging to identify. As vulnerability is not directly observable, the pragmatic approach is often to measure and classify relative vulnerability (Luers et al., 2003).

Given the heavy-tailed distribution of vulnerability results, we selected a relative classification algorithm, the Head/Tail Breaks method (Jiang, 2013), to categorize the basins into vulnerability classes and subsequently hotspot basins. The Head/Tail Breaks classification scheme was developed to better represent the hierarchical structure of heavy-tailed distributions compared to other common methods such as Jenks natural breaks optimization. The classification scheme partitions the data into 'head' and 'tail' classes based on the arithmetic mean of the distribution and recursively re-partitions the 'head' class based on the arithmetic mean of the 'head' values until a skewness threshold is reached. For consistency between dimensions, we applied three iterations of the algorithm to all vulnerability distributions (i.e., social, ecological, and social-ecological) to partition the data consistently into four classes rather than use a skew-based stopping criterion. We classified the lowest-level class as non-hotspots (low vulnerability), the 2nd-level class as transitional basins (moderate vulnerability), and the 3rd-level and 4th-level classes as hotspots (high and very high vulnerability). As vulnerability is derived as the product of basin freshwater status and social ecological sensitivity, the class breaks can be represented by reciprocal function curves as shown in Figure 6.4b (labeled as "Curves of equal vulnerability").

While implementing theory-based thresholds of vulnerability would be preferred, it is not realistic given current data availability and process knowledge of ecohydrological and sociohydrological systems at the global scale. We justify our approach as a data-driven, natural classification that characterizes the global gradient in social-ecological vulnerability to freshwater stress and storage loss.

6.3.8 Uncertainty and sensitivity analyses

We performed two uncertainty analyses to explore the impact on our hotspot mapping from potential input data uncertainty, and we performed one sensitivity analysis to identify the impact of subjective decision making in our methodology. These analyses are presented and described in Appendix VI Section VI.4 and Figures VI.8–VI.10.

6.4 Results

6.4.1 The global co-occurrence of freshwater stress and freshwater storage trends

We mapped freshwater stress and trends in freshwater storage at the basin scale and analyzed the co-occurrence of these phenomena (Figure 6.2). Freshwater stress represents the state of demand-driven water scarcity (Kummu et al., 2016) and is defined as the ratio of freshwater withdrawal to streamflow (Figure 6.2a). Trends in freshwater storage, conversely, represent the evolution of total storage, defined as the vertical sum of groundwater, soil moisture, surface water, and snow water equivalent storages (Figure 6.2b). Freshwater stress and storage are linked, as freshwater storage becomes a required source of water during periods when demands exceed supply. As climate change intensifies hydrological extremes globally, the strategic importance of the world's largest store of liquid freshwater, groundwater, will only continue to increase (Taylor et al., 2013). Though studies have focused on global assessments of freshwater stress (Vörösmarty et al., 2000; Kummu et al., 2010; Kummu et al., 2016) and trends in freshwater storage (Rodell et al., 2018), no study to date has mapped these two variables against one another. Doing so provides important context to differentiate basins of equal freshwater stress, as drying trends are likely to exacerbate challenges derived from freshwater stress, while wetting trends may yield offsetting effects. However, as freshwater stress calculations do not differentiate between withdrawals sourced from streamflow or storage, the two variables are not necessarily independent.

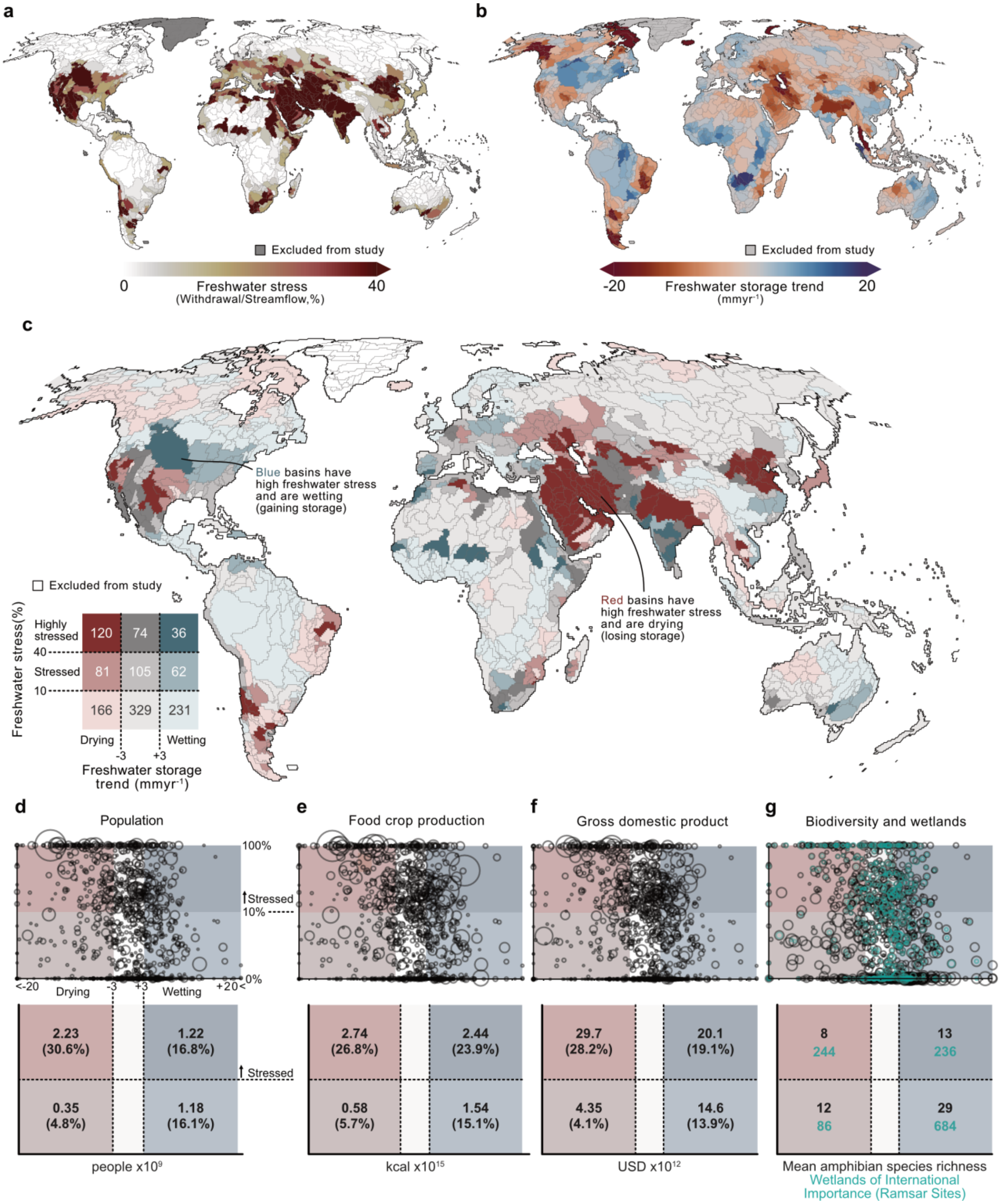


Figure 6.2. Global co-occurrence of freshwater stress and storage trends.

(a) Freshwater stress, derived from freshwater withdrawal and streamflow datasets (see Methods, section 6.3). (b) Freshwater storage trend per basin. (c) Combinations of freshwater stress and storage trend per basin, which together derive basin freshwater status (shown in Figure 6.3b). Values overlaying the legend indicate the number of basins satisfying each set of conditions. For categorical plotting purposes only, ±3 mm year^{-1} is used as the threshold denoting a clear directional storage trend, based on the error level of the underlying observations (Vishwakarma et al., 2018). (d–g) The exposure of social-ecological activity to freshwater stress and storage trends. Each plot represents storage trends as the x-axis coordinate, and log-transformed freshwater stress as the y-axis coordinate with the size of each circle based on the basin's value respective to each plotting dimension.

We found that 201 (42%) of the 478 currently stressed basins (withdrawal/streamflow > 0.10) are simultaneously losing freshwater storage (Figure 6.2c). These basins are located in south and southwestern USA, northeastern Brazil, central Argentina, Algeria, and concentrate throughout the Middle East, the Caucasus, northern India, and northern China. Predominantly, these regions are agriculturally significant and heavily irrigated (Rodell et al., 2018), with the exception of a few basins in South America whose trends are likely the product of natural variability (Rodell et al., 2018). Conversely, 98 (21%) of the currently stressed basins are gaining freshwater storage. The storage trends in these basins have largely been attributed to natural variability with the exception of central India, whose trends are partially attributed to groundwater recovery following groundwater policy change (Rodell et al., 2018). The remaining 179 stressed basins have freshwater storage trends that are smaller than can be definitively interpreted from the satellites monitoring these trends (Vishwakarma et al., 2018). This skew towards negative storage trends (i.e., drying) in the world's water-stressed basins dissipates and even reverses in the non-stressed basins, where drying and wetting trends are found in 23% and 32% of the 726 non-stressed basins, respectively. While previous work has shown that the world's dry regions are becoming drier while the wet regions are becoming wetter (Famiglietti, 2014), this work reveals that the stressed regions of the world are becoming drier while the non-stressed regions of the world have no clear overall trend in freshwater storage.

The encompassed human population, food crop production, gross domestic product (GDP), biodiversity, and wetlands enumerate the potential social-ecological impacts from the current state of global freshwater stress and storage trends. Around 2.2 billion people, 27% of global food crop production, and 28% of global GDP live, grow, and situate in freshwater stressed basins that are drying (Figure 6.2d–f). These totals represent an upper limit as not all social and ecological activity within these basins will be affected by freshwater stress and storage loss, which will depend on local levels of adaptive capacity and ecological sensitivity (Chapin et al., 2009) (our focus in the subsequent sections). Conversely, 1.2 billion people, 24% of global food crop production, and 19% of global GDP are found in stressed basins that are wetting. We find less taxonomic biodiversity in the freshwater stressed and drying basins, and greater biodiversity in unstressed and wetting basins. Roughly the same number of wetlands of international importance are found in stressed and drying basins as in stressed and wetting basins. While these totals represent the magnitude of potentially affected biodiversity and wetlands, taxonomic biodiversity is only one of many critical facets of biodiversity (Su et al., 2021), and freshwater stress and storage trends are but two of many variables impacting global biodiversity (Blowes et al., 2019). Thus, we urge caution in interpreting the role of freshwater stress and storage in driving differences in these biodiversity distributions.

6.4.2 The most vulnerable populations to freshwater stress and storage loss

To better characterize social vulnerability, freshwater stress and storage loss must be placed in the context of social adaptability. We mapped and analyzed the co-occurrence of freshwater stress and storage trends with an existing global dataset of social adaptive capacity (Varis et al., 2019a) summarized at the basin scale (Figure 6.3). Social adaptive capacity (Figure 6.3a), or adaptability, represents “the ability of the system to respond to disturbances” (Turner et al., 2003) and is derived based on input indicators of governance, economic strength, and human development. This consideration of social adaptability enables more representative estimates of social, agricultural, and economic activity that are vulnerable to the co-occurrence of freshwater stress and storage loss. To consider freshwater stress and storage loss together, we developed the basin freshwater status indicator (Box 6.1) where higher values indicate co-occurring freshwater stress and storage loss (Figure 6.3b).

We found 73 basins to possess low levels of social adaptability and severe basin freshwater status (Figure 6.3c). These basins concentrate in Northern, and Eastern Africa, the Arabian Peninsula, and Western, Central, and Southern Asia; although vulnerable basins are also found in northeast Brazil, Southern Africa, and northern China. These basins encompass approximately 1.2 billion people, 12% of global food crop production, and 6% of global GDP (Figure 6.3d–f). Conversely, 119 and 49 basins are found to have similarly severe basin freshwater status yet have moderate or high levels of social adaptability, respectively. These basins are located in southwestern USA and Mexico, Chile and Argentina, the Arabian Peninsula, regions surrounding the Caspian Sea, western Australia, and the North China Plain.

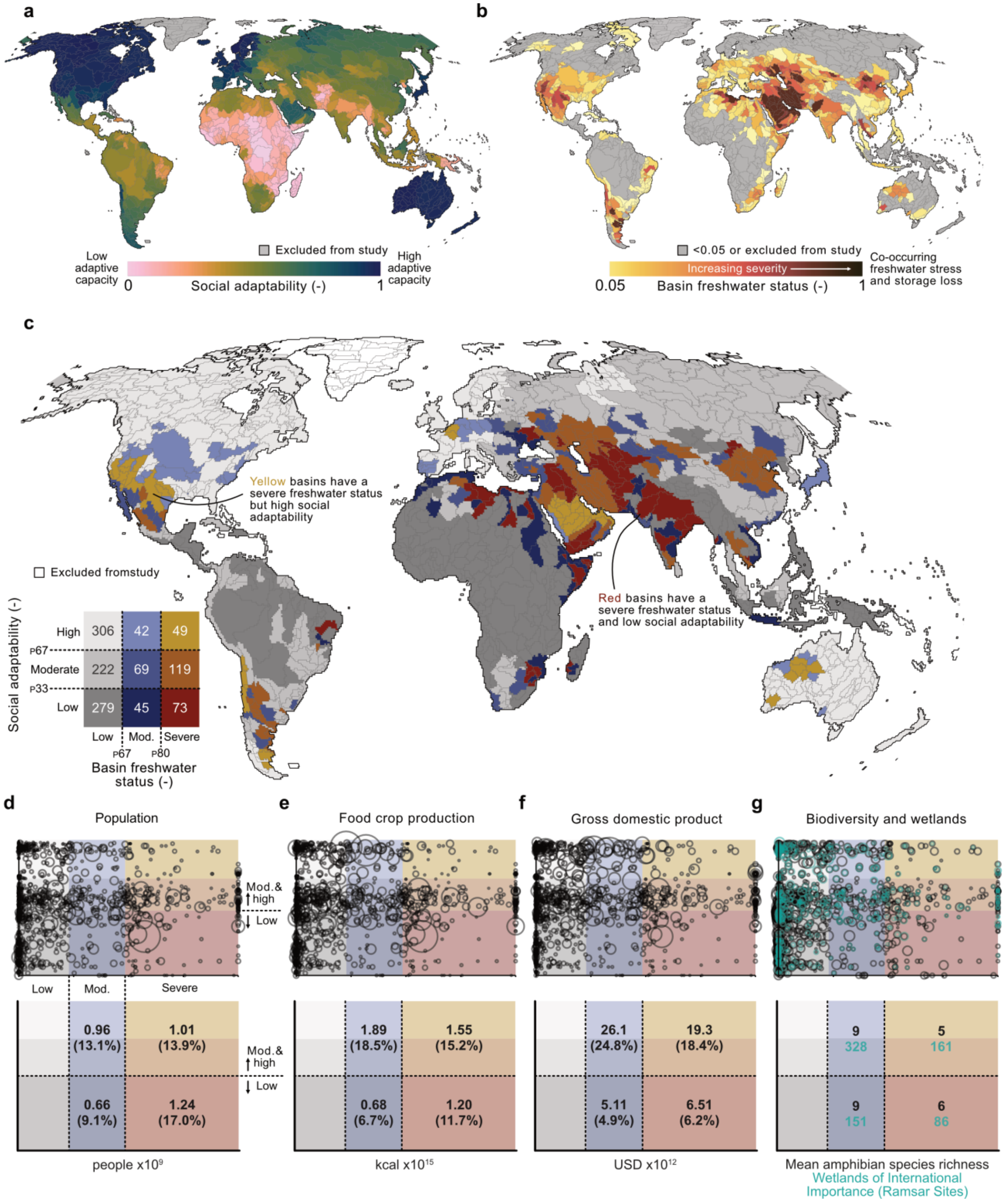


Figure 6.3. The relationship between basin freshwater status and social adaptive capacity. (a) Social adaptive capacity, or adaptability, per basin. (b) Basin freshwater status, representing the combination of freshwater stress and storage trend per basin (see Methods, section 6.3). (c) Combinations of basin freshwater status and social adaptability. Values overlaying the legend indicate the number of

basins satisfying each set of conditions. (d–g) The exposure of social-ecological activity to basin freshwater status (x-axis coordinate) and social adaptive capacity (y-axis coordinate), with the size of each circle scaled based on the basin's value respective to each plotting dimension. These distributions are summarized below each plot. P notation represents the percentile distribution.

These differences in social adaptability across basins with severe freshwater status (i.e., co-occurring freshwater stress and storage loss) raise important economic considerations. First, greater social adaptability likely coincides with greater technological and economic capacity to pursue development. This development may consume greater volumes of freshwater and drive basins towards greater levels of freshwater stress or storage loss, while simultaneously increasing institutional and technical capacity to cope with limited water resources. Furthermore, freshwater stress and storage loss are not certain to induce negative economic impacts on basins, and can lead to positive impacts if a region is able to leverage its comparative advantages (e.g., irrigation efficiency) among other stressed regions (Dolan et al., 2021). Second, the divergent economic situation facing basins with severe freshwater status is particularly evident on a per-capita basis. In severe freshwater status, low adaptability basins, there resides 17% of the global population yet only 6% of global GDP. Conversely, in severe freshwater status basins with moderate-and-greater social adaptability, there resides 14% of the global population and an outsized 18% of global GDP (Figure 6.3d, f). It is thus paramount that global initiatives prioritize and link economic inequality with freshwater goals. One such example is Sustainable Development Goal (SDG) 6.4 ("reduce the number of people suffering from water scarcity"), which we argue should increasingly be linked to targets of SDG 10 ("reduce inequality within and among countries").

6.4.3 Hotspot basins found on all continents

We mapped the global gradient in social-ecological vulnerability to freshwater stress and storage loss at the basin scale and, from this, identified those with the greatest vulnerability as hotspot basins (Figure 6.4). Hotspot mapping has been a successful endeavor within the field of conservation biogeography (Possingham et al., 2005; Myers et al., 2000), and many global hydrology studies have identified regions of exceptional water scarcity and security challenges (e.g., Vörösmarty et al., 2000; Kummu et al., 2010; Kummu et al., 2016; Padowski et al., 2015; Gain et al., 2016; Vörösmarty et al., 2010). Here, we seek to combine and apply these concepts in an integrated global social-ecological vulnerability context. As a useful reference, biodiversity hotspots aim to "maximize the number of species 'saved' given available resources" by asking "where are places rich in species and under threat?" (Whittaker et al., 2005). For comparison, the aim of our hotspot mapping is to ‘minimize the social and ecological impacts from freshwater stress and storage loss given available resources’ by asking ‘what basins with sensitive ecosystems and limited social adaptive capacity are exposed to freshwater stress and storage loss?’

We conceptualize vulnerability as the product of (i) ecological sensitivity, (ii) social adaptive capacity, and (iii) basin freshwater status. To represent ecological sensitivity, we derived an indicator using data products from two global ecohydrological studies that assess broad ecosystem sensitivity to freshwater storage and use (see Methods, section 6.3). To represent social adaptability, we utilized the same adaptive capacity dataset as used in the previous section (Figure 6.3a). To classify the derived global vulnerability results into hotspot basins, we implemented a simple classification algorithm developed for heavy-tailed distributions (Jiang, 2013), which appropriately describes the global vulnerability distribution.

The most vulnerable basins are constrained to regions confronting co-occurring freshwater stress and storage loss. When considering social and ecological vulnerability individually (Figure 6.4e, f), we find spatial variation between ecological vulnerability (Figure 6.4e) and social vulnerability (Figure 6.4f). For instance, several basins in affluent nations with sensitive ecosystems reveal high ecological vulnerability but low social vulnerability (southwestern USA; western Australia). Conversely, several basins in Eastern Africa and northeastern India possess high social vulnerability but low to moderate ecological vulnerability. While these differences are notable and could impact regional strategies, it remains essential in most, if not all, regions that social and ecological vulnerabilities be confronted simultaneously (Folke et al., 2016). For this purpose, we combined ecological sensitivity and adaptive capacity indicators into a combined social-ecological sensitivity indicator (see Section 6.3.6) to map combined social-ecological vulnerability (Figure 6.4a).

We identify 168 basins, representing 14% of all basins and 11% of the global land area considered in our study, as vulnerability hotspots (Figure 6.4a–c). These hotspot basins consist of basins receiving ‘high’ and ‘very high’ vulnerability scores through our classification procedure. Of the 168 basins, 78 (6% of all basins) are classified in the most-severe ‘very high’ vulnerability class, while 90 (7% of all basins) are classified in the ‘high’ vulnerability class. We also identified 232 basins (19% of all basins) as ‘transitional’ basins, which are not classified alongside basins with null vulnerability yet also do not possess extreme values within the global vulnerability distribution. The 78 hotspot basins with ‘very high’ vulnerability represent the multiple epicenters for potential social and ecosystem impacts from freshwater stress and storage loss. These basins are found in Argentina, northeastern Brazil, the American southwest, Mexico, Northern, Eastern, and Southern Africa, the Middle East and Arabian Peninsula, the Caucasus, West Asia, northern India and Pakistan, Southeastern Asia, and northern China.

A total of over 1.5 billion people, 17% of global food crop production, and 13% of global GDP are found within hotspot basins (Figure 6.4d). Of these, ~300 million people, 4% of global food crop production, and 4% of global GDP situate within the 78 ‘very high’ vulnerability basins. Consistent with the relationship between biodiversity and basin freshwater status, we find the most vulnerable basins to be less taxonomically biodiverse than less vulnerable basins. While it is possible that these lower biodiversity levels may have eroded due to freshwater stress and storage loss, a proper investigation is outside the scope of this study and would require a wider array of pressures to be considered. The hotspot basins encompass 157 wetlands of international importance, which we highlight to prioritize their conservation in these vulnerable environments (Table VI.2).

While the degree of social-ecological activity within hotspot basins is substantial, the global proportion of each dimension found in hotspot basins is roughly proportional to the fraction of basins within each vulnerability class. Thus, as the hotspot basins do not contribute disproportionately to global totals of social-ecological activity, we find it important to restate and clarify the motivating purpose of this hotspot mapping. The hotspot basins do not identify the greatest contributors to global social-ecological activity that face severe freshwater challenges. Rather, the hotspot basins are those with sensitive ecosystems and adaptability-limited societies exposed to the co-occurrence of freshwater stress and storage loss, and thus are the basins most likely to suffer social and ecological harms due to these freshwater conditions.

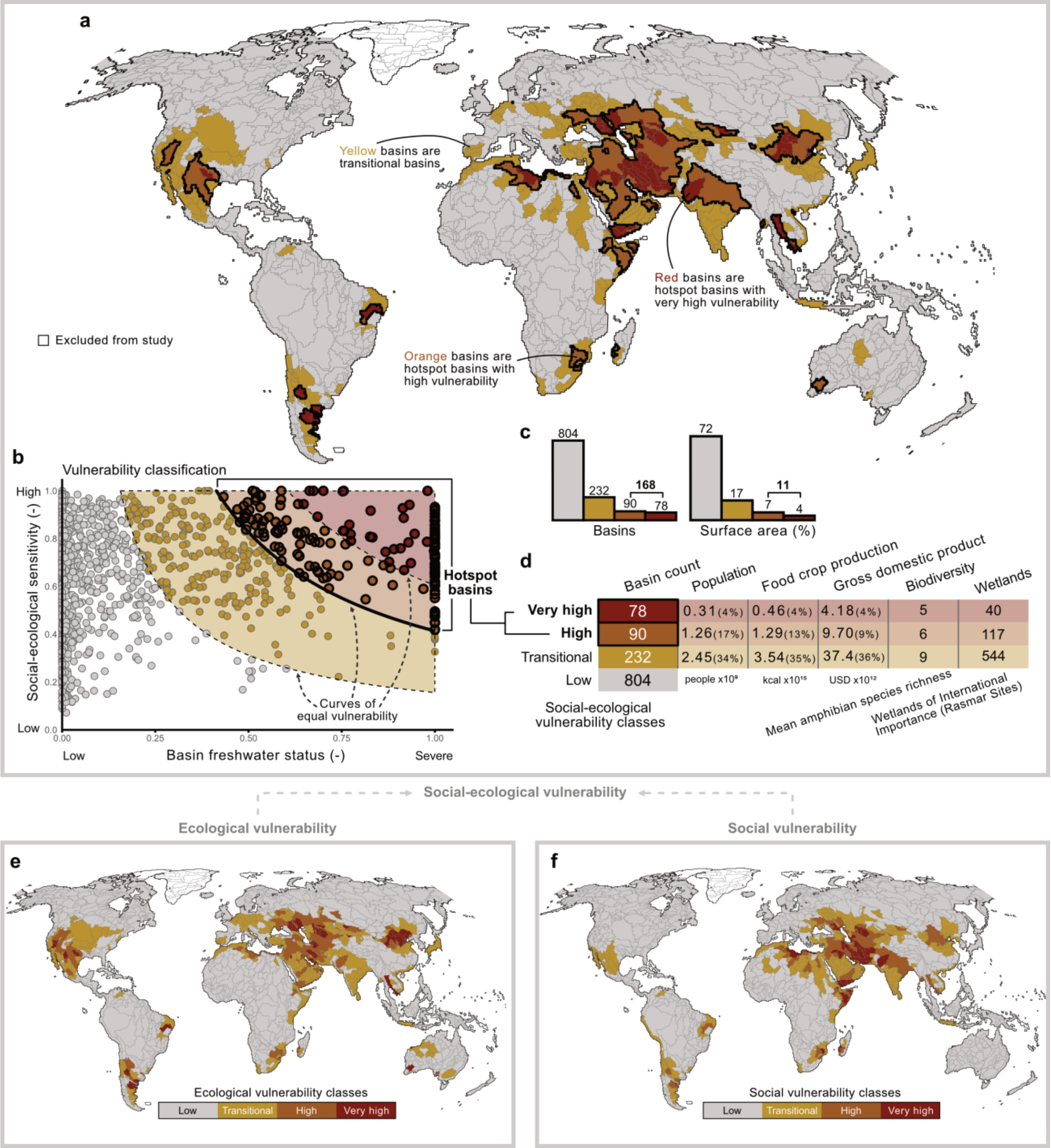


Figure 6.4. Hotspot basins for social and ecological impacts from freshwater stress and storage loss.

(a–d) Social-ecological vulnerability results. (a) Hotspot basins of social-ecological vulnerability to freshwater stress and storage loss. (b) Vulnerability classification, based on the product of basin freshwater status and social-ecological sensitivity to freshwater stress and storage loss (see Section 6.3.7). (c) Histograms of the global distribution of vulnerability classes by basin count and surface area. (d) Summarized social-ecological activity within transitional and hotspot basins. (e) Ecological vulnerability results, presented as vulnerability classes. (f) Social vulnerability results, presented as vulnerability classes. Vulnerability classes for (e) and (f) are derived using the same methods as shown for social-ecological vulnerability in (b).

The identification of hotspot basins shows high levels of consistency across two uncertainty analyses and a sensitivity analysis focused on the impacts of subjective methodological decisions (Appendix VI Section VI.4). We consider individually the impacts of (i) uniform over-estimation and under-estimation of each data input (spatially uniform uncertainty) and (ii) heterogeneous uncertainty in each data input (spatially variable uncertainty) on our hotspot basin results. Performing 10,000 realizations for each uncertainty analysis, we find that 98% of the identified transitional and hotspot basins are identified as at least transitional basins in over 50% the realizations considering spatially uniform uncertainty, and 96% when considering spatially variable uncertainty (Figure VI.8 and Figure VI.9). The subjectivity-focused sensitivity analysis considered 24 alternative methodological configurations and revealed that our identified transitional and hotspot basins are consistently identified across the majority of configurations (Figure VI.10).

6.4.4 Implementation of integrated water resources management is inconsistent across hotspot basins

We compared national implementation levels of integrated water resources management (IWRM) with our global vulnerability results (Figure 6.5). For IWRM implementation data, we rely on the IWRM Data Portal (UNEP-DHI, 2021) which tracks progress on SDG 6.5.1 ("IWRM implementation at the national scale").

IWRM is defined as "a process which promotes the co-ordinated development and management of water, land and related resources, in order to maximize the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems" (Hassing et al., 2009), while the SDG framework notes that IWRM implementation "supports all Goals across the 2030 Agenda" (UN Environment, 2018). Thus, as the IWRM paradigm seeks to guide management of water resources to minimize trade-offs between human well-being, ecological health, and water resources sustainability, assessing implementation levels of IWRM against our vulnerability results provides insight regarding the performance of IWRM globally while simultaneously emphasizing the broad sustainability implications within hotspot basins.

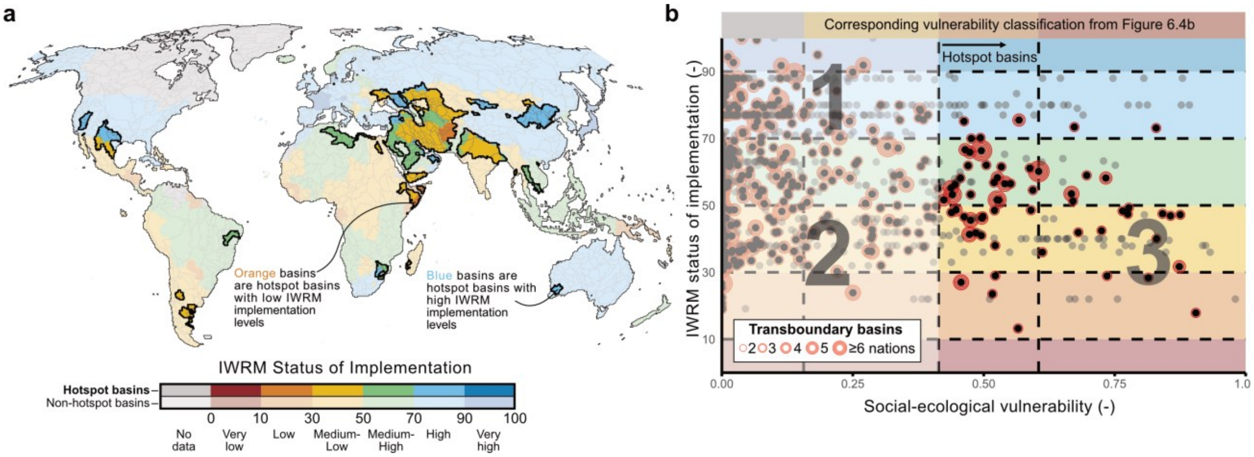


Figure 6.5. Integrated water resources management in hotspot basins.

(a) Map of IWRM implementation overlaid by hotspot basin results. (b) Scatterplot of individual basin values of social-ecological vulnerability (x-axis) and IWRM implementation (y-axis). Transboundary basins are represented by concentric red circles, with the number of circles representing the number of nations present within each basin. See text for interpretation of labels 1, 2, and 3.

Globally, we find no direct relationship between vulnerability and IWRM implementation at the basin scale. There is thus a wide range of IWRM implementation across all levels of social-ecological vulnerability to freshwater stress and storage loss, and there is no indication that IWRM implementation levels are greatest where they are most needed. This finding likely derives from variations in proactive versus reactive governance and management approaches to freshwater challenges across the globe. As our analysis is conducted at a snapshot in time (input data align to ~2015), we can only generate hypotheses about the performance of IWRM globally. For example, basins with high levels of IWRM implementation and low vulnerability (label 1 in Figure 6.5b) have either proactively implemented IWRM, have effectively reduced their vulnerability through IWRM implementation, or simply benefit from a favorable intersection of regional climate and economy.

Alternatively, basins with low levels of IWRM and low vulnerability can be categorized as non-proactive in their IWRM implementation (label 2 in Figure 6.5b). We place particular emphasis here on basins with low levels of IWRM where vulnerability is high (label 3 in Figure 6.5b), which we argue should be the priority basins and regions of SDG 6.5-focused initiatives. Identified nations with low levels of IWRM implementation and very high vulnerability include Afghanistan, Algeria, Argentina, Egypt, India, Iraq, Kazakhstan, Mexico, Somalia, Ukraine, Uzbekistan, and Yemen. As one-third (36%) of all hotspot basins are transboundary (Figure 6.5b), improving basin-level IWRM implementation will require multilateralism and hydro-diplomacy and cannot be left to individual nations acting alone. Furthermore, we observe a lower level of IWRM implementation across hotspot basins that are transboundary versus non-transboundary hotspot basins (mean basin IWRM Data Portal score = 50 vs. 56), suggesting greater multilateralism and cooperation are needed in transboundary basins.

6.5 Discussion

6.5.1 Implications for hotspot basins

There are many possible social and ecological implications for hotspot basins, however these depend on basin-specific relationships among freshwater stress, storage trends, ecological sensitivity, and adaptive capacity. Ecologically, hotspot basins are more likely to suffer from transgressed environmental flows (Richter et al., 2012) with ensuing impacts on freshwater and riparian ecosystems (Poff et al., 2010). Freshwater storage loss is linked to increased drought frequency (Pokhrel et al., 2021), and falling water tables simultaneously decrease ecosystem resilience to drought by limiting root water-uptake (Fan et al., 2017), harming groundwater-dependent ecosystems (Eamus et al., 2015), and perturbing the land surface energy balance (Kollet & Maxwell, 2008). Where freshwater stress and depletion are driven by irrigation, the ecological implications are not limited to within the basin, as irrigation can modify moisture recycling and precipitation patterns across precipitationsheds (Keys et al., 2019), with cascading impacts on potentially distant ecosystems (Keys et al., 2018). Socially, hotspot basins are forced to confront shrinking water resources to satisfy domestic, industrial, and agricultural demands (Falkenmark et al., 1989; Gleick, 1996). Where water tables are dropping and wells are at risk of running dry (Jasechko & Perrone, 2021), groundwater access may intensify social inequalities as only the wealthy may be able to afford to drill deeper wells (Perrone, 2020). Such conditions may trigger conflict and have implications on international security (Varis et al., 2019a). Of the near-700 water conflicts documented since 2000 by The Water Conflict Chronology (Pacific Institute, 2019), two-thirds (68%) are found within either transitional or hotspot basins from this analysis. As this study considers only a subset of water related stresses, this serves to indicate the potentially leading role freshwater stress and storage loss may play in instigating, contextualizing, and sustaining social conflict.

6.5.2 Opportunities for global ecohydrology and sociohydrology

This analysis is a representation of the best-available data for addressing global social-ecological impacts from freshwater stress and storage loss. However, it simultaneously highlights important limitations of existing ecohydrology and sociohydrology research at the global scale. For instance, we were required to develop an indicator to represent ecological sensitivity to freshwater from two existing ecohydrological datasets. However, these two datasets do not represent an exhaustive set of ecosystem processes and functions which may be impacted by changes in freshwater. We also do not address sub-grid variability considerations in either ecological sensitivity or social adaptability, which would independently benefit from specific study. Further, owing to a lack of alternatives, we adopted a rather reductionist and general derivation of social adaptive capacity as a proxy for social ability to respond and adapt to freshwater stress and storage loss. Future work that targets societal responses and relationships to specific freshwater stresses (e.g., as in Di Baldassarre et al. (2013) but applied at the global scale and for various hydrological hazards) would be highly relevant for subsequent studies of this kind. We also only considered direct impacts of freshwater stress and storage loss, and thus did not consider indirect and non-local impacts such as the water and food security impacts of virtual water trade (Dalin et al., 2012; Rosa et al., 2019), which does not yet have data available globally at a sub-national scale.

In closing, we seek to establish a thematic connection between this work and Abbott et al. (2019), who found human activity to be largely absent in water cycle diagrams and suggested this inaccuracy contributes to a “misunderstanding of global hydrology by policymakers, researchers and the public”. In a similar spirit, we observe that social-ecological system impacts of hydrological change are under-considered in global hydrological studies and we argue this underrepresentation contributes to a lack of awareness or misunderstanding of ecohydrological and sociohydrological connections at the global scale. Addressing hydrological phenomena simultaneously as products and drivers of change within the global social-ecological system can only elevate the consideration freshwater will receive in complex, multi-objective, multi-disciplinary decision making. Freshwater stress and storage trends are only two of several critical aspects of freshwater with broad social-ecological sustainability and resilience implications (Falkenmark et al., 2021). Pending data availability, similar analyses can be performed for seasonal and inter-annual variability in water storage and several quality considerations. Developing such a network of studies will support a more comprehensive understanding of the social-ecological resilience implications of global hydrological change.

6.6 Code and data availability

The basin vulnerability and hotspot basin classification results from this study have been deposited in the University of Victoria's Scholars Portal Dataverse (see below). The raw data used in this study are accessible through several data and code repositories. We provide data sources, persistent web links, and descriptions of all data used in this study in Table VI.1.

Environment	R project for statistical computing (R Core Team, 2021)
R packages	
raster	Hijmans (2021) https://cran.r-project.org/package=raster
sf	Pebesma et al. (2021) https://cran.r-project.org/package=sf
gdalUtils	Greenberg et al. (2020) *since removed from the CRAN repository.
spatstat.geom	Baddeley et al. (2021) https://cran.r-project.org/package=spatstat.geom
ggplot2	Crameri et al. (2020); Crameri et al. (2021); Pedersen (2020) https://cran.r-project.org/package=scico
tmap	Tennekes et al. (2023) https://cran.r-project.org/package=tmap
scico	Wickham et al. (2024) https://cran.r-project.org/package=ggplot2
Script repository	https://github.com/XanderHuggins/gwscape-risks; Archived on Zenodo: https://doi.org/10.5281/zenodo.5728475
Data repository	Borealis: https://doi.org/10.5683/SP3/SLR3GF

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Chapter 7

OVERLOOKED RISKS AND OPPORTUNITIES IN GROUNDWATERSHEDS OF THE WORLD'S PROTECTED AREAS

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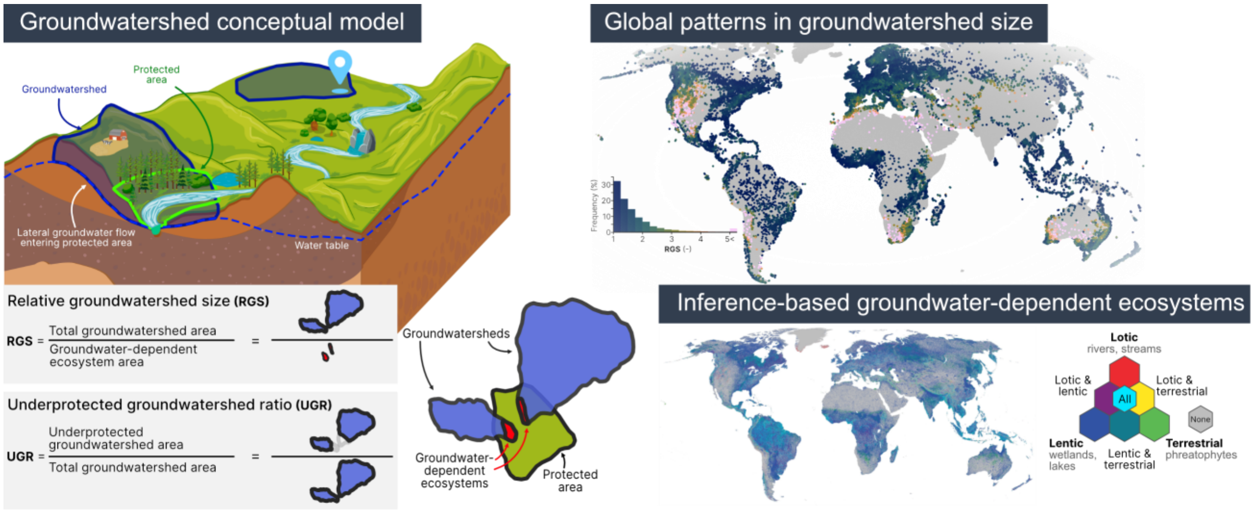
Key points:

- Delineates groundwatersheds (groundwater catchments) for the world's protected areas that contain groundwater-dependent ecosystems.
- Generates a global map of groundwater-dependent ecosystems using an inference-based approach.
- The majority (85%) of protected area groundwatersheds lie partially outside protected area boundaries, highlighting the need to incorporate groundwater protections in area-based conservation strategies.

Figure 7.1. Graphical abstract.



7.1 Abstract

Protected areas are a key tool for conserving biodiversity, sustaining ecosystem services and improving human well-being. Global initiatives that aim to expand and connect protected areas generally focus on controlling 'above ground' impacts such as land use, overlooking the potential for human actions in adjacent areas to affect protected areas through groundwater flow. Here we assess the potential extent of these impacts by mapping the groundwatersheds of the world's protected areas. We find that 85% of protected areas with groundwater-dependent ecosystems have groundwatersheds that are underprotected, meaning that some portion of the groundwatershed lies outside of the protected area. Half of all protected areas have a groundwatershed with a spatial extent that lies mostly (at least 50%) outside of the protected area's boundary. These findings highlight a widespread potential risk to protected areas from activities affecting groundwater outside protected areas, underscoring the need for groundwatershed-based conservation and management measures. Delineating groundwatersheds can catalyse needed discussions about protected area connectivity and robustness, and groundwatershed conservation and management can help protect groundwater-dependent ecosystems from external threats.

7.2 Introduction

Protected and conserved areas are fundamental tools for safeguarding biodiversity and play an important role in improving human well-being and sustaining ecosystem services (Gray et al., 2014; Jones et al., 2018; Janishevski et al., 2014; Possingham et al., 2006; Belote et al., 2017). With the newly agreed Convention on Biological Diversity's Kunming-Montreal Global Biodiversity Framework, Parties to the Convention have committed to ensuring and enabling that by 2030 at least 30 percent of terrestrial, inland water, and of coastal and marine areas are effectively conserved through protected areas and other effective area-based conservation measures (OECMs) (CBD, 2022). Current and past approaches to inland protection, which have focused principally on land-based objectives and measures, have had clear limitations in conserving freshwater ecosystems and species, which have shown staggering declines (Davidson, 2016; Tickner et al., 2020). One frequently discussed reason for the poor performance of protection initiatives for these ecosystems is the lack of consideration given to external (upstream) hydrologically connected freshwater systems (Acreman et al., 2020; Abell et al., 2017). The need to manage human activities in connected lands and waters outside protected areas has been absent from effectiveness discussions and indicators (Geldmann et al., 2021), and the designation of new protected areas is rarely based solely on hydrologic boundaries. Connecting protected area initiatives to surface water processes is an important step; and the potential is apparent across the World Database on Protected Areas, which intersect with hundreds of watersheds.

However, consideration must also be given to groundwater systems. Protected areas face impacts from activities occurring outside of the protected area, such as agricultural drainage, mining, contamination and groundwater pumping, which are transmitted to protected areas through connected groundwater systems. Doñana National Park (Spain) (Suso et al., 1993; Camacho et al., 2022) and Grand Canyon National Park (USA) (Mueller et al., 2017) are two iconic examples where groundwater-mediated impacts have been documented.

The consideration and management of groundwater becomes increasingly important as land and water use intensify around protected areas (Jones et al., 2018; Hansen et al., 2013). Yet, no study has systematically investigated the potential for human activities outside protected areas to impact terrestrial and aquatic groundwater-dependent ecosystems (GDEs) in protected areas through groundwater flow (Figure 7.2). Lateral groundwater flow supplies a substantial proportion of water used by vegetation (Maxwell & Condon, 2016), and changes in land use or land cover can impact downgradient terrestrial ecosystems by changing the quantity and distribution of groundwater (Zipper et al., 2017; Zipper et al., 2018). Groundwater pumping can reduce streamflow and change streamflow from perennial to intermittent or ephemeral (Kustu et al., 2010; de Graaf et al., 2019). Since groundwater has distinct chemical and temperature characteristics and can transmit contaminants such as nutrients (Wondzell, 2015), changes in groundwater levels and flow can introduce pollutants or otherwise alter water quality in protected areas (Martin et al., 2017).

In this study, we estimate the area from which human impacts outside protected areas may propagate to protected areas through groundwater flow systems. We employ a generic, reproducible workflow to map groundwatersheds for GDEs in protected areas (Box 7.1 and Figure 7.3) at the spatial resolution of 30 arcsecond or ~1 km at the equator (see Methods, Section 7.3). We conclude by identifying risks posed to existing protected areas based on levels of human activity and land use modification in the underprotected portions of groundwatersheds and discuss opportunities for improved conservation outcomes.

Box 7.1. What are groundwatersheds?

Note: key terms are bolded and summarized at the bottom of the box.

A **groundwatershed** is the contributing area from which a groundwater system flows to a feature or set of features of interest (Figure 7.3). In this respect, groundwatersheds are the groundwater analogue of surface watersheds. The groundwatershed concept was first introduced by Haitjema (1995) to evaluate groundwater residence times, and similar concepts have been called 'groundwater catchments' (Parker et al., 2016), 'groundwater basins' (Tiedeman et al., 1998) and mapping of 'groundwater divides' (Boutt et al., 2001). However, the concept has seen limited uptake in water science and management, possibly owing to groundwater being an often-overlooked resource and also possibly due to some characteristic differences between groundwatersheds and surface watersheds, such as what features are used as outlet points, which make their delineation and use more challenging.

In arid environments, flat topographies, and regions with complex geologies, groundwatershed divides can be spatially unaligned with surface watershed divides (Winter et al., 1998). Groundwatersheds also differ from surface watersheds in their ability to fluctuate with time. Whereas surface watersheds are defined by static topography, groundwatersheds are dynamic and their size and shape can change due to pumping, climate change, land use change or seasonality. Therefore, groundwatersheds can be affected by a multitude of natural and human factors. However, in this analysis, we expect the majority of each groundwatershed's spatial extent to be consistent through time as fluctuations in the water table will only correspond to changes in the groundwatershed extent if the locations of water table divides are altered.

Here we derive groundwatersheds using the water table instead of the land surface in a standard watershed delineation algorithm (see Section 7.3 Methods). Using the water table to derive groundwatersheds enables a computationally simple approach to delineate groundwatershed extents. This approach generates groundwatersheds that reflect shallow, local groundwater flow systems (Figure VII.1) but it does not represent nested regional groundwater flow systems that require particle-tracking simulations that are currently infeasible at the global scale.

The groundwatersheds we derive in this study are for the world's **protected areas**. Unlike surface watersheds, we do not use river outlets as outlet locations in our groundwatershed delineation approach. Instead, we use the locations of **GDEs** (Figure 7.2a) that lie inside protected areas (Figure 7.2c) as outlet features. Thus, we do not derive contributing areas of groundwater flow for one location per protected area, but for all GDEs inside each protected area.

We identify groundwater-dependent terrestrial and aquatic ecosystems using datasets of groundwater-dependent wetlands, root zone intersections with the water table, and surface water features such as perennial rivers and streams, since there is no existing global dataset of GDEs (Gleeson et al., 2022). The GDEs we identify in this study do not represent a comprehensive and refined global database but rather indicate locations where these ecosystems can potentially occur.

To evaluate the potential importance of groundwatersheds and analyse their relationship with protected areas globally, we defined two metrics (Figure 7.3c): relative groundwatershed size (RGS) and the underprotected groundwatershed ratio (UGR). RGS is an ecohydrological index representing size of the groundwatershed relative to the size of GDEs inside the groundwatershed. UGR is primarily a socio-hydrological conservation index that represents the underprotected proportion of each groundwatershed.

Groundwatersheds: Contributing areas of shallow local groundwater flow to a feature.

Protected areas: IUCN protected area categories Ia, Ib, II and III (mapped as the 'high protection' group of protected areas in Figure 7.2c).

Underprotected areas: All areas outside of protected areas, as defined above. We use the term underprotected rather than unprotected as other forms of protection and management can exist in these areas, such as the European Union Water Framework Directive or California's Sustainable Groundwater Management Act.

GDEs: Terrestrial or aquatic ecosystems that contain species or habitats that rely on groundwater (Gleeson et al., 2022). Lotic GDEs have running water (for example, rivers and streams), whereas lentic GDEs refer to those with standing waters (for example, lakes and wetlands).

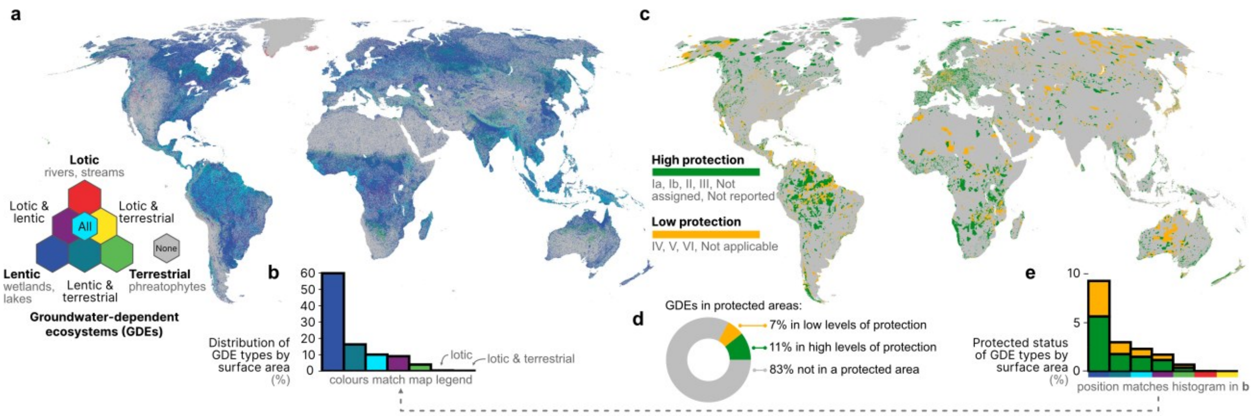


Figure 7.2. GDEs and protected areas.

(a) Location and co-occurrence of GDE types per grid cell. (b) Area distribution of GDE types. (c) Spatial distribution of protected areas. In this study, we map the groundwatersheds of the 'high protection' group of protected areas and not for the 'low protection' group. (d) Area distribution of all GDEs across protected area groups. (e) Area distribution of protection groups across each GDE type. Note that the vertical axis is the same across (b) and (e) (for example, ~6% of all GDEs are lentic with high levels of protection).

7.3 Methods

Note: section order has been rearranged from the manuscript version of this chapter so that the methods section is located before the results.

We implemented a simple geospatial methodology using best-available, openly accessible global data (Table VII.1) to map the groundwatersheds of the world's protected areas. A flow chart of our methodology is shown in Figure VII.7. All analyses in this study were performed at the spatial resolution of 30 arcsecond (~1 km grids at the equator), matching the resolution of the water table data used in this study. See Section VII.5.3 for a discussion on the implications of using this spatial resolution.

7.3.1 A computationally simple approach to groundwatershed mapping

Groundwatersheds were derived by making minor modifications to the D8 surface watershed delineation method (O'Callaghan & Mark, 1984). Whereas surface watersheds are derived using an outlet location (or 'pour point') and a digital elevation model of the land surface, groundwatersheds are derived using potentially multiple outlet locations and the water table instead of the land surface. Whereas a surface watershed identifies the contributing area of overland flow to a specified outlet, a groundwatershed identifies the contributing area of local (shallow) groundwater flow to groundwater-connected features. In this study, these features are GDEs inside protected areas.

Using this water table-driven D8 flow direction algorithm to derive groundwatersheds does not enable representation of nested and deeper regional groundwater flow systems. For an extended discussion on our approach and its limitations, see Appendix VII Sections 1 and 2. In the following sections, we summarize our methods to identify GDEs and protected areas, which are combined to derive groundwatershed outlet locations.

7.3.2 Water table

Our analysis is based on a global depth to water table dataset (Fan et al., 2017). This dataset contains mean monthly water table depths averaged over a 2004–2013 model run. To generate a mean annual water table depth layer, we calculated the weighted mean of the monthly mean water table depths using the number of days in each month as weights. As we required water table elevations to derive flow direction, we converted water table depth to water table elevation by subtracting water table depth from the land surface elevation. We used mean monthly water table elevations in our derivation of GDEs and in our groundwatershed uncertainty analysis, and we used the mean annual water table elevation in our core groundwatershed delineation.

7.3.3 Groundwater-dependent ecosystems

Although we mapped the groundwatersheds of the world's protected areas, we did not map groundwatersheds using the entire extent of protected areas as outlet features. Rather, we identified and used areas inside the protected areas where there are likely GDEs. To identify areas with likely GDEs, we considered ecosystems reliant on surficial expressions (for example, wetlands and rivers) and subsurface expressions (for example, phreatophytes) of groundwater, but not subterranean (for example, hyporheic or karst ecosystems) (Eamus et al., 2006). GDEs were mapped globally using an inference-based approach based on the following: (1) the interaction between rooting depths and the depth to the water table (terrestrial GDEs), (2) the presence of groundwater-dependent wetlands (lentic aquatic GDEs) and (3) surface water systems interconnected with groundwater (lotic aquatic GDEs) systems. Together, these interactions connect groundwater to terrestrial and aquatic ecosystems and are represented by available global data. Our process to identify these interactions is summarized in the numbered paragraphs below.

(1) To identify likely terrestrial GDEs, we considered the relationship between rooting depth and depth to the water table. We identified grid cells where root systems are probably sourcing groundwater by comparing mean monthly depths to the water table with the depth of the root zone (Fan et al., 2017). Any grid cell in which the root zone intersects the water table for at least one month per year is identified as a terrestrial GDE.

(2) To identify likely aquatic GDEs, we considered multiple forms of groundwater-surface water interactions and classified aquatic GDEs as either lotic or lentic systems. To identify lentic systems, we used existing maps of groundwater-dependent wetlands (Tootchi et al., 2019) and lake extents (Messager et al., 2016).

(3) To identify lotic (riverine) aquatic GDEs, we used a network of perennial rivers (Messager et al., 2021). Although not all rivers and surface water bodies depend on groundwater discharge, global data availability did not permit the consideration of hydrologically disconnected surface water bodies. However, our use of only perennial river reaches minimizes this impact. We also did not remove losing river and stream reaches as surrounding water table levels regulate the hydraulic gradient across groundwater-surface water interactions. Losing stream reaches are reflected in our analysis by lower surrounding water table levels and thus do not receive an associated contributing groundwatershed beyond the GDE grid cell(s). In particular, intermittent rivers with a seasonal connection between the groundwater and surface water system (that is, gaining during the wet season, losing during the dry season) (Shanafield et al., 2021) may be sensitive to changes in seasonal groundwater levels, but may have been missed in our analysis that focuses on mean annual conditions.

We then combined these three GDE types (terrestrial, lotic and lentic) into a single GDE map. Among these GDEs, those that are located inside protected areas (see below) were used as outlet features in our groundwatershed delineation.

7.3.4 Protected areas

From the World Database on Protected Areas (UNEP-WCMC & IUCN, 2021), we created two groups of IUCN terrestrial protected areas: those with high levels of protection that restrict human activity inside their boundaries, and those with lower levels of protection that are more permissive of human activity. The protected area classes we considered as highly protective are: Ia (Strict

Nature Reserve), Ib (Wilderness Area), II (National Park), III (National Monument or Feature), as well as protected areas with 'Not Reported' or 'Not Assigned' categories. We included 'Not Reported' and 'Not Assigned' protected areas in this high protection grouping as we found these categories to be more prevalent in countries with lower levels of development where reporting of protected areas may be less comprehensive. By including these categories, we retained a greater global coverage across the protected areas dataset. The remaining protected area categories, IV (Habitat/Species Management Area), V (Protected Landscape/Seascape), VI (Protected area with sustainable use of natural resources) and 'Not Applicable', were grouped together and were considered as having lower levels of protection.

We rasterized both sets of protected areas to our operating resolution, including all grid cells touching a protected area. As the spatial resolution of our analysis is 30 arcsecond (~1 km), we filtered out any protected areas with a reported surface area less than 1 km^2 before rasterization. As protected areas can overlap or border one another, we subsequently identified all spatially contiguous protected areas when representing the protected areas in a binary map at 30 arcsecond, and when applying an 8-connectedness criterion for spatial contiguity.

This set of spatially contiguous protected areas is the protected area set we used as the basis for all calculated metrics (that is, the RGS and the UGR) and to report summary statistics. Using this spatially contiguous but flattened representation of protected areas enabled a more streamlined approach to handle and report global protected area results. These contiguous protected areas differ in total count from the original protected area dataset from which they are derived (Table VII.2).

7.3.5 Groundwatershed delineation

Our groundwatershed delineation process followed conventional watershed delineation approaches that generate a flow direction raster which is used to derive watersheds for specified features. We did not apply hydrological preconditioning steps to the water table surface, including the removal of depressions as depressions in the water table represent local water table gradients which we sought to represent in our study. The flow direction raster was generated using the D8 flow direction algorithm, which can represent 8 possible flow directions to adjacent cells according to the direction of the steepest water table gradient. Although the D8 algorithm has known limitations, such as generating parallel flow paths and poorly depicting watersheds in coastal and endorheic basins for example (Wilson et al., 2007; Rahman et al., 2010), it remains a common, simple, deterministic and widely used approach to derive flow direction. Improving the sophistication of our flow direction derivation may be unwarranted as our analysis was performed at a spatial resolution (30 arcsecond) that is much coarser than conventional watershed-specific delineation studies.

Once the flow direction raster was generated, groundwatersheds were delineated for each GDE cell inside a protected area. Subsequently, groundwatersheds for individual GDE grid cells were grouped across all GDEs found in each contiguous protected area. To avoid double-counting of groundwatershed area, we assigned a single groundwatershed per protected area even if groundwatershed extents may overlap between protected areas. This is possible when a protected area is found inside the groundwatershed of another protected area (Figure VII.7). In these cases, the groundwatershed area for the nested protected area is assigned to this protected area, while the remaining groundwatershed area is assigned as the groundwatershed for the downgradient protected area. For a discussion on how this methodological decision affects our calculated summary metrics (RGS and UGR), see Appendix VII Section 1.3. To generate flow direction rasters and delineate groundwatersheds, we used the 'D8Pointer' and 'Watershed' functions in the Hydrological Analysis toolbox of the open-source geospatial platform Whitebox Geospatial (Lindsay, 2016).

7.3.6 RGS and UGR metrics

Once groundwatersheds were delineated for each contiguous protected area, we subsequently evaluated the two metrics (RGS and UGR) developed to understand regional patterns in groundwatersheds. RGS was calculated by dividing the surface area of each groundwatershed by the surface area of GDEs inside the groundwatershed. Importantly, we also considered GDE surface area that exists outside of the protected area but is inside the groundwatershed as stopping at the protected area boundary introduces a social influence on the ecohydrological metric. The UGR was calculated by dividing the surface area of the groundwatershed that lies outside of the protected area by the total surface area of the groundwatershed. These metrics are summarized in Table VII.3.

7.3.7 Uncertainty analysis

As groundwatersheds are dynamic (that is, fluctuate with the water table), we performed an uncertainty analysis to quantify how groundwatershed extents change throughout a typical year.

To accomplish this, we repeated our groundwatershed delineation process for mean monthly water table depths and evaluated the variability in total groundwatershed size throughout a year. The results of this uncertainty analysis are included in Section VII.4 and Figures VII.8 and VII.9.

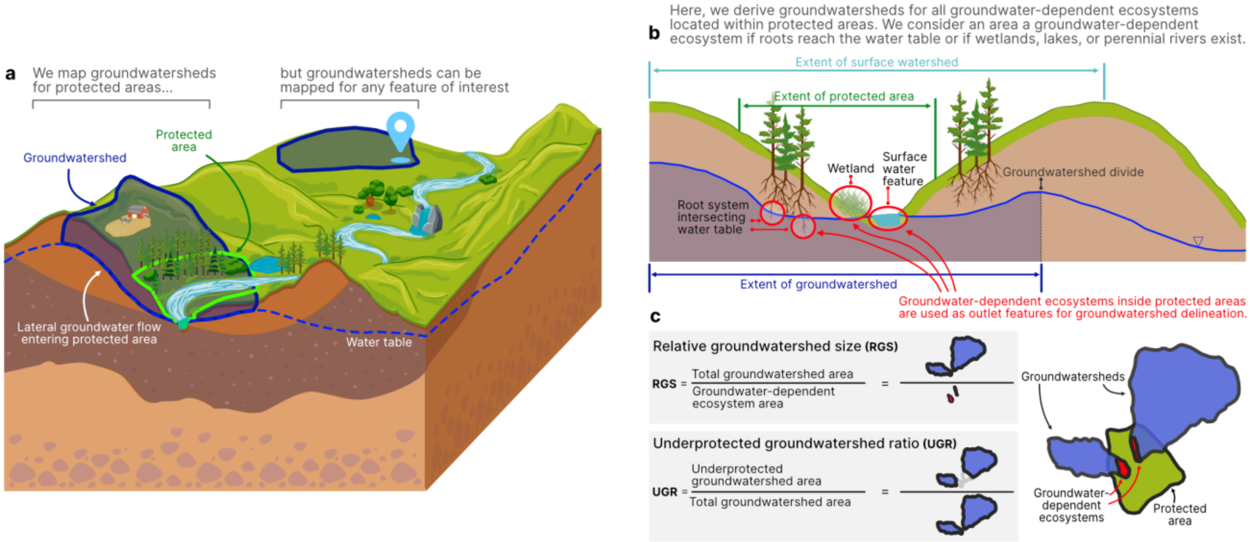


Figure 7.3. Overview of groundwatersheds and our application of groundwatersheds in this study.

(a) Conceptual model of groundwatersheds. (b) Mapping the groundwatershed of a protected area, shown for the cross-section of (a). (c) Metrics used to study patterns in groundwatersheds.

7.4 Results

7.4.1 Groundwatersheds are often larger than their protected areas

The combined size of the groundwatersheds of the world's protected areas is 75% larger (22.0 million km^2, n = 32,490 groundwatersheds) than the combined size of the protected areas that we analysed (12.6 million km^2). Most groundwatersheds (85%; 27,651 of 32,490) extend beyond their protected area boundary. Larger protected areas generally have larger groundwatersheds (Figure VII.2). Thus, patterns in total groundwatershed size largely reflect patterns in protected area size. Groundwatersheds also span international borders and raise transboundary management concerns. We find that 1,379 groundwatersheds cross international borders, 454 of which do so despite their associated protected area existing entirely inside a single country.

The median RGS is 1.46 (Figure 7.4a), with an interquartile range of 1.17–1.94. Overall, RGS tends to be larger in arid regions (Figure 7.4b), which means that the size of the area contributing groundwater flow is greater in proportion to the size of the GDEs they are connected to in arid regions compared with more humid regions. These larger RGS values in arid regions are consistent with previous modelling of the impact of aridity on regional groundwater flow (Gleeson & Manning, 2008; Liu et al., 2020). Larger RGS values in arid regions (for example, Figure 7.4e) suggest that nested and regional flow paths, which are not represented in our water table-based approach, are particularly important in these settings (Section VII.1). Lower RGS values, as found in the boreal forest of central North America (for example, Figure 7.4c), correspond to groundwatersheds where vegetation is highly connected to shallower water tables. In these humid regions, convergence patterns in the water table are more localized and lead to smaller, shallow flow systems.

Trends in RGS do not differentiate a vulnerability gradient in protected areas but rather provide insights regarding the hydrogeological systems the protected areas depend on, and also provide context to inform protection strategies. That groundwatersheds exist outside of protected areas may appear as an intuitive finding, given that protected areas are rarely established on the basis of hydrological system boundaries or processes. Yet the global prevalence of this misalignment between protected areas and their groundwatersheds necessitates that these ecologically important areas of contributing groundwater flow be considered in conservation priority-setting, and RGS is a metric to help inform these efforts.

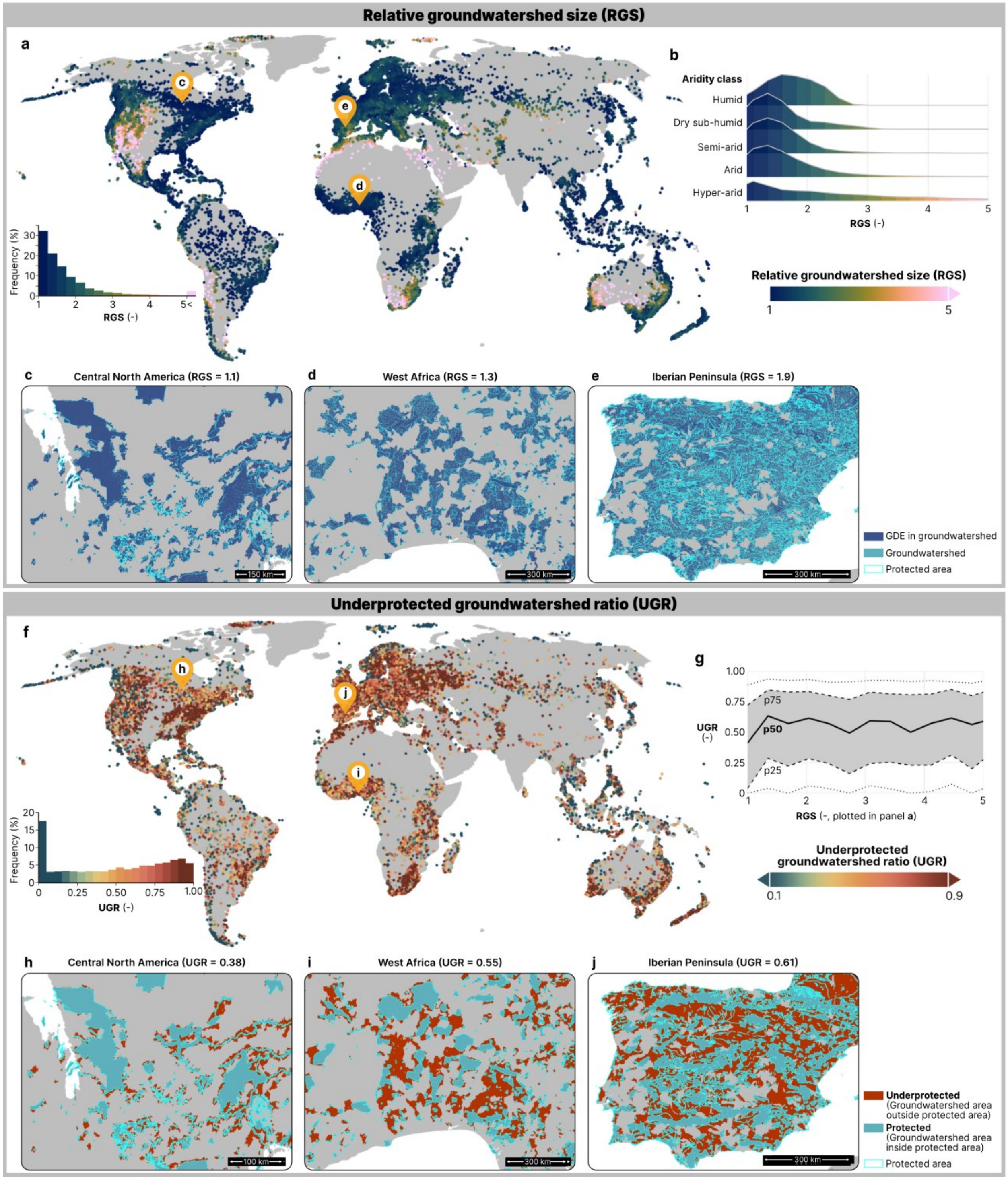


Figure 7.4. Mapping the groundwatersheds of the world's protected areas.

(a–e), RGS. (a), RGS of protected areas, plotted as points at the centroid of each protected area. (b), Distribution of RGS across aridity classes (Zomer et al., 2022) (c–e), Extent of groundwatersheds and GDEs inside groundwatersheds, which are the two inputs used to calculate RGS, shown for (c) central North America, (d) central West Africa and (e) the Iberian Peninsula. (f–j), UGR. (f), UGR of protected areas.

(g), Relationship between RGS and UGR. (h–j), Extent of underprotected groundwatershed area and protected groundwatershed area, which are the inputs used to calculate UGR, shown for (h) central North America, (i) central West Africa and (j) the Iberian Peninsula. (c–e) and (h–j) show the same groundwatershed extents: (c–e) show the distribution and density of GDEs inside groundwatersheds, while panels (h–j) show the distribution of protected and underprotected groundwatershed areas.

For instance, larger RGS generally implies that there is a larger area of contributing groundwater flow to manage. Further, larger groundwater flow systems generally have longer system response and residence times (Cuthbert et al., 2019), meaning that human impacts in larger groundwatersheds may potentially have longer legacy impacts on GDEs than in smaller groundwatersheds. Conversely, protected areas with smaller RGS are typically in regions with a greater density of GDEs. Generally smaller groundwatershed sizes in these humid regions mean that human impacts in these groundwatersheds may more rapidly affect GDEs.

The median UGR is 0.52 (Figure 7.4f), with an interquartile range of 0.17–0.80. This means that the median protected area's groundwatershed extent exists 52% outside of the protected area boundary. There are no regional trends visible in UGR; however, we do find that larger protected areas generally have lower UGRs in comparison with smaller protected areas (Figure VII.2), meaning that larger protected areas tend to have more protected groundwatersheds.

UGR does not vary substantially with RGS (Figure 7.4g), nor with national levels of land protection (Figure VII.3). Even in countries where international area-based conservation targets have been met, groundwatersheds of protected areas remain underprotected by conventional protected area initiatives. Combined, these findings reveal a global misalignment between protected areas and their connected groundwater flow systems and underscore the challenge of conserving protected area ecosystems above and below ground without consideration of groundwatersheds.

7.4.2 Human activity in groundwatersheds may undermine protection

GDEs inside protected areas could be affected directly by external groundwater pumping and contamination, and indirectly via climate change or land use change through their impact on groundwater recharge (Burns et al., 2017; Brown et al., 2011; Zipper et al., 2022). Activities such as mining, agriculture and urban expansion are key determinants of the potential risk to the quality and quantity of groundwater flow to protected areas. Thus, human activity and land modification in underprotected portions of groundwatersheds represent a potential vulnerability for GDEs in protected areas (Figure 7.5a). The timing and severity of these impacts are a function of the type, location and magnitude of the activities in conjunction with the local hydrogeological setting. Assessing this timing and severity of impacts is beyond the scope of this study but could enable improved management as we describe below.

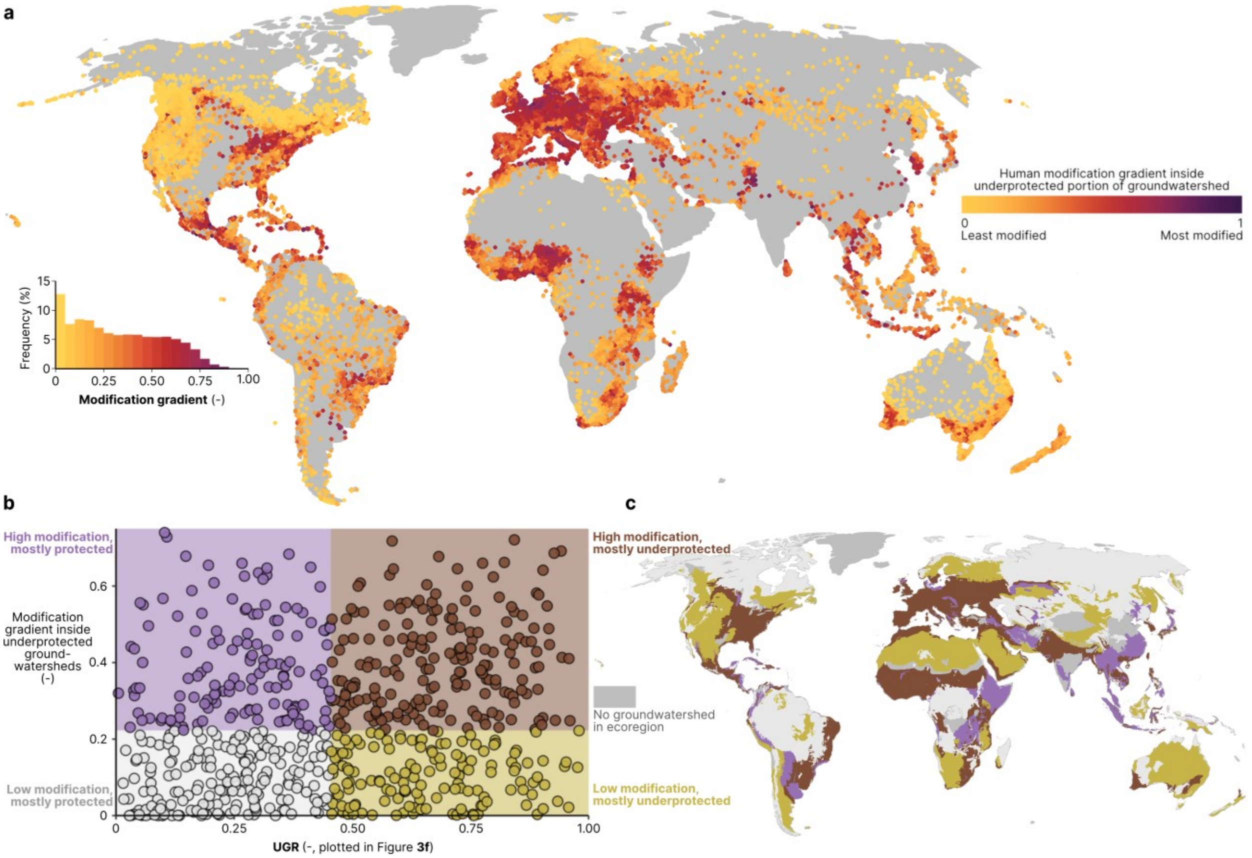


Figure 7.5. Implications of groundwatersheds for conservation initiatives.

(a) The human modification gradient of terrestrial lands inside the underprotected portions of protected area groundwatersheds. Protected areas with no underprotected groundwatershed area are not shown (15% of protected areas). (b,c) The relationship between UGR and the human modification gradient inside underprotected portions of groundwatersheds, summarized per terrestrial ecoregion (Olson, 2001) (n = 679 ecoregions that contain groundwatersheds mapped in this study). (b) A scatterplot of UGR and human modification gradient is split into four quadrants on the basis of the median ecoregion value of each axis dimension. The colour scheme in the scatterplot is also the map legend for (c). (c), The accompanying ecoregion map corresponding to the four quadrants of (b).

Comparing patterns in human land modification (Kennedy et al., 2019) with UGR provides insight on the vulnerability of protected areas (Figure 7.5b,c). To assess regional patterns, we summarized the level of land modification and UGR per terrestrial ecoregion. Ecoregions where we find high human modification levels and underprotected groundwatersheds are probably areas where GDEs inside protected areas are most vulnerable to potential impacts through groundwater flow. These ecoregions are scattered across the world and include: the USA's Midwest to east coast, Central America, coastal Brazil, the majority of Europe, northern Africa, the Sahel and Sudanian savanna in sub-Saharan Africa, Iran, Pakistan and Northern India.

We have contained our analysis to higher levels of protection (International Union for Conservation of Nature (IUCN) management categories I–III). However, we found most groundwatersheds to remain underprotected even when expanding our analysis to include all IUCN management categories (categories I–VI). The median national percentage of underprotected groundwatershed surface area that is in fact protected by lower levels of protection (categories IV–VI) is only 5%. However, Germany, Guatemala, Morocco, Myanmar, South Korea and Uruguay are among the few nations whose lower levels of protected areas cover substantial portions (all over 30%) of their underprotected groundwatershed area identified in this analysis (Figure VII.4).

7.5 Discussion

Here we have used groundwatersheds to reveal the global potential for distant and long-term subsurface impacts on GDEs inside protected areas. Yet, our mapping of groundwatersheds for protected areas is only one of many possible applications and this work can serve as a proof-of-concept for wider use. Groundwatersheds, such as surface watersheds, can be identified for any feature, such as a protected area, groundwater well, wetland or community. Groundwatersheds have a strong potential to inform a range of decisions and management approaches for sustainability planning and resilience-building, especially when used in conjunction with surface watershed approaches.

As countries work towards expanding and strengthening their systems of protected and conserved areas, in alignment with Target 3 (the 30×30 target) of the Kunming-Montreal Global Biodiversity Framework, there are several pathways that could reduce groundwater-mediated threats to existing and new protected areas (as well as to designated OECMs, which equally contribute to the 30×30 target). First, formal area-based protections could be expanded to cover critical groundwatersheds, especially in conjunction with the addition of new protected areas. Extending the boundaries of existing protected areas to encompass their groundwatersheds may be more challenging, as protected areas are often surrounded by land uses that would limit protected area expansion and there is very little land that can be considered free of trade-offs (Meyfroidt et al., 2022). The expansion of formal protected areas and OECMs may in fact be unnecessary if activities in groundwatersheds can be managed through means such as groundwater permitting or other restrictions (Higgins et al., 2021). For any of these pathways, additional information about the timescale, magnitude and distribution of impacts on protected areas and people is necessary.

The variability of potential human impacts and the social, economic and political differences across regions imply that a diverse portfolio of approaches is necessary to protect groundwatershed water quality and quantity. Enhanced conservation and management of groundwatersheds could be achieved through adoption or expansion of strategies such as groundwater regulation (for example, well permitting), sustainable water policies (for example, the Sustainable Groundwater Management Act in California, USA), source water protection (for example, Edwards aquifer protection in Texas, USA), hydrological protection zones for wetlands (Schouten, 2022), Indigenous-led land and water management and monitoring (for example, guardian programmes such as in northwestern Australia), conservation or regenerative agriculture (for example, practices that reduce groundwater pumping) and nature-based solutions (for example, invasive species removal for the Greater Cape Town Water Fund in South Africa). Management strategies could be borrowed or adapted from these and other conservation and source water protection approaches, rather than developing entirely new policy or management approaches. Selecting an appropriate strategy depends on the social, economic and political context, as well as on the type of possible impact, from severe (nearby, large magnitude pumping or contamination) to less impactful (distant or minor land use change).

While we have mapped groundwatersheds of protected areas to place greater focus on groundwater in conservation initiatives, it is important to note that many protected areas also have surface watersheds extending beyond their boundaries. Directly comparing groundwatersheds with surface watersheds is non-trivial as important differences exist in the conceptualization and analysis of these two different types of watersheds, and a detailed comparison is beyond our scope. In this study, we included groundwater-dependent wetlands and root zone intersections with the water table to derive outlet features for groundwatersheds, but these features are not typical outlets for surface watersheds. Furthermore, as surface watersheds are nested and hierarchical, their delineation also hinges on the spatial scale of study. For example, the surface watershed for Mangroves National Park in the Democratic Republic of Congo (located at the outlet of the Congo River) could range from a localized sub-basin to the entire Congo Basin, depending on the scale of analysis. Yet, it holds that for effective conservation, approaches must consider both contributing areas of groundwater and surface water flow that extend beyond protected area boundaries and the human impacts on these systems. For more discussion on comparing groundwatersheds to surface watersheds for protected areas, see Appendix VII Section 1.2 and Figure VII.5.

Our results importantly highlight the connection between groundwater and protected areas and reveal the vulnerability of protected areas to potential groundwater impacts. However, our approach has limitations (see Section VII.5). For instance, we used a simplified approach to identify potential GDEs, focused on higher levels of protection, and mapped only the spatial extent but not the timing of human impacts acting on protected areas through groundwater flow. This first-order global analysis is not intended to lead to recommendations for specific protected areas but rather identifies regional trends in these relationships and discusses potential strategies. With more detailed information, our water table-based approach can be applied to smaller, specific areas. Alternatively, numerical models including particle-tracking approaches that are computationally feasible at local scales can provide more detailed information about the full hydrogeological system and can produce critical insights when combined with the groundwatershed concept and motivation introduced here. As governments around the world commit to new protected area targets and other actors make their own conservation commitments, our analysis serves as a reminder that protection neither stops at protected area borders, nor at the ground surface.

7.6 Code and data availability

This study uses data from multiple open-access datasets. Source data are documented in Table VII.1 and can be downloaded from the persistent web-links provided. Data produced in this study, including the GDE map, groundwatershed extents and protected area metrics have been deposited on Borealis, the Canadian Dataverse Repository (see below).

Environment	R project for statistical computing (R Core Team, 2021)
R packages	
terra	Hijmans (2021) https://cran.r-project.org/package=terra
gdalUtilities	O’Brien (2022) https://cran.r-project.org/package=gdalUtilities
rasterDT	O’Brien (2022) https://cran.r-project.org/package=rasterDT
whitebox	Wu & Brown (2022) https://cran.r-project.org/package=whitebox
rnaturalearth	Massicotte et al. (2023) https://cran.r-project.org/package=rnaturalearth
ggplot2	Wickham et al. (2024) https://cran.r-project.org/package=ggplot2
tmap	Tennekes et al. (2023) https://cran.r-project.org/package=tmap
MetBrewer	Mills (2022) https://cran.r-project.org/package=MetBrewer
scico	Crameri et al. (2020); Crameri et al. (2021); Pedersen (2020) https://cran.r-project.org/package=scico
Script repository	https://github.com/XanderHuggins/groundwatersheds-for-PAs Archived with data repository (see below)
Data repository	Borealis: https://doi.org/10.5683/SP3/P3OU3A

7.7 Acknowledgements

Figure 7.3a was produced by the authors with support from L. Bueno and S. Lopez by adapting vector graphics by artists ‘brgxfx’ and ‘freepik’, accessed on the graphic repository Freepik (www.freepik.com); artists T. Saxby, J. Hawkey and J. C. Fisher, accessed on the Integration and Application Network (ian.umces.edu/media-library) under CC BY-SA 4.0; and artist A. Coquet, accessed on The Noun Project (thenounproject.com) under CC BY-NC-ND 2.0. Composite figures were assembled in Affinity Designer (https://affinity.serif.com/en-us/designer/).

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Chapter 8

CONCLUSIONS & REFLECTIONS

8.1 Opening reflections

I began by situating this dissertation at the intersection of two crises: the global groundwater crisis (a crisis in the resource) and the provocation of groundwater science as a “zombie science” (a crisis in the science). Both are, and will continue to be, grand challenges. Thus, my intention has not been to approach these challenges as if identifying ‘solutions’ were plausible outcomes but rather to develop a body of research that is enriching and relevant to the continued ‘living’ with these grand challenges and the questions they pose. With this view, I have not sought to pursue traditional groundwater research outcomes (for instance, to constrain flux uncertainty or improve model performance) but to revisit fundamental traditions and research directions in groundwater science. Doing so represents my attempt to lead research that is responsive to these intersecting crises.

One reading of this dissertation, as a collective body of work, is that it represents a waymark of progress regarding the growing interdisciplinarity of groundwater science (Barthel & Seidl, 2017). More simply, this dissertation actively moves the tense of interdisciplinary groundwater science from being a “*future* of hydrogeology” (Gleeson & Cardiff, 2013) to its *present*. Since concepts and methods used herein span groundwater science, data science, sustainability science, and civil engineering, this dissertation can be understood as a fundamentally interdisciplinary work. Indeed, the focus on developing a relation-centered understanding of groundwater through the groundwater-connected systems framing has been equally motivated by natural science research on groundwater system interactions with the physical environment as it has by complexity science principles (“systems are constituted not only by parts and mechanical kinds of interactions, but that come about as a result of relations and organizational processes that constitute [their] interactions”; Preiser et al., 2021) and as it has by social science perspectives of, for instance, hydrolectics (“in every case, it is the relation that defines the essence of what water is”; Linton, 2010). As I have conducted this work without a need to call individual disciplines into use by name or by obligation, but rather drew from individual perspectives, tools, and methods based on problem-based needs, it is possible to also consider this dissertation as undisciplinary (Haider et al., 2018).

In all, I have worked to develop one possible approach to conduct groundwater sustainability science, which is found in the groundwater-connected systems framing and its application. The application of this framing, as I have both argued and demonstrated, can generate insights that are relevant for both groundwater science and groundwater sustainability needs. The contributions I make span both overarching conceptual and methodological advances to research fields, and specific, empirical insights generated through investigation of individual research questions. In the remainder of this chapter, I summarize these contributions, discuss the central role of conceptual modelling in this work, identify future research pathways that address limitations of the research, and close with reflections on the research process.

8.2 Contributions of the dissertation

I have built an argument that a meaningful and generative line of inquiry can be found by explicitly focusing groundwater research on groundwater system interactions rather than on groundwater as an isolated resource. I have demonstrated that doing so can generate novel insights on groundwater dynamics in the Anthropocene where human actions increasingly dominate biophysical systems and can better align groundwater science with sustainability needs. Simply making a call to place greater attention on socioeconomic, ecological, or Earth system interactions with groundwater is not the contribution of this work as I draw on a rich literature that already does so (Cantonati et al., 2020; Castilla-Rho et al., 2020; Gleeson et al., 2020; Re, 2015; Zellner, 2008; Zwarteveen et al., 2021). Rather, I have sought to develop and apply a conceptual framing to guide groundwater research towards these objectives in a way that integrates and balances the representation of the various socioeconomic, ecological, and Earth systems that are dynamically connected with groundwater and with the goal of identifying social-ecological system and complex adaptive system properties and outcomes.

The contributions I make emerge from two distinct but interlinked forms of novelty: novelty in the research process, and novelty in research outcome. I find these forms of novelty to be useful to separate as they serve unique purposes and audiences. My perception is that research process contributions can have broader impacts by reaching wider audiences and enabling a variety of applications, whereas research outcome contributions investigate hypotheses and lines of inquiry in groundwater sustainability science and deepen this specific literature.

Given the interdisciplinary nature of this work, my contributions to research process will depend on the training and community of the reader. Core themes of this dissertation include social-ecological systems, interdisciplinary data science, large-scale groundwater systems, data-driven applications of sustainability frameworks, and problem framing (Figure 8.1a). For instance, large-scale investigations of groundwater systems are commonplace to the groundwater scientist but may represent new topic matter to the sustainability scientist. Similarly, interdisciplinary problem framing is a core competency of the sustainability scientist, but it has only recently been recognized as a relevant skill for the civil engineer (Engineers Canada, 2016). Thus, the core themes of this dissertation can be perceived as working at and *widening* these margins of groundwater science, sustainability science, and civil engineering (Figure 8.1b). In this way, the dissertation can serve an enabling purpose by acting to expand the scope and thus the potential of these fields.

The research process contributions made here take three forms: conceptual model development, method development, and method application. I make a distinction here between method development and method application contributions, where I understand method development as contributions that offer fundamentally new methodologies whereas method application contributions are understood as the application of existing methodologies to new topics. Each data-driven research chapter (Chapters 4-7) is based on a unique conceptual model of groundwater systems as social-ecological systems (I reflect specifically on the conceptual modelling process in Section 8.3). These conceptual models are based on research question-specific integration and interpretation of sustainability science principles to the groundwater domain, and thus represent a reference process of interpreting sustainability science for application to new fields and topic matter.

Examples of method development include the derivation of the groundwaterscape risk concept (Paper IV), and the approach to delineate protected area groundwatersheds (Paper VI). The derivation process of groundwaterscape risk, where overlaying clusters of groundwater system functions (i.e., groundwaterscapes) with groundwater system with risk types generates a transferrable methodology to explicitly delineate unique sustainability challenges for other systems. Indeed, this methodology is already being adapted to similarly characterize sustainability challenges of global grazing lands (Piipponen et al., in preparation). Alternatively, the methodology of delineating groundwatersheds by linking groundwater-dependent ecosystem mapping with water table-based watershed delineation provides a workflow that can be performed at other scales and for other research questions (e.g., for administrative jurisdictions rather than for protected areas, as done in Paper VI).

A number of existing methods are applied to novel topics in this dissertation. These include the application of hotspot mapping to river basin social-ecological vulnerability (Paper V), the application of social-ecological archetype analysis to large-scale groundwater systems (Paper III), the application of a self-organizing map archetype analysis method to global geospatial datasets (Paper III and Paper IV), and the application of landscape metrics to understand landscape complexity of groundwater system functions within large aquifer systems (Paper IV).

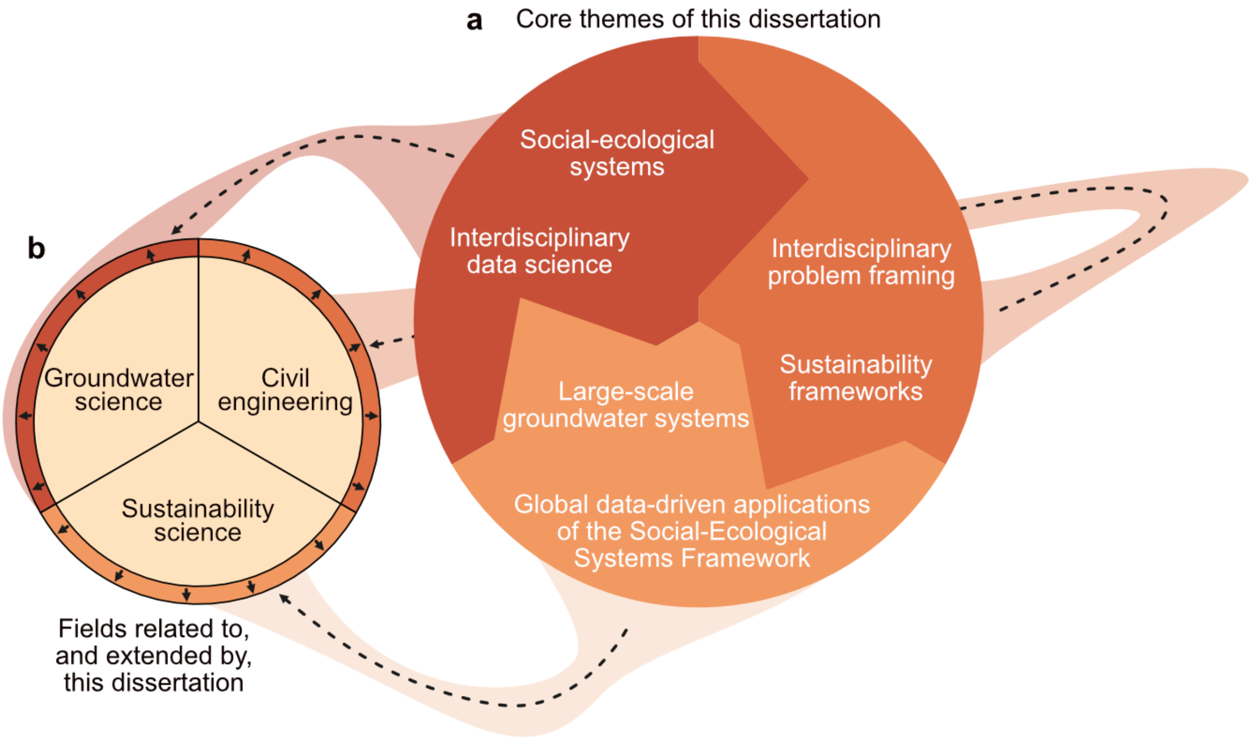

Figure 8.1. Core themes of the dissertation and how they work to extend related research fields.

(a) Core themes of the dissertation are located within segments of the circle colour-coded to the field they extend, as shown in (b). For instance, large-scale groundwater system applications represent an extension to sustainability science, whereas interdisciplinary problem framing represents an extension to civil engineering and social-ecological system analysis represents an extension to groundwater science.

The core research contributions of this dissertation emerge from the central column of research studies (Figure 1.1), and which constitute an 'insight pathway' of: conceptual framing development (Chapter 2, Paper I) ➜ data review (Chapter 3, Paper II) ➜ landscape units corresponding to the conceptual framing (i.e., groundwaterscapes; Chapter 4, Paper III) ➜ sustainability implications of the conceptual framing and the derived landscape units (i.e., groundwaterscape risks, Chapter 5, Paper IV). The extended research chapters (Chapters 6 and 7, Papers V and VI) do not continue along this pathway but demonstrate the wide applicability of the groundwater-connected systems framing. Core results and insights from all research chapters are summarized in Table 8.1.

Below, I will focus on the specific groundwater sustainability science insights produced in the four global-scale studies (Papers III-VI). Owing to the novel and interdisciplinary conceptual models underpinning these analyses, while each study is data-driven, quantitative, and fully reproducible, I understand the overarching insights provided through these works as principally qualitative.

For instance, in Chapter 4, I conduct a global mapping of groundwaterscapes which produces a novel set of 15 landscape units and provides intrinsic insights on the dominant co-occurrences of groundwater's social-ecological functions that characterize the groundwaterscapes. However, in my reading, the more significant contribution is found in the groundwaterscape concept itself, which could be adapted and derived at other scales or using alternative conceptual models. Furthermore, the qualitative findings that groundwaterscapes are heterogeneously distributed across the world's large aquifer systems and that groundwaterscapes are unevenly monitored by the global network of monitoring wells are possibly more impactful insights than their quantitative underpinnings.

This interpretation of research outcomes may reflect the nascent state of the groundwater sustainability science I conduct in this dissertation. Drawing on Shneider's (2009) proposed stages of a scientific discipline, the work presented here can be understood as first stage (introduction of new language and objects as subject matter) and second stage (development of methods and techniques) science. This dissertation has also largely applied inductive (rather than deductive) and generative (rather than verificative) analytical strategies. Inductive strategies build theoretical categories from relationships discovered among data, whereas deductive strategies seek to match concepts of a theory to data (Bourgeron et al., 2018). Generative analysis, which is often inductive, attempts to discover theory through data exploration, whereas verificative analysis seeks to establish the boundaries for which a theory applies (Bourgeron et al., 2018). As each global-scale study I conduct develops fundamentally novel global analyses and insights, they must take on an inductive form (i.e., there are no estimates to refine, contest, bound, or verify).

In Paper III, I derive groundwaterscapes, which represent a significant contribution to global groundwater science through the development of a novel landscape unit for use in scientific, management, data collection, and educational contexts. The 15 groundwaterscapes offer fundamental insights on groundwater systems as they reveal the spatial co-occurrence and underlying patterns across groundwater's large-scale (order of 10^4 km^2) Earth system, ecosystem, food system, and water management system functions for the first time at the global scale. Groundwaterscapes can be used as a spatial template for testing hypotheses on behavioural differences across groundwaterscapes, for improved management by tailoring management goals to specific function configurations, and to motivate additional and equitable monitoring capacity across groundwaterscapes. As multiple groundwaterscapes are found within all major aquifer systems of the world, the concept can act as a tool to embed social-ecological context in global groundwater science and underscores the information loss that occurs when aquifers are treated as lumped systems in global assessments. Furthermore, groundwaterscapes can support targeted data collection to remedy observation imbalances in existing global monitoring capacity.

In Paper IV, I directly build on the groundwaterscape mapping and investigate the groundwater sustainability implications of the groundwater-connected systems framing. I accomplish this through a complementary global classification of groundwater system risk. Groundwater system risks are identified through an Anthropocene risk framing, representing the first application of this risk concept to global groundwater sustainability challenges. The derived groundwater system risk types are overlaid and compared with the groundwaterscapes to identify a global set of groundwaterscape risks, which represent spatially explicit maps of unique configurations of groundwater system functions and groundwater system risks. This study thus represents the most comprehensive approach to date to map the breadth of challenges that relate to the global groundwater crisis and offers a powerful framework to define problem spaces with potential applications across a range of human-environmental systems. The groundwaterscape risk concept can be an effective tool to re-frame (*sensu* Jerneck & Olsson, 2011) the global groundwater crisis and address sustainability impasses by prompting consideration of the alternative and multiple forms of groundwater sustainability challenges. Groundwaterscape risks also offer a new conceptual and practical tool for groundwater managers to use in developing context-appropriate sustainability strategies, to facilitate solution transfer between regions with similar groundwaterscape risks, and to develop solution networks.

In Paper V, I present the first global-scale social-ecological vulnerability assessment of freshwater stress that explicitly incorporates groundwater storage trends. I do so by building on existing approaches to evaluate river basin resilience (Varis et al., 2019), but refine the scope of this assessment to consider the likelihood to experience impacts from the linked and specific threats of freshwater stress and storage loss. This study also represents the first traceable application of the hotspot concept as defined in conservation biogeography (Whittaker et al., 2005) to global freshwater systems. Doing so advances an understanding of hotspots that are not defined simply by the magnitude of their hydrological state or rate of change (e.g., Rodell et al., 2018) but rather by the likelihood of social-ecological impacts stemming from this hydrological condition.

In Paper VI, the groundwater-connected system framing supported an analysis on the role of groundwater in global area-based conservation initiatives. This study presents the first global-scale inference mapping of groundwater-dependent ecosystems and the first global-scale application of the groundwatershed concept. This study offers a simple methodology to develop first-order insights on the connectivity of ecosystems with groundwater flow systems. By evaluating the protected status of the groundwatersheds of the world's protected areas, this study identifies the globally widespread omission of groundwater considerations from area-based conservation initiatives. We find that omitting groundwater considerations in this way undermines protection efforts as human impacts occurring outside of protected areas can be transmitted to ecosystems within these protected areas through groundwater flow in up to 85% of protected areas worldwide.

These global studies also gesture towards and substantiate the six reasons (see numbered reasons in Chapter 1) for global groundwater science as argued by Gleeson (2020). For instance, the derivation of groundwaterscapes synthesize global studies quantifying interactions between groundwater and other components of the Earth system (reason 5); the derivation of groundwaterscape risks supports the systematic analysis of groundwater sustainability problems in support of solution transfer (reason 2) and facilitates global networks of interdisciplinary experts (reason 6); the identification of social-ecological vulnerability hotspot basins offers systematic analysis to guide regional prioritization (reason 2) and inform water governance in transboundary aquifers (reason 1); and the delineation and assessment of protected area groundwatersheds highlights the importance of groundwater in global sustainability frameworks (reason 4). All analyses create visualizations that improve the understanding and appreciation of groundwater (reason 3).

Table 8.1. Overview of study objectives, methods, results, and insights of this dissertation.

Paper	Objective (➜Conceptual model elements)	Methods	Core result	Insight(s)
I	Develop an approach to study groundwater systems as social-ecological systems. ➜ Social-Ecological System Framework elements: resource systems, resource units, actors, governance systems, ecosystems, political setting (McGinnis & Ostrom, 2014).	Literature synthesis	Development of the **groundwater-connected systems framing.**	Studying groundwater by explicitly focusing on system interactions with social-ecological and Earth systems introduces new methods for groundwater science and better align the science with sustainability needs.
II	Review global datasets relevant for the study of groundwater systems as social-ecological systems, with a focus on physical groundwater system interactions. ➜ Hydrosphere, biosphere, lithosphere, atmosphere, food systems, governance, other socioeconomic systems.	Data review	Collection and review of open-access global datasets (n = 131).	There is a considerable volume of underutilized socioeconomic and biophysical data for global-scale groundwater science and sustainability applications. Most global data that explicitly consider groundwater system interactions are temporally static.
III	Derive landscape units of groundwater-connected systems at the global scale. ➜ Large-scale Earth system, ecosystem, food system, and water management system functions.	Conceptual model Self-organizing maps Landscape metrics	**Groundwaterscapes**: landscape units (n = 15) representing configurations of groundwater system functions.	A social-ecological conceptualization of groundwater is not intractable, and we find a set of broadly occurring configurations of groundwater's social-ecological system functions.
IV	Characterize the global groundwater sustainability problem space through unique, spatially explicit configurations of system functions and risks. ➜ Nine Anthropocene risks consisting of Earth system change and potential for inequality emergence in socioeconomic systems.	Conceptual model Self-organizing maps Anthropocene risk	**Groundwaterscape risks**: spatially explicit and unique configurations of groundwater sustainability challenges, characterized by similar groundwater system functions and groundwater system risks (n = 270).	Mapping groundwaterscape risks provides a spatially explicit characterization of regional patterns in global groundwater sustainability challenges. Groundwaterscape risks can act as a tool to facilitate solution network development and solution transfer between regions facing similar challenges.
V	Evaluate river basin social-ecological vulnerability and identify hotspot basins for global prioritization. ➜ Freshwater stress, freshwater storage loss, ecological sensitivity, social adaptive capacity, IWRM implementation.	Conceptual model Vulnerability analysis Hotspot identification	**Hotspot basins** (n = 168) for global prioritization to minimize social-ecological impacts from freshwater stress and storage loss.	Vulnerability hotspots are spatially constrained to regions facing freshwater stress and storage loss but vulnerability patterns within hotspot basins can differ from patterns in baseline hydrological trends.
VI	Evaluate the spatial extent and protected status of groundwater catchments (groundwatersheds) of the world's protected areas. Determine the magnitude of human activity in underprotected groundwatersheds. ➜ Groundwater flow, area-based conservation, groundwater-dependent ecosystems, human modification of land systems.	Conceptual model Groundwatershed delineation Groundwater-dependent ecosystem mapping	Inference-based map of **groundwater-dependent ecosystems**. **Groundwatersheds** (groundwater catchments) of protected areas (n = 32,490).	Groundwater-dependent ecosystems are abundant in the world's protected areas. The majority of protected areas have groundwatersheds that are underprotected (85%). Groundwater protections are needed for robust area-based conservation.

8.3 Conceptual models

A fundamental goal of the groundwater-connected systems framing is to encourage a revisiting of groundwater system conceptual models and to advocate for their widening through the integration of social-ecological system elements. (While I appreciate distinctions drawn between mental, perceptual, and conceptual models, I do not find it necessary to belabour this distinction here.) Given the inductive nature of this dissertation, conceptual modelling has served a critical role in the development and refinement of the groundwater-connected systems approach. Thus, one reading of this dissertation is that it represents an extended, applied rumination on the conceptual modelling process for groundwater sustainability science. The enduring value of this exercise is illuminated through one of the guiding principles of conceptual modelling: to ‘capture essential features’ of a system rather than to ‘model everything’ (Clark et al., 2011; Sivakumar, 2008). This focus on capturing essential features should not be understated, as a potential trap of the groundwater-connected systems framing is to think it seeks to integrate all possible social-ecological features into any analysis – this is not the case. Rather, it is a framing that seeks to encourage deeper consideration on the inclusion of social-ecological features and on what features are essential for a given research question; and supports the development of multiple models for a given system to better represent diverse and potentially competing perspectives (cf. Ferré, 2018). This is particularly encouraged in cases where models omit these considerations entirely as hydrological processes can be insufficient to capture critical eco-hydrological and socio-hydrological functions, thresholds, or processes (e.g., drawdown “triggers”; Currell, 2016). Yet, the exercise of identifying essential relationships is not trivial, highly normative, and engaging reflexively in such endeavours has long been a weakness of natural scientists (Lélé & Norgaard, 1996). Thus, the groundwater-connected system framing was intentionally designed offer non-prescriptive guidance for conceptual model development and does not seek to establish boundaries for future applications and development of the framing.

The conceptual models developed and applied in this dissertation (Figure 8.2) serve a wide variety of purposes. The process of developing these models has proven useful to guide data collection, identify new and meaningful research questions, challenge the robustness of prevailing theories (for instance, how groundwaterscapes contrast with the boundaries of large aquifer systems), enrich a policy dialogue (for instance, how groundwatersheds identify an overlooked consideration for robust area-based conservation), and reveal the apparently simple to be complex (for instance, how groundwaterscape risks refine and add regional nuance to global groundwater crisis characterizations). All of these contributions correspond to Epstein's (2008) list of reasons other than prediction to build models. Given the conceptual broadening of groundwater systems imparted by taking a groundwater-connected systems approach (which challenges the ability for process-based models to represent these more complex and interdisciplinary systems), I have pragmatically sought to develop first-order global-scale insights through data synthesis rather than process-based model outcomes. Indeed, this dissertation is a testament to the volume and strength of scientific outcomes that can be generated simply through confronting carefully developed conceptual models with global data.

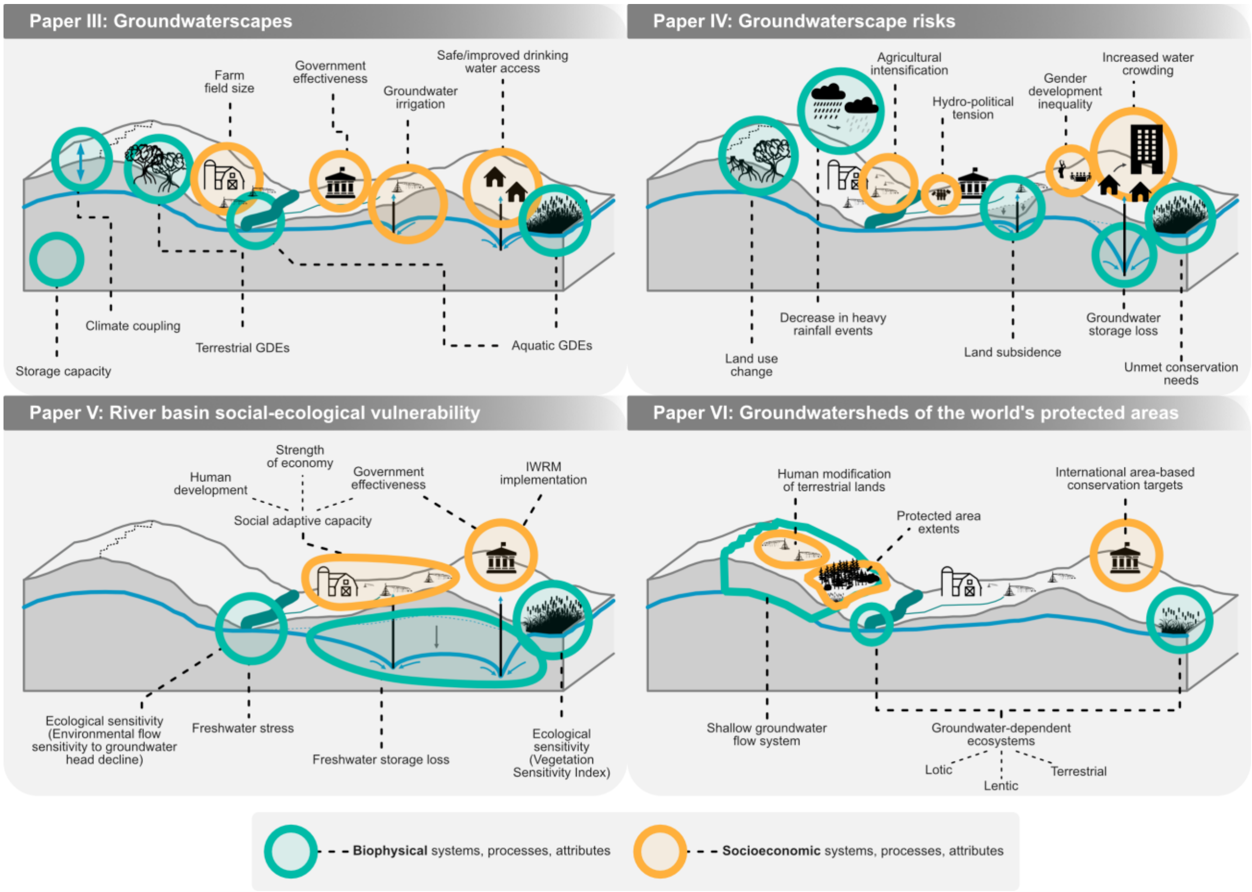

Figure 8.2. The conceptual models of this dissertation.

Simplified conceptual models of all global-scale research chapters (Papers III-VI), using the same landscape schematic. Most studies specify their conceptual model elements beyond this first-order classification as relating to either biophysical or socioeconomic systems, however this scheme is used here for a more straightforward visual comparison between studies.

8.4 Contributions to global groundwater data

This work builds on the rich, diverse, and rapidly growing landscape of global groundwater data. In total, I have incorporated 40 global datasets across the four global-scale research chapters (Figure VIII.1). Thus, this work represents a deep and sustained engagement with global groundwater data. While limitations in the existing data landscape have been identified in Paper II, such as how data explicitly relating to groundwater are mostly temporally static and are nearly exclusively generated within institutions in the global North, I have also found the existing availability of data more than sufficient to reveal substantial new insights for both groundwater science and groundwater sustainability. In this regard, the dissertation owes a tremendous debt of gratitude to the community of scientists and data authors that have built and shared this collection of data.

Yet, my engagement with global data has not been simply extractive as I have generated multiple datasets that now contribute to this collection. The global datasets this dissertation generates include *classifications* of groundwaterscapes (1), groundwater risk types (2), groundwaterscape risks (3), and groundwater-dependent ecosystems (4), *basin attributes* of social-ecological vulnerability (5), and *zonal features* of protected area groundwatersheds (6). Data generated for individual research chapters are deposited on Borealis (https://borealisdata.ca/) in study-specific repositories, as documented in each chapter. In Figure VIII.1, I visualize the relationships between all input and output datasets used and generated in this dissertation.

8.5 Future research pathways to address limitations

The groundwater-connected systems framing, as presented in Chapter 2, establishes wide foundations for future work across groundwater science, data collection, management, and education. The global scale analyses conducted in this dissertation represent only an initial sample of these applications. Instead of attempting to exhaustively overview future directions for the framing, I constrain the discussion here to future work that can address specific limitations in the research presented in this dissertation. For this purpose, I make a distinction between two types of limitations: limitations in framing application and framing implementation. I use the term 'framing application limitations' to represent biases in conceptual model development, and the term 'framing implementation limitations' to represent data and method constraints. Addressing framing application limitations can be accomplished by future studies that broaden and diversify the focus and leadership of groundwater-connected system assessments. Conversely, framing implementation limitations can be addressed through work that improves data availability or fidelity, or resolves specific method constraints. For instance, one major limitation of the work presented here originates from the lack of temporally dynamic global datasets characterising groundwater-connected systems, as identified in the Chapter 3. This reality limits our capacity to understand how social, ecological, and groundwater systems have dynamically interacted and co-evolved historically and doubles as a limitation for making future predictions or projections about these integrated systems. As implementation limitations are discussed in individual research chapters, I dedicate the remained of this section a reflection on the role of bias in this dissertation, which I understand as an important lens through which to understand and redress framing application limitations.

The core forms of bias in this dissertation are found in system understanding, geospatial scale, and method selection. This work has focused on understanding shallow terrestrial groundwater systems as social-ecological systems using geospatial data and data science methods, and has operated exclusively at the global scale. Each of these forms of bias is described below. It is crucial that the science conducted in this dissertation is not understood as the only or definitive approach in terms of scale, set of methods, and system scope for groundwater-connected systems research.

My work here has focused on shallow terrestrial groundwater systems, leaving many environments underrepresented. Future work can conduct similar analyses of groundwaterscapes, groundwaterscape risks, and social-ecological vulnerability on coastal, mountainous, and cold regions. Additionally, while this work has focused on physical groundwater systems, future work can build on this foundation by incorporating groundwater quality and biogeochemical functions into groundwater-connected system conceptual models.

The focus on global scale characterization in the included studies lays global scaffolding for future work to generate similar insights at local, basin, and regional scales. Developing, for instance, the concept of groundwaterscapes within a particular region can generate more place-based insights regarding groundwater system functions and can be informed by regionally specific conceptual models and data. Indeed, one motivation for the global scale approach undertaken throughout this dissertation is to create globally available baseline insights which can be refined at more specific scales where such advances are sought. Cross-scale interactions are underexplored in this dissertation, as the studies are uniformly conducted at the global scale. Developing groundwater-connected system insights at local and regional scales can offer one pathway to understand the scaling of processes and system patterns.

This dissertation has a clear orientation towards spatial data science techniques. While this focus has been effective in generating a wide variety of global scale insights (as described in section 8.2), there are many methods and ways of fact making in social-ecological systems research (Biggs et al., 2021) that have not been applied in this work. For instance, participatory methods aimed at narrative development and scenario building (e.g., Carpenter et al., 2015), qualitative network models (e.g., Dambacher et al., 2003), and agent-based models (e.g., Castilla-Rho et al., 2017) all represent various approaches outside the scope of traditional hydrological research methods that can be applied to better understand groundwater-connected systems across local to global scales.

Returning to some of my opening reflections (Chapter 1) on the need to acknowledge the role of the scientist in the scientific process (a basic insight in the social sciences but one that is undervalued and overlooked in traditional engineering education), many of these biases can be understood or explained by the social location from which this dissertation was developed. I, as a white settler living in Canada, conducted this dissertation within a civil engineering department and in a research group that typically focuses on groundwater science and sustainability at the global scale (http://www.groundwaterscienceandsustainability.org/). My immediate research community is predominantly located in the global North, with extended research stays during this work in Sweden and Austria. My intention in identifying this location is to recognize its profound impact on myriad decisions and outcomes of my research process. For example, decisions about what research questions to pursue, which ideas to engage with (e.g., Western science or other knowledge systems), and my understanding of the research community that I wish to participate in through my work have been and continue to be shaped by my social location. Therefore, to develop a more balanced and comprehensive literature on groundwater-connected systems, it is necessary to diversity authorship, adapt conceptual models to reflect different environments, conduct local and regional scale investigations, and engage with and apply other research methods.

8.6 Reflections on the research process: threshold concepts and unlearning

A generous reading allows this dissertation to hold the potential to serve as a ‘threshold concept’ for groundwater sustainability science. Threshold concepts are described as being transformative (i.e., they significantly alter perceptions of a subject), integrative (i.e., they reveal connections that explain phenomena in new ways), and troublesome (i.e., they deviate from basic assumptions and challenge existing paradigms) (Meyer & Land, 2006; and as interpreted by Loring, 2020). Threshold concepts are “akin to a portal, opening up a new and previously inaccessible way of thinking about something” (Meyer & Land, 2006).

The groundwater-connected systems framing can foster a number of shifts in groundwater sustainability science. These include shifts toward adopting more interdisciplinary methods, expanded system boundary conditions, more social-ecologically aligned thinking, and a greater engagement with the overall complexity of groundwater systems. I have found the insistent force of the framing to continually observe groundwater in this relational sense to be a powerful, challenging, disarming, and generative concept to work with. In these ways, the groundwater-connected systems framing has acted on me in ways consistent with a threshold concept. Yet, as reminded by Loring (2020), threshold concepts are not necessarily “gateways from lesser to more advanced ways of thinking: rather, they simply indicate important discontinuities among different ways of seeing and knowing the world.”

Doing so has accompanied a substantial amount of unlearning in my research process. This unlearning has been essential to navigate the tensions conventionally found between engineering and social science disciplines, as outlined in Chapter 1. These unlearnings have centered on reevaluating the positivistic and often reductionist orientations of traditional engineering scholarship and shifting to develop a more reflexive, complexity-oriented, and critical realist approach to science.

In what ways has this unlearning materialised in the work I have led here? It is hard to say exactly, as this recalibrating of fundamental research philosophies impacts, on a fundamental level, what has value and legitimacy as a research activity. For instance, this process has helped clarify and refine my personal sustainability orientations, in which I am more interested in shaping and exploring sustainability discourses and transformations rather than seeking to contribute to “fix and control” types of solutions (McCrory et al., 2022). In this regard, this unlearning has shaped how I offer and present my research, as I do not intend for outcomes to be understood as definitive and singular accounts of research questions but rather as robust and partial insights for consideration in a wider body of knowledge and context.

Inwardly, this has lent me a greater humility as I have become more aware of the partiality inherent in my (and in any individual's) research practice. I am sympathetic to readings of this work that may find the way that I have combined sustainability science and groundwater science as reliant on accessible and easily quantifiable social-ecological system elements. For instance, I have largely adapted natural science methods to fit with sustainability science data and frameworks, rather than engage with qualitative methods that sustainability and social scientists may be more accustomed to. Doing so may artificially mask over or undervalue the challenging reality that social system dynamics may simply not fit well with any quantification approach. These are valid and important criticisms. Yet, these realizations double as opportunities to facilitate integration between diverse research communities through the groundwater-connected systems framing.

The groundwater-connected systems framing has guided and enriched my perspective that groundwater sustainability is a powerful and underutilized tool to support wider social, economic, ecological, and Earth system goals. Leading research that reflects this perspective has revealed groundwater science as a more diverse, pluralistic, contested, and interesting field than the one I believed I was stepping into at the beginning of my PhD. My experience as a groundwater scientist has been profoundly enriched by this process. I hope the groundwater-connected systems framing will foster similar experiences for others.

8.7 References

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APPENDICES

Appendix I

GLOSSARY OF ABBREVIATIONS

ABM	Agent-based model
CAS	Complex adaptive system
EGBC	Engineers and Geoscientists British Columbia
GDE	Groundwater-dependent ecosystem
GDP	Gross domestic product
GRACE	Gravity Recovery and Climate Experiment
GWS	Groundwater storage
IUCN	International Union for Conservation of Nature
IWRM	Integrated water resources management
OECM	Other effective area-based conservation measures
PA	Protected area
SES	Social-ecological system
TWS	Terrestrial water storage
RGS	Relative groundwatershed size
SDGs	Sustainable Development Goals
SOM	Self-organising map
UGR	Underprotected groundwatershed ratio
VSI	Vegetation sensitivity index

WDPA	World Database on Protected Areas
WGS84	World Geodetic System 1984 (geodetic datum and coordinate system)
WHYMAP	World-wide Hydrogeological Mapping and Assessment Programme
UNEP	United Nations Environment Programme
UNESCO	United Nations Educational, Scientific and Cultural Organization

Appendix II

SUPPLEMENTARY INFORMATION FOR CHAPTER 2

II.1 Extended description of the groundwater-connected systems conceptual diagram

Figure 2.3a illustrates our understanding of groundwater-connected systems as forms of social-ecological systems. We do not perceive this claim to be particularly contentious as groundwater irrigation communities were among the first social-ecological systems studied (Ostrom, 1993). Yet, there is no consensus methodology to define a social-ecological system. There are, however, several frameworks to study social-ecological systems, including the prominent Social-Ecological Systems Framework (Figure 2.3b; Partelow, 2018). In Figure 2.3a, we attempt to sufficiently visualise the social-ecological nature of groundwater-connected systems based on associations with the Social-Ecological Systems Framework.

In the Social-Ecological System Framework, a social-ecological system is understood as a system comprised of resource systems, resource units, governance systems, and actors. The interactions between these elements occur in the context of related ecosystems, and social, economic, and political settings (Ostrom, 2009; McGinnis & Ostrom, 2014). These interactions constitute a core consideration of social-ecological system analysis (Reyers & Selomane, 2018) and are represented in the Social-Ecological Systems Framework as action situations, where interactions occur and produce outcomes. While the Social-Ecological Systems Framework's prominence has led to its near inseparability from the underlying social-ecological systems concept, we note there are many other frameworks to study social-ecological systems (Binder et al., 2013). Beside from the framework's prominence, we selected the Social-Ecological Systems Framework for our illustration as it has useful structural properties that enable consideration of social and ecological components in equal depth, enables consideration of interactions and feedbacks between social and ecological processes, and provides a thorough index of social-ecological system components to possibly consider in an individual system's study (Binder et al., 2013; Partelow, 2018).

In the paragraphs below, we summarize the groundwater-connected systems we associated with Social-Ecological System Framework elements in Figure 2.3a. We note that we did not follow a specific method of application nor forced our list to conform exactly to individually specified elements across the multiple tiers of the framework. Rather, we simply used the structure of the framework as an underlying mental model that guided our identification and classification of groundwater-connected systems.

Resource units are the physical resources of the social-ecological system. These are often the resources considered in common-pool resource studies. In the context of groundwater-connected systems, the resource units are elements of the terrestrial water cycle and include stores of groundwater, soil moisture, and surface water. We list these other freshwater stores to align with conjunctive management and One Water paradigms (Villholth, 2021) that call for freshwater resources to be considered together rather than in isolated contexts.

Resource systems are the physical systems that contain and/or control attributes of the *resource units*. For groundwater resources, these are geologic units, topographic regions, and climate zones (Gleeson & Manning, 2008). Humans alter these systems through land use change (Scanlon et al., 2005), climate change (Taylor et al., 2013), and large infrastructure projects. Yet, these actions and impacts are better understood not as part of the *resource systems* but as *outcomes* that are produced by interactions between these resources and human actors. These outcomes then feed back to change the resource systems listed above.

Governance systems include governmental and civil society organizations, their structure, the rules they impose, and the monitoring and sanctioning they perform. The *governance systems* we list for groundwater-connected systems are:

- International policy goals, such as the United Nations Sustainable Development Goals.
- Groundwater monitoring programs and funding
- The integrated water resources management (IWRM) paradigm,
- Groundwater governance institutions and arrangements

These various systems, forms, and attributes of governance set the structure and rules for the actors in the system. Although the example of "groundwater governance institutions and arrangements" most directly represents the governance systems Framework element, we also include international policy goals and the IWRM paradigm as these concepts provide examples of the backdrop, setting, and influences under which governance systems are developed. Groundwater monitoring programs are included as a core component of groundwater management.

Actors represent the various people embedded in the social-ecological system, as well as their attributes that include norms, socioeconomic traits, and relationships with the resource and governance systems. The *actors* element also enables representation of collective entities (McGinnis & Ostrom, 2014). Listed in the figure are:

- *Agricultural* actors which we include on the basis that the agricultural sector represents the largest consumer of groundwater resources globally, and farmers are a core, heterogeneous demographic in many groundwater-connected systems.
- *Domestic* actors such as rural communities and households who rely on private wells for water security.
- *Communities* which we use to represent individuals and groups of people who use or are in relationship with groundwater or surface expressions of groundwater (e.g., springs, rivers) through recreational, cultural, spiritual, religious practices.
- *Researchers* who observe, collect data, and study these systems. In studies on groundwater-connected systems, researchers need to develop awareness about the role their training and positionality affects the ways they interact with, observe, and study these systems (*sensu* Lane, 2014).
- *Industrial* actors that represent the industries which use groundwater for mining, manufacturing, and electricity generation, among other uses.

Related ecosystems to groundwater-connected systems are easily identifiable as groundwater-dependent ecosystems (GDEs). These include terrestrial, aquatic, and subterranean GDEs. We note that while this element in the Social-Ecological Systems Framework is called ‘related ecosystems’, these GDEs are integral components to the groundwater-connected systems framing. The limited ability of the framework to consider ecological complexities has spurred the development of derivative frameworks (e.g., Vogt et al., 2015). While this limitation does not affect our system mapping exercise, it is a notable reflection for those looking to find appropriate social-ecological frameworks for specific applications to groundwater-connected systems.

Social, economic, and political settings are included in the framework to represent the external conditions that situate the system. We include virtual water trade to represent the role of international trade networks in driving groundwater depletion (e.g., Dalin et al., 2017). However, the view of these trade networks as an external setting or governance system depends on the scale of analysis. In most local to regional scale studies (such as the diagram in Figure 2.3a depicts), international trade agreements operate at larger scales and can be considered as a broader economic and political setting. We also include ‘Human development dimensions and context’ to represent the variability in social norms, colonial legacies, and development pathways that contextualize groundwater-connected systems.

Action situations represent the interactions between the system components listed above and the outcomes they produce. In Figure 2.3a, we use the label *outcomes* as using the term *action situations* would require additional explanation for readers not familiar with the Social-Ecological Systems Framework. The outcomes we highlight in the figure are:

- *Saltwater intrusion* which can occur through coastal pumping and sea-level rise. Thus, saltwater intrusion can be driven by both local and global processes and can require adaptation to maintain agricultural productivity and water security in coastal regions.
- *Deteriorated ecosystem services*, such as a reduced ability to buffer floods and droughts, attenuate contaminants, and support groundwater-dependent ecosystems. Deteriorated ecosystem services directly feed back to affect people, and this linkage is made evident through a social-ecological systems framing.
- *Decreased crop yields and food security*, whose dynamics and implications vary widely across different types of groundwater economies, such as in arid agricultural systems, industrial agricultural systems, smallholder farming systems, or groundwater-supported pastoralism (Shah et al., 2007).
- *Dry wells and reduced rural water security*, as discussed in Box 2.1 in the main text.
- *Environmental and cultural flows not satisfied*, such as the disappearance of springs or reduction of streamflow due to groundwater pumping that harms practices such as ceremonies, sites of cultural significance, or recreational swimming.
- *Groundwater-based natural infrastructure (GBNI) projects*, such as managed aquifer recharge, riparian buffers, conservation agriculture, or aquifer storage and recovery. These GNBI projects promote physical groundwater sustainability and preserve groundwater-connected system functioning but require suitable conditions, that include legal frameworks, adequate funding, and physiographic conditions (Ulibarri et al., 2021).

II.2 Supplementary tables

Table II.1. Groundwater-connected systems that exhibit behaviour shared with common principles of complex adaptive systems. For additional reading, Bouchet et al. (2019) also provide evidence to consider groundwater systems as complex adaptive systems.

Complex adaptive system principle (from Preiser et al., 2018)	**Groundwater-connected system examples**
Constituted relationally meaning formed by network of system components which can be diverse and heterogeneous	Spatially distributed wells across a physiographically heterogeneous landscape. Multiple levels of decision making and policy, which includes United Nations Sustainable Development Goals, national groundwater monitoring programs, sub-national groundwater legislation, water boards and community groups, and individual well owners.
Adaptive capacities, which are a product of self-organization, decentralized control mechanisms, and system memory and resilience	Forming of water boards in response to declining water levels. Farmer adaptation strategies in response groundwater depletion. Infrastructure such as managed aquifer recharge. Establishing monitoring networks to support adaptive management.
Dynamic processes, which are characterized by multiple possible trajectories and multiple stable states, non-linear processes, thresholds, regime shifts, and feedback mechanisms	Multiple stable states in groundwater-surface water interactions. Rainfall-recharge thresholds impacted by climate change and land use. Water security thresholds such as water table depths at which wells run dry. Pumping induced sea water intrusion. Land subsidence from groundwater depletion that reduces aquifer storage capacity.
Radically open systems, where system boundaries are porous with free exchange of matter, information, and energy which contribute to teleconnections and telecoupling	International virtual water trade networks. Distal precipitation patterns driven by atmospheric moisture transport of evapotranspirated groundwater irrigation. Open-access scientific practices.

Contextually determined, meaning the function of the system changes as the system changes itself and where behavior can also exhibit path dependence	Place-based human preferences and belief systems. Ecosystem specific sensitivities to pumping and altered flow regimes. Social norms and preferences concerning collective action, compliance with regulation, and role of government. Local impacts of climate change on groundwater resources.
Emergence through complex causality, characterized by recursive causal mechanisms, multiple pathways of causality, and the possibility for various outcomes from the same starting conditions	Multiple drivers of groundwater storage trends. Evolution of social values of water. Social tipping points in response to groundwater regulations.

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Appendix III

SUPPLEMENTARY INFORMATION FOR CHAPTER 3

Table III.1 begins on the next page in landscape orientation.

Table III.1. Datasets included in global review.

Dataset	System	Specificity	Format	Type	Spatial resolution	Start date	End date	Time step	Authors	DOI
IPCC reference regions	Atmosphere	2_Implicit	Polygon	Zonal					Iturbide et al., 2020	https://doi.org/10.5194/essd-12-2959-2020
Köppen Geiger climate zones	Atmosphere	2_Implicit	Raster	Zonal	0.0100	1901-1930	2071-2099	30 years	Beck et al., 2023	https://doi.org/10.1038/s41597-023-02549-6
Change in extreme precipitation events	Atmosphere	2_Implicit	Raster	Static	0.2500	1971-2000	2071-2100		Gründemann et al., 2022	https://doi.org/10.1038/s43247-022-00558-8
Water table ratio	Atmosphere	1_Explicit	Raster	Static	1 km				Cuthbert et al., 2019	https://doi.org/10.1038/s41558-018-0386-4
CHELSA-BIOCLIM+ global climate variables	Atmosphere	2_Implicit	Raster	Time series	0.0083	1980	2018	Month	Brun et al., 2022	https://doi.org/10.5194/essd-14-5573-2022
CHELSA precipitation	Atmosphere	2_Implicit	Raster	Time series	0.0083	1979	2013	Month	Karger et al., 2017	doi.org/10.1038/sdata.2017.122
CHELSA precipitation projection	Atmosphere	2_Implicit	Raster	Time series	0.0449	2006	2100	Month	Karger et al., 2020	doi.org/10.1038/s41597-020-00587-y
Hydrometeorological CRU TS	Atmosphere	2_Implicit	Raster	Time series	0.5000	1901	2022	Month	Harris et al., 2020	https://doi.org/10.1038/s41597-020-0453-3
Precipitation GPCC	Atmosphere	2_Implicit	Raster	Time series	0.2500	1891	2019	Day	Schneider et al., 2022	http://dx.doi.org/10.5676/DWD_GPCC/FD_M_V2022_025

Table III.1. (cont'd, 2/n): Datasets included in global review.

Dataset	System	Specificity	Format	Type	Spatial resolution	Start date	End date	Time step	Authors	DOI
Biomes ecoregions	Biosphere	2_Implicit	Polygon	Zonal					Dinerstein et al., 2017	https://doi.org/10.1093/biosci/bix014
Freshwater ecoregions	Biosphere	2_Implicit	Polygon	Zonal	~300,000 km^2				Abell et al., 2008	https://doi.org/10.1641/B580507
Amphibian richness	Biosphere	2_Implicit	Raster	Static	0.0083	2015			IUCN & CIESIN, 2015	https://doi.org/10.7927/H4RR1W66
Ecohydrological classes forest growth	Biosphere	1_Explicit	Raster	Static	0.0083	NA			Roebroek et al., 2020	doi.org/10.5194/hess-24-4625-2020
Ecological conservation prioritization	Biosphere	2_Implicit	Raster	Static	10 km				Jung et al., 2021	https://doi.org/10.1038/s41559-021-01528-7
Environmental flow head limit	Biosphere	1_Explicit	Raster	Static	0.0833	1960	2100		de Graaf et al., 2019	doi.org/10.1038/s41586-019-1594-4
Environmental flow transgression year	Biosphere	1_Explicit	Raster	Static	0.0833	1960	2100		de Graaf et al., 2019	doi.org/10.1038/s41586-019-1594-4
GDE inference	Biosphere	1_Explicit	Raster	Static	0.0833	2015			Huggins et al., 2023	doi.org/10.1038/s41893-023-01086-9
Global lakes and wetlands	Biosphere	2_Implicit	Polygon	Static	0.0833				Lehner & Döll, 2004	doi.org/10.1016/j.jhydrol.2004.03.028

Table III.1. (cont'd, 3/n): Datasets included in global review.

Dataset	System	Specificity	Format	Type	Spatial resolution	Start date	End date	Time step	Authors	DOI
Groundwater dependent ecosystem probability	Biosphere	1_Explicit	Raster	Static	0.5000				Link et al., 2023	https://doi.org/10.1088/1748-9326/acea97
Groundwater driven wetlands	Biosphere	1_Explicit	Raster	Static	0.0042				Tootchi et al., 2019	https://doi.org/10.5194/essd-11-189-2019
Groundwater ecosystem biodiversity	Biosphere	1_Explicit	Raster	Static	0.0833				Saccò et al., 2024	https://doi.org/10.1111/gcb.17066
IUCN Red List richness	Biosphere	2_Implicit	Raster	Static	30 km	2024			IUCN, 2024	https://www.iucnredlist.org/resources/other-spatial-downloads
Mammal richness	Biosphere	2_Implicit	Raster	Static	0.0083	2015			IUCN & CIESIN, 2015	https://doi.org/10.7927/H4N014G5
Ramsar wetlands	Biosphere	1_Explicit	Point	Static					UNESCO	https://www.ramsar.org/
Root zone depth Stocker	Biosphere	2_Implicit	Raster	Static	0.0500	2013	2018		Stocker et al., 2023	doi.org/10.1038/s41561-023-01125-2
Root zone water storage capacity	Biosphere	2_Implicit	Raster	Static	0.0500				Stocker et al., 2023	doi.org/10.1038/s41561-023-01125-2
Ecological vulnerability	Biosphere	2_Implicit	Raster	Time series	0.0833	1990	2015	Annual	Varis et al., 2019	10.1029/2019EF001239

Table III.1. (cont'd, 4/n): Datasets included in global review.

Dataset	System	Specificity	Format	Type	Spatial resolution	Start date	End date	Time step	Authors	DOI
GWL FCS30 Wetland classification	Biosphere	2_Implicit	Raster	Time series	30 m	2000	2022	Annual	Zhang et al., 2024	doi.org/10.1038/s41597-024-03143-0
Maximum rooting depth	Biosphere	1_Explicit	Raster	Time series	0.0083	2004	2013	Month	Fan et al., 2017	https://doi.org/10.1073/pnas.1712381114
Vegetation health index	Biosphere	2_Implicit	Raster	Time series	4 km	1981	2021	Month	Zeng et al., 2023	https://doi.org/10.1038/s41597-023-02255-3
Vegetation Index Phenology VIP	Biosphere	2_Implicit	Raster	Time series	0.0500	1981	2014	Month	Didan & Barreto, 2016	doi.org/10.5067/MEaSUREs/VIP/VIP30.004
Vegetation indices MODIS NDVI EVI	Biosphere	2_Implicit	Raster	Time series	250m	2000	-	16 days	Didan, 2021	https://doi.org/10.5067/MODIS/MOD13Q1.061
BioTIME biodiversity time series	Biosphere	2_Implicit	Point	Historical record	Variable	1874	2016	Minimum annual	Dornelas et al., 2018	https://doi.org/10.1111/geb.12729
Area equipped for irrigation from groundwater	Food systems	1_Explicit	Raster	Static	0.0833	2005			Siebert et al., 2010	doi.org/10.5194/hess-14-1863-2010
Crop allocation food feed nonfood	Food systems	2_Implicit	Raster	Static	0.0833	2000			Cassidy et al., 2013	doi.org/10.1088/1748-9326/8/3/034015
Crop harvest area GAEZ	Food systems	2_Implicit	Raster	Static	0.0833	2015			Grogan et al., 2022	doi.org/10.1038/s41597-021-01115-2

Table III.1. (cont'd, 5/n): Datasets included in global review.

Dataset	System	Specificity	Format	Type	Spatial resolution	Start date	End date	Time step	Authors	DOI
Crop yield GAEZ	Food systems	2_Implicit	Raster	Static	0.0833	2015			Grogan et al., 2022	doi.org/10.1038/s41597-021-01115-2
Cropland area	Food systems	2_Implicit	Raster	Static	0.0833	2000			Ramankutty et al., 2008	10.1029/2007GB002952
Farm field size	Food systems	2_Implicit	Raster	Static	1 km	2017			Lesiv et al., 2019	https://doi.org/10.1111/gcb.14492
GFSAD global cropland datasets	Food systems	2_Implicit	Raster	Static	0.0083	2010			Teluguntla et al., 2016	https://doi.org/10.5067/MEaSUREs/GFSAD/GFSAD1KCM.001
Gridded livestock systems	Food systems	2_Implicit	Raster	Static	0.0833	2010			Gilbert et al., 2018	doi.org/10.1038/sdata.2018.227
Pasture area	Food systems	2_Implicit	Raster	Static	0.0833	2000			Ramankutty et al., 2008	10.1029/2007GB002952
SPAM agricultural production maps	Food systems	2_Implicit	Raster	Static	0.0833	2010			Yu et al., 2020	https://doi.org/10.5194/essd-12-3545-2020
Virtual water trade agriculture	Food systems	2_Implicit	Tabular	Static	Nation	2000	2015		Rosa et al., 2019	doi.org/10.1088/1748-9326/ab4bfc
Cropland expansion	Food systems	2_Implicit	Raster	Time series	0.0003	2003	2019	4 years	Potapov et al., 2022	https://doi.org/10.1038/s43016-021-00429-z

Table III.1. (cont'd, 6/n): Datasets included in global review.

Dataset	System	Specificity	Format	Type	Spatial resolution	Start date	End date	Time step	Authors	DOI
Irrigation areas	Food systems	2_Implicit	Raster	Time series	0.0833	2001	2015	Annual	Nagaraj et al., 2021	10.1016/j.advwatres.2021.103910
Water footprint crops	Food systems	2_Implicit	Raster	Time series	0.0833	1990	2019	Annual	Mialyk et al., 2024	https://doi.org/10.1038/s41597-024-03051-3
Yield gaps	Food systems	2_Implicit	Raster	Time series	0.0833	1975	2010	Annual	Gerber et al., 2024	doi.org/10.1038/s43016-023-00913-8
Administrative units	Governance	2_Implicit	Polygon	Zonal	Nation				GADM	https://gadm.org
Indigenous colonial treaties	Governance	2_Implicit	Polygon	Zonal					Native Land Digital	https://native-land.ca/
Indigenous territories	Governance	2_Implicit	Polygon	Zonal					Native Land Digital	https://native-land.ca/
Peak RepRisk ESG risk index	Governance	2_Implicit	Polygon	Static	Nation	2016	2018		Kuzma et al., 2023	10.46830/writn.23.00061
Environmental performance index	Governance	2_Implicit	Tabular	Time series	Nation	1950	2022	Annual	Wolf et al., 2022	https://epi.yale.edu/downloads/epi2022report06062022.pdf
IWRM SDG6.5.1	Governance	1_Explicit	Polygon	Time series	Nation	2017	2023	3 years	IWRM Data Portal	http://iwrmdataportal.unepdhi.org/
Varieties of democracy	Governance	2_Implicit	Tabular	Time series	Nation	1789	2023	Annual	Coppedge et al., 2011	https://v-dem.net/documents/38/v-dem_codebook_v14.pdf

Table III.1. (cont'd, 7/n): Datasets included in global review.

Dataset	System	Specificity	Format	Type	Spatial resolution	Start date	End date	Time step	Authors	DOI
Worldwide governance indicators	Governance	2_Implicit	Tabular	Time series	Nation	1996	2022	Annual	Kaufmann et al., 2011	doi.org/10.1017/S1876404511200046
International river basin organizations	Governance	1_Explicit	Tabular	Historical record					Program in Water Conflict Management and Transformation	https://transboundarywaters.ceoas.oregonstate.edu/
International water event	Governance	1_Explicit	Tabular	Historical record		1948	2008		As above	https://transboundarywaters.ceoas.oregonstate.edu/
Water conflict map	Governance	1_Explicit	Point	Historical record					Pacific Institute, 2023	https://www.worldwater.org/water-conflict/
Water related intrastate conflict cooperation	Governance	2_Implicit	Tabular	Historical record					Kåresdotter et al., 2023	https://doi.org/10.1016/j.scitotenv.2023.161555
HydroBASINS	Hydrosphere	2_Implicit	Polygon	Zonal	Variable				Lehner & Grill, 2013	https://doi.org/10.1002/hyp.9740
Transboundary aquifers	Hydrosphere	0_Ground water	Polygon	Zonal	1 : 50,000,000				IGRAC, 2021	https://www.un-igrac.org/resource/transboundary-aquifers-world-map-2021
WHYMAP aquifers	Hydrosphere	0_Ground water	Polygon	Zonal					Richts et al., 2011	https://doi.org/10.1007/978-90-481-3426-7_10
Groundwater recharge	Hydrosphere	0_Ground water	Tabular	Static					Moeck et al., 2020	https://doi.org/10.1016/j.scitotenv.2020.137042

Table III.1. (cont'd, 8/n): Datasets included in global review.

Dataset	System	Specificity	Format	Type	Spatial resolution	Start date	End date	Time step	Authors	DOI
Groundwater response time	Hydrosphere	0_Groundwater	Raster	Static	1 km				Cuthbert et al., 2019	https://doi.org/10.1038/s41558-018-0386-4
Groundwater vulnerability floods droughts	Hydrosphere	1_Explicit	Polygon	Static					Richts & Vrba, 2016	https://doi.org/10.1007/s12665-016-5632-3
HydroLAKES	Hydrosphere	2_Implicit	Polygon	Static					Messager et al., 2016	https://doi.org/10.1038/ncomms13603
Lake bathymetry	Hydrosphere	2_Implicit	Raster	Static	0.0002777				Khazaei et al., 2022	doi.org/10.1038/s41597-022-01132-9
Modern groundwater volume	Hydrosphere	0_Groundwater	Polygon	Static		2010			Gleeson et al., 2016	doi.org/10.1038/ngeo2590
Perennial intermittent rivers streams	Hydrosphere	1_Explicit	Polyline	Static					Messager et al., 2021	https://doi.org/10.1038/s41586-021-03565-5
River classification	Hydrosphere	2_Implicit	Polyline	Static					Dallaire et al., 2019	https://doi.org/10.1088/1748-9326/aad8e9
River reach fragmentation	Hydrosphere	2_Implicit	Polyline	Static					Grill et al., 2019	https://doi.org/10.1038/s41586-019-1111-9
River width	Hydrosphere	2_Implicit	Polyline	Static					Allen & Pavelsky, 2018	https://doi.org/10.1126/science.aat0636

Table III.1. (cont'd, 9/n): Datasets included in global review.

Dataset	System	Specificity	Format	Type	Spatial resolution	Start date	End date	Time step	Authors	DOI
Streamflow indices	Hydrosphere	2_Implicit	Raster	Static	0.0500				Beck et al., 2015	https://doi.org/10.1175/JHM-D-14-0155.1
Surface water extent change	Hydrosphere	2_Implicit	Raster	Static	0.0003	2000	2020		Pickens et al., 2020	https://doi.org/10.1016/j.rse.2020.111792
ESA soil moisture	Hydrosphere	2_Implicit	Raster	Time series	0.2500	1978	2022	Day	Dorigo et al., 2017	https://doi.org/10.1016/j.rse.2017.07.001
Groundwater storage anomalies	Hydrosphere	0_Groundwater	Raster	Time series	0.2500	2003	-	Day	B. Li et al., 2019	https://doi.org/10.1029/2018wr024618
Runoff reconstruction	Hydrosphere	2_Implicit	Raster	Time series	0.5000	1902	2014	Month	Ghiggi et al., 2019	https://doi.org/10.5194/essd-11-1655-2019
Stream temperature projections	Hydrosphere	2_Implicit	Raster	Time series	0.0833	1976	2099	Week	Bosmans et al., 2022	doi.org/10.1038/s41597-022-01410-6
Terrestrial water storage anomalies	Hydrosphere	2_Implicit	Raster	Time series	0.5000	2002	-	Month	Watkins et al., 2015	doi.org/10.1002/2014JB011547
Water table depth	Hydrosphere	0_Groundwater	Raster	Time series	0.0083	2004	2013	Month	Fan et al., 2017	https://doi.org/10.1073/pnas.1712381114
Global dam watch consensus	Hydrosphere	2_Implicit	Point	Historical record					Pending	https://www.globaldamwatch.org/database

Table III.1. (cont'd, 10/n): Datasets included in global review.

Dataset	System	Specificity	Format	Type	Spatial resolution	Start date	End date	Time step	Authors	DOI
River discharge GRDC	Hydrosphere	2_Implicit	Point	Historical record					GRDC	doi.org/10.5675/GRDC_Report_46
Anthropogenic biomes	Integrative	2_Implicit	Raster	Zonal	0.0833				Ellis & Ramankutty, 2008	https://doi.org/10.1890/070062
Proximity to surface freshwater	Integrative	2_Implicit	Raster	Static	0.0083				Kummu et al., 2011	https://doi.org/10.1371/journal.pone.0020578
Dynamic world land use land cover	Integrative	2_Implicit	Raster	Time series	10 m	2015	-		Brown et al., 2022	doi.org/10.1038/s41597-022-01307-4
ESA land cover	Integrative	2_Implicit	Raster	Time series	300 m	1992	2015	Annual	ESA, 2017	http://maps.elie.ucl.ac.be/CCI/viewer/download/ESACCI-LC-Ph2-PUGv2_2.0.pdf
River basin resilience	Integrative	2_Implicit	Raster	Time series	0.0833	1990	2015	Annual	Varis et al., 2019	10.1029/2019EF001239
Global lithological map	Lithosphere	2_Implicit	Polygon	Static	0.5				Hartmann & Moosdorf, 2012	https://doi.org/10.1029/2012GC004370
Land subsidence	Lithosphere	1_Explicit	Raster	Static	0.0083	2010	2040		Herrera-García et al., 2021	https://doi.org/10.1126/science.abb8549
Near surface permeability	Lithosphere	1_Explicit	Polygon	Static	123 km^2	NA			Huscroft et al., 2018	10.1002/2017GL075860

Table III.1. (cont'd, 11/n): Datasets included in global review.

Dataset	System	Specificity	Format	Type	Spatial resolution	Start date	End date	Time step	Authors	DOI
Soil grids	Lithosphere	2_Implicit	Raster	Static	250 m				Hengl et al., 2017	https://doi.org/10.1371/journal.pone.0169748
World soil database	Lithosphere	2_Implicit	Raster	Static	0.0083				FAO/IIASA/ISRIC/ISS-CAS/JRC, 2009	https://www.fao.org/3/aq361e/aq361e.pdf
Indigenous languages	Socioeconomic	2_Implicit	Polygon	Zonal					Native Land Digital	https://native-land.ca/
Accessibility to cities	Socioeconomic	2_Implicit	Raster	Static	0.0083	2015			Nelson et al., 2019	doi.org/10.1038/s41597-019-0265-5
Development pressure indices	Socioeconomic	2_Implicit	Raster	Static	1 km				Oakleaf et al., 2019	https://doi.org/10.1038/s41597-019-0084-8
Global roads GRIP	Socioeconomic	2_Implicit	Raster	Static	0.0833				Meijer et al., 2018	https://doi.org/10.1088/1748-9326/aabd42
Global roads gROADS	Socioeconomic	2_Implicit	Polyline	Static					CIESIN & ITOS, 2013)	https://doi.org/10.7927/H4VD6WCT
Power plants	Socioeconomic	2_Implicit	Point	Static		2021			Global Energy Observatory et al., 2018	https://datasets.wri.org/dataset/globalpowerplantdatabase
Protected areas OECMs	Socioeconomic	2_Implicit	Polygon	Static					UNEP-WCMC & IUCN, 2024	https://www.protectedplanet.net/en

Table III.1. (cont'd, 12/n): Datasets included in global review.

Dataset	System	Specificity	Format	Type	Spatial resolution	Start date	End date	Time step	Authors	DOI
Electricity consumption	Socioeconomic	2_Implicit	Raster	Time series	1 km	1992	2019	Annual	Chen et al., 2022	doi.org/10.1038/s41597-022-01322-5
GDP projections	Socioeconomic	2_Implicit	Raster	Time series	0.0083	2030	2100	10 years	Wang & Sun, 2022	doi.org/10.1038/s41597-022-01300-x
Gini index nation WB	Socioeconomic	2_Implicit	Polygon	Time series	Nation	1963	2022	Annual	World Bank, 2024	https://datanalytics.worldbank.org/PIP-Methodology/
Gridded freshwater withdrawal sectoral	Socioeconomic	2_Implicit	Raster	Time series	0.5000	1970	2010	Month	Huang et al., 2018	https://doi.org/10.5194/hess-22-2117-2018
Gross domestic product GDP	Socioeconomic	2_Implicit	Raster	Time series	0.0833	1990	2015	Annual	Kummu et al., 2018	doi.org/10.1038/sdata.2018.4
Groundwater withdrawal	Socioeconomic	1_Explicit	Tabular	Time series	Nation	1964	-	Annual	FAO, 2023	https://data.apps.fao.org/aquastat/?lang=en
Human development index HDI	Socioeconomic	2_Implicit	Raster	Time series	0.0833	1990	2015	Annual	Kummu et al., 2018	doi.org/10.1038/sdata.2018.4
Human footprint	Socioeconomic	2_Implicit	Raster	Time series	0.0833	1990	2015	Annual	Varis et al., 2019	10.1029/2019EF001239
Human modification gradient	Socioeconomic	2_Implicit	Raster	Time series	300 m	1990	2017	5 years	Theobald et al., 2020	https://doi.org/10.5194/essd-12-1953-2020

Table III.1. (cont'd, 13/n): Datasets included in global review.

Dataset	System	Specificity	Format	Type	Spatial resolution	Start date	End date	Time step	Authors	DOI
Human settlement layer	Socioeconomic	2_Implicit	Raster	Time series	10m	1975	2030	5 years	Pesaresi & Politis, 2023	https://doi.org/10.2905/9F06F36F-4B11-47EC-ABB0-4F8B7B1D72EA
Land use change historical	Socioeconomic	2_Implicit	Raster	Time series	0.0100	1960	2019		Winkler et al., 2021	https://doi.org/10.1038/s41467-021-22702-2
LandScan ambient population	Socioeconomic	2_Implicit	Raster	Time series	0.0083	2000	2022	Annual	Sims et al., 2023	https://doi.org/10.48690/1529167
Migration grids historical	Socioeconomic	2_Implicit	Raster	Time series	0.0083	1970	2000	10 year	Sherbinin et al., 2012	http://iopscience.iop.org/1748-9326/7/4/045602
Migration grids recent	Socioeconomic	2_Implicit	Raster	Time series	0.0833	2000	2019	Annual	Niva et al., 2023	https://doi.org/10.1038/s41562-023-01689-4
ND-GAIN Adaptation Index	Socioeconomic	2_Implicit	Tabular	Time series	Nation	1995	2023		Chen et al., 2023	https://gain.nd.edu/assets/522870/nd_gain_countryindextechreport_2023_01.pdf
Nighttime lights	Socioeconomic	2_Implicit	Raster	Time series	0.0833	1992	2018	Annual	Li et al., 2020	doi.org/10.1038/s41597-020-0510-y
Population grids	Socioeconomic	2_Implicit	Raster	Time series	0.0083	2000	2020	5 years	CIESIN, 2018	https://sedac.ciesin.columbia.edu/downloads/docs/gpw-v4/gpw-v4-documentation-rev11.pdf

Table III.1. (cont'd, 14/14): Datasets included in global review.

Dataset	System	Specificity	Format	Type	Spatial resolution	Start date	End date	Time step	Authors	DOI
Population projections	Socioeconomic	2_Implicit	Raster	Time series	0.0083	2020	2100	Annual	Wang et al., 2022	https://doi.org/10.1038/s41597-022-01675-x
Social adaptive capacity	Socioeconomic	2_Implicit	Raster	Time series	0.0833	1990	2015	Annual	Varis et al., 2019	10.1029/2019EF001239
Subnational gender development inequality	Socioeconomic	2_Implicit	Polygon	Time series	Nation	1990	2017	Annual	Smits & Permanyer, 2019	doi.org/10.1038/sdata.2019.38
Urban land fraction	Socioeconomic	2_Implicit	Raster	Time series	1 km	2000	2100	10 years	Gao & Pesaresi, 2021	https://doi.org/10.1038/s41597-021-01052-0
Managed aquifer recharge schemes	Socioeconomic	1_Explicit	Point	Historical record	Site				Stefan & Ansems, 2018	https://doi.org/10.1007/s40899-017-0212-6

Table III.2. Example datasets excluded for not meeting certain search criteria but that are still potentially relevant for global groundwater studies. This list is not exhaustive but illustrates decision making processes which led to dataset inclusion or exclusion from our reviewed collection.

Dataset	**Rationale for exclusion from collection**
Spatial index of hydro-political issues. (Farinosi et al., 2018)	Requires emailing author to access data.
Sub-national economic output (Wenz et al., 2023)	Incomplete global coverage.
LivWell: subnational dataset on living conditions for women and their well-being (Belmin et al., 2022)	Incomplete global coverage.
Global Indigenous and community lands (https://www.landmarkmap.org/)	Community level data not available due to data sharing agreements.
International Tree-Ring Data Bank (Zhao et al., 2019)	Would benefit from analysis-ready data synthesis before integration in large-scale groundwater assessments.

Table III.3. Existing observational capacity and proposed spatial and temporal resolution for potential essential variables.

Essential variable	Justification / rationale
Water table	**Existing observational capacity**: Yes, but limited to global monitoring network or model results. **Spatial availability at proposed 5 arcminute resolution**: Would require model-based interpolation between monitoring locations. **Temporal frequency**: Monthly frequency is available at monitoring locations but is currently not provided across the global land surface.
Groundwater storage anomaly	**Existing observational capacity**: Yes, through GRACE but requires model-based estimates of other terrestrial water storage components to isolate groundwater storage signal. **Spatial availability at proposed 5 arcminute resolution**: No, the base resolution of GRACE is on the order of 200,000 km^2. **Temporal frequency**: Yes, currently collected at monthly time step.
Groundwater baseflow	**Existing observational capacity**: No, only available through model-based estimations. **Spatial availability at proposed 5 arcminute resolution**: Yes, for model data, no for observational data. **Temporal frequency**: Monthly frequency is possible for model data.
Groundwater-dependent ecosystems	**Existing observational capacity**: Partially. Existing global groundwater-dependent ecosystem maps have been produced through inference-based methods and random forest predictions. Neither offer direct observations of groundwater-dependent ecosystems. **Spatial availability at proposed 5 arcminute resolution**: Current global maps have been produced at 1km and 30m resolution. **Temporal frequency**: Groundwater-dependent ecosystem prediction using random forests at monthly time steps are possible through remote sensing data, but not for direct observations.
Groundwater irrigation	**Existing observational capacity**: No. Current irrigation data are based on sub-national irrigation statistics but are not actively updated. **Spatial availability at proposed 5 arcminute resolution**: Existing data are available at 5 arcminute resolution.

	Temporal frequency: There is currently no temporally dynamic open data on global groundwater irrigation rates.
Groundwater supply of domestic water	**Existing observational capacity**: No. **Spatial availability at proposed 5 arcminute resolution**: Data available through statistical models to estimate at 5 arcminute resolution but no existing open dataset provides this information. **Temporal frequency**: There is currently no temporally dynamic open data on global groundwater supply of domestic water.
Observation well density	**Existing observational capacity**: Yes, through the global groundwater monitoring network, but limited to certain jurisdictions. **Spatial availability at proposed level of subnational jurisdictions**: Yes, but without complete global coverage. **Temporal frequency**: Unclear update frequency of the global groundwater monitoring network, and how active wells are identified from inactive observation wells.
Integrated groundwater management implementation	**Existing observational capacity**: Yes, through the IWRM Data Portal. **Spatial availability at proposed level of subnational jurisdictions**: No, current global groundwater management data are only available at the national scale. **Temporal frequency**: Yes, this data is currently collected at 3-year intervals.

III.1 Supplementary references

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Appendix IV

SUPPLEMENTARY INFORMATION FOR CHAPTER 4

IV.1 Supplementary figures

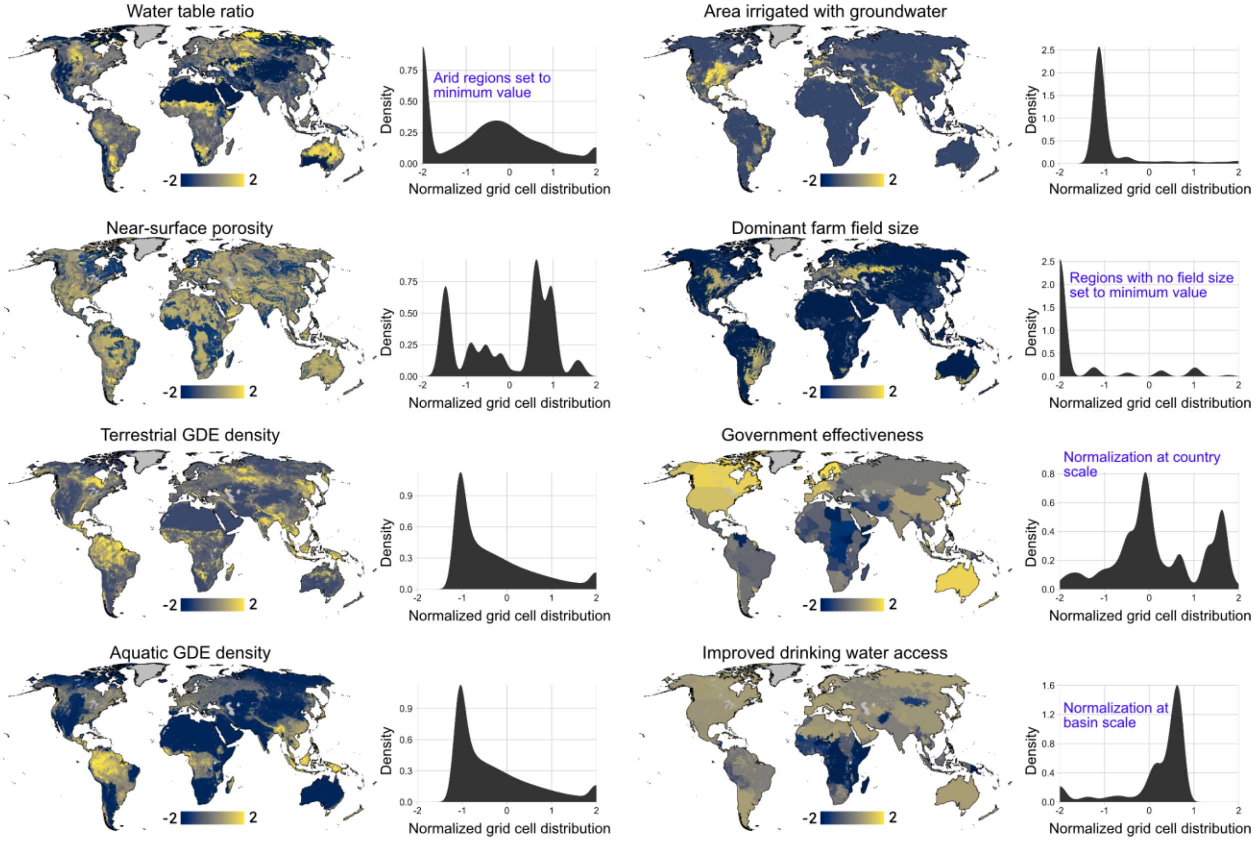


Figure IV.1. Input raster layers after preprocessing and normalization.

Grid cell distributions (not area distributions) for each raster layer are shown to the right of each map.

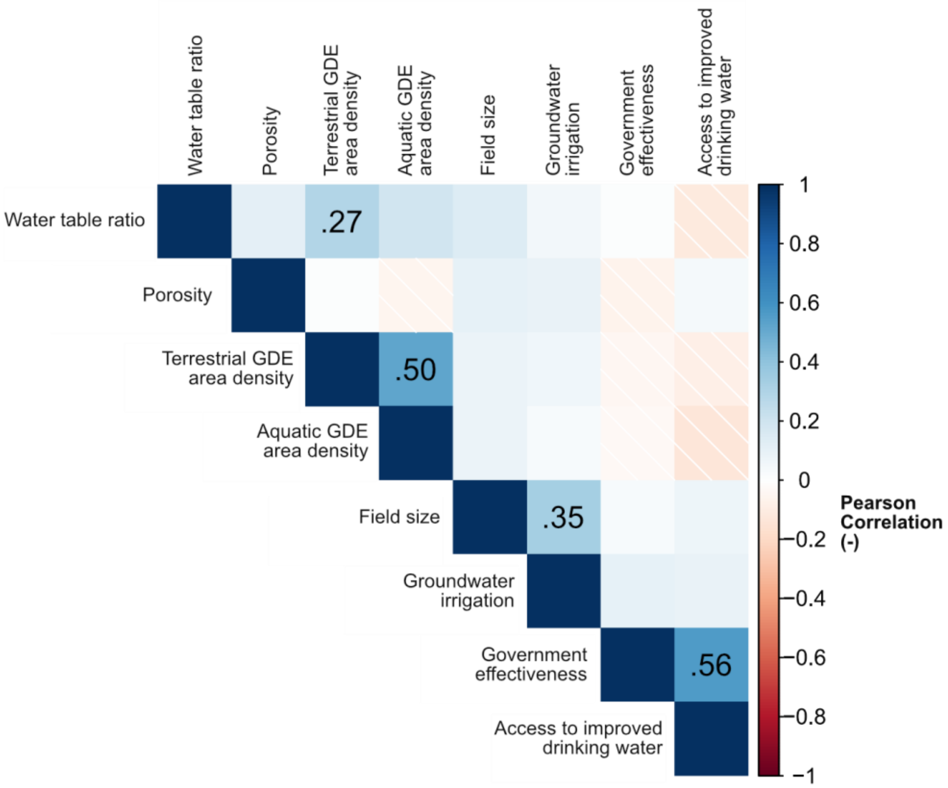


Figure IV.2. Pearson correlation coefficients of input datasets.

The four strongest correlations have their coefficients annotated.

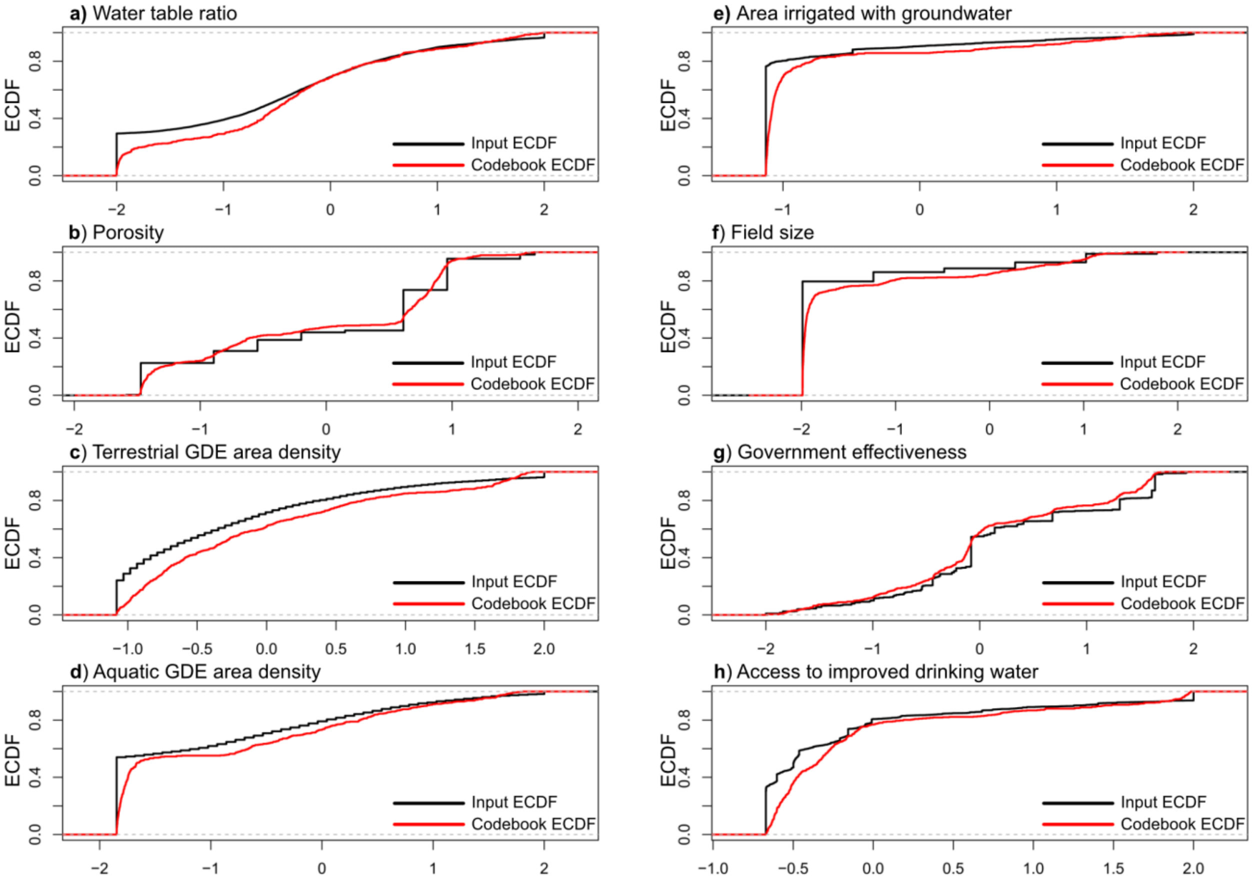


Figure IV.3. Comparison of input data empirical cumulative distribution functions (ECDFs) with first-stage SOM codebook vector ECDFs.

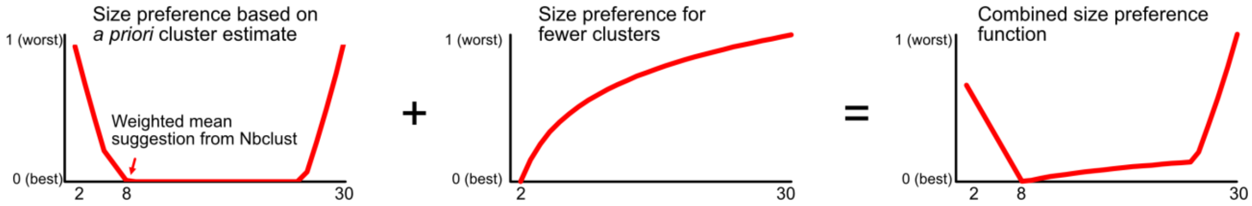


Figure IV.4. Development of the cluster count preference function that is implemented in the second-stage SOM performance sensitivity analysis.

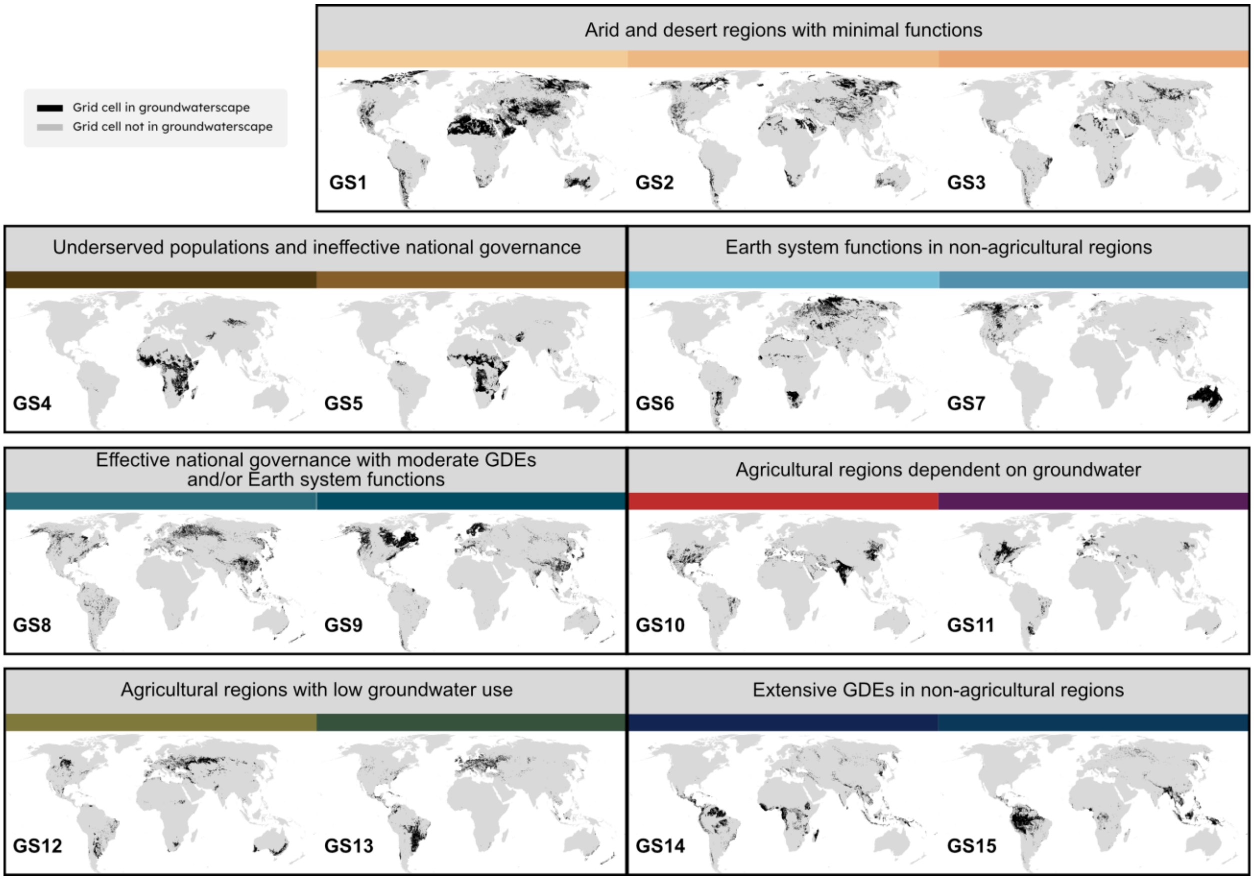


Figure IV.5. Spatial extent of individual groundwaterscapes.

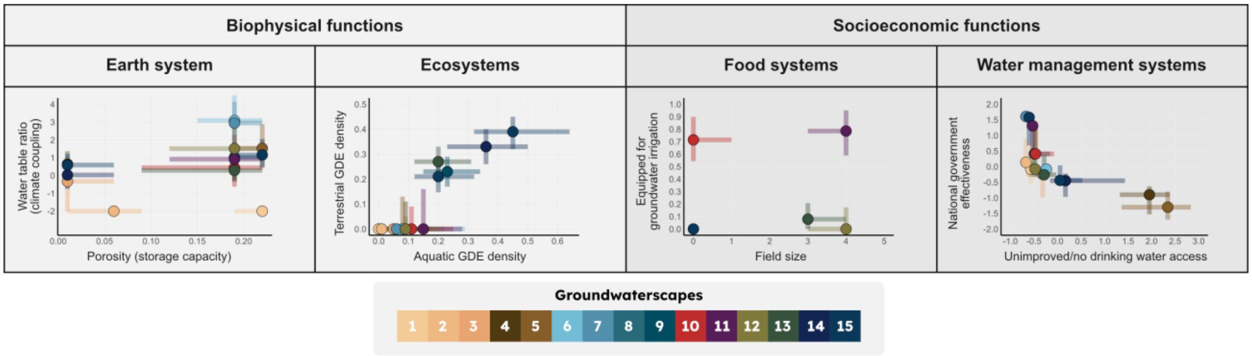


Figure IV.6. Interquartile function ranges for each groundwaterscape.

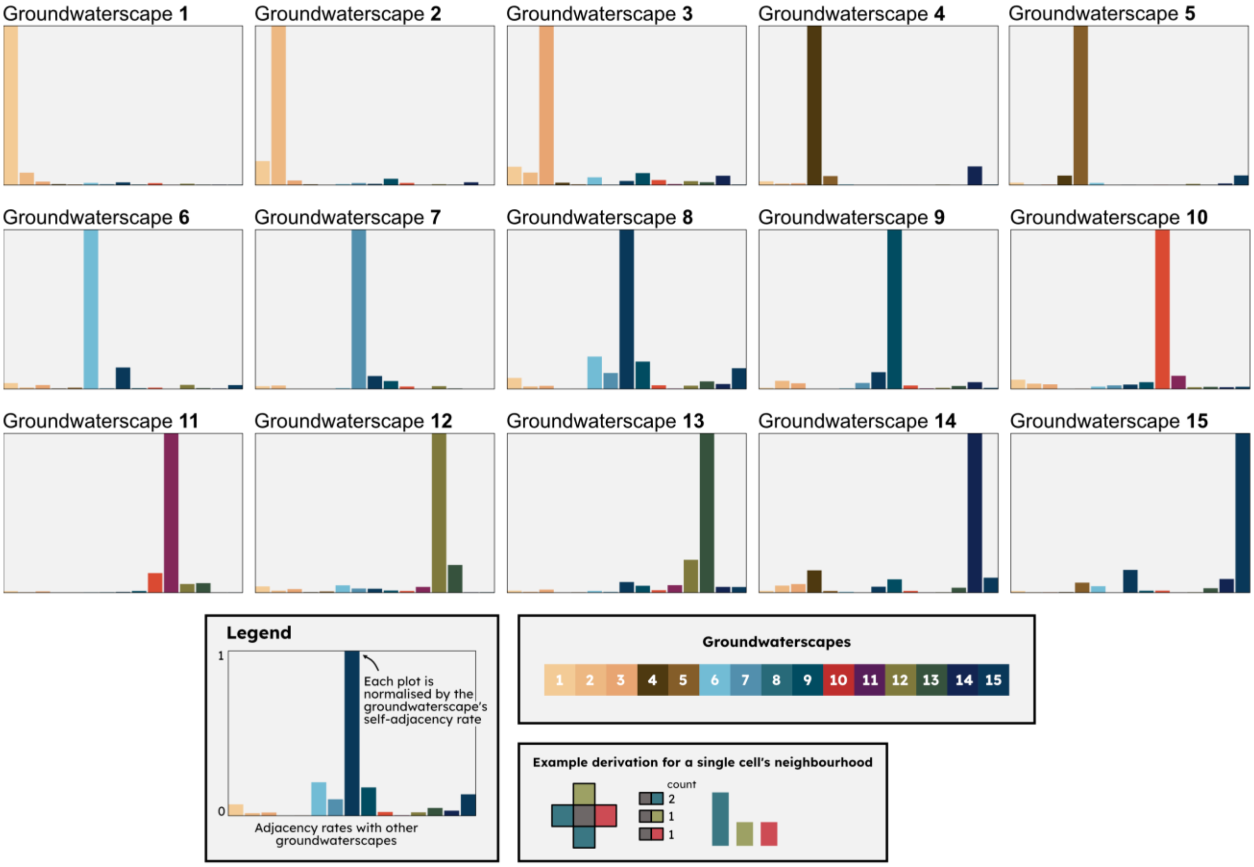


Figure IV.7. Grid cell adjacency analysis of groundwaterscapes reveal the frequency of neighbouring groundwaterscape pairs.

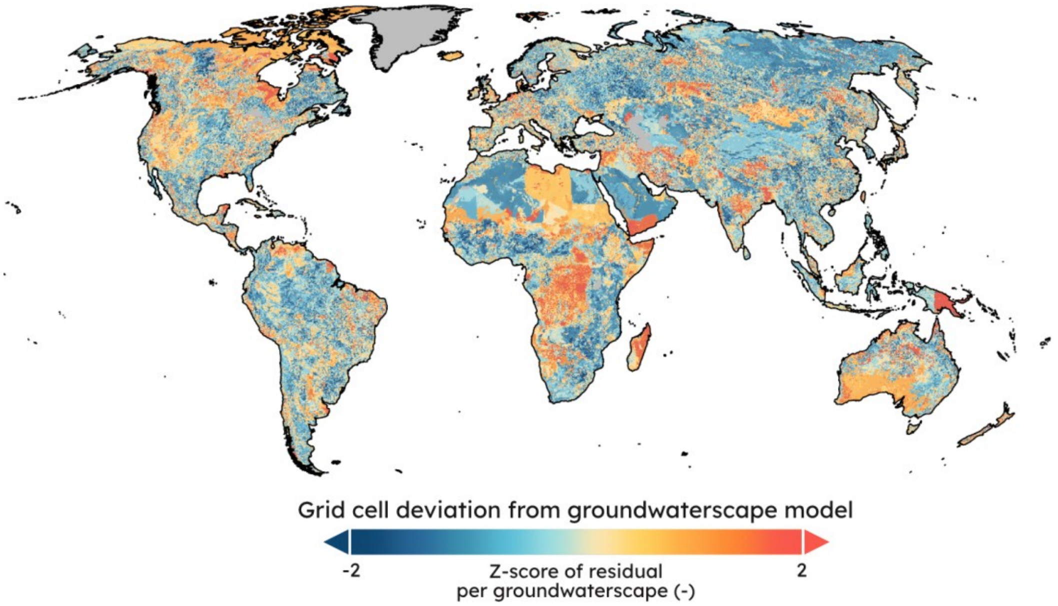


Figure IV.8. Groundwaterscape residuals.

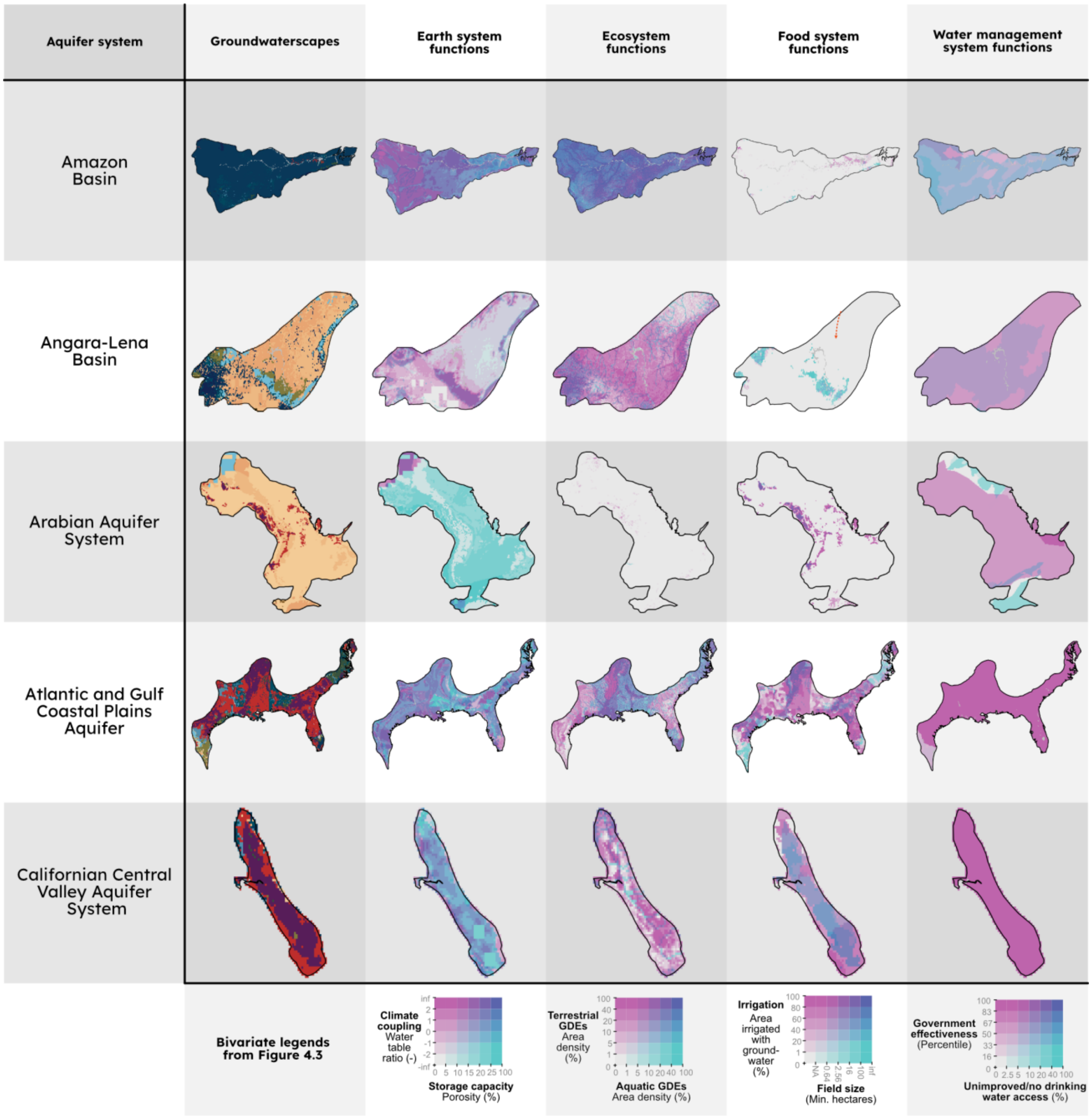


Figure IV.9. Groundwaterscape and function distributions within WHYMAP aquifers (Aquifer IDs 1-5).

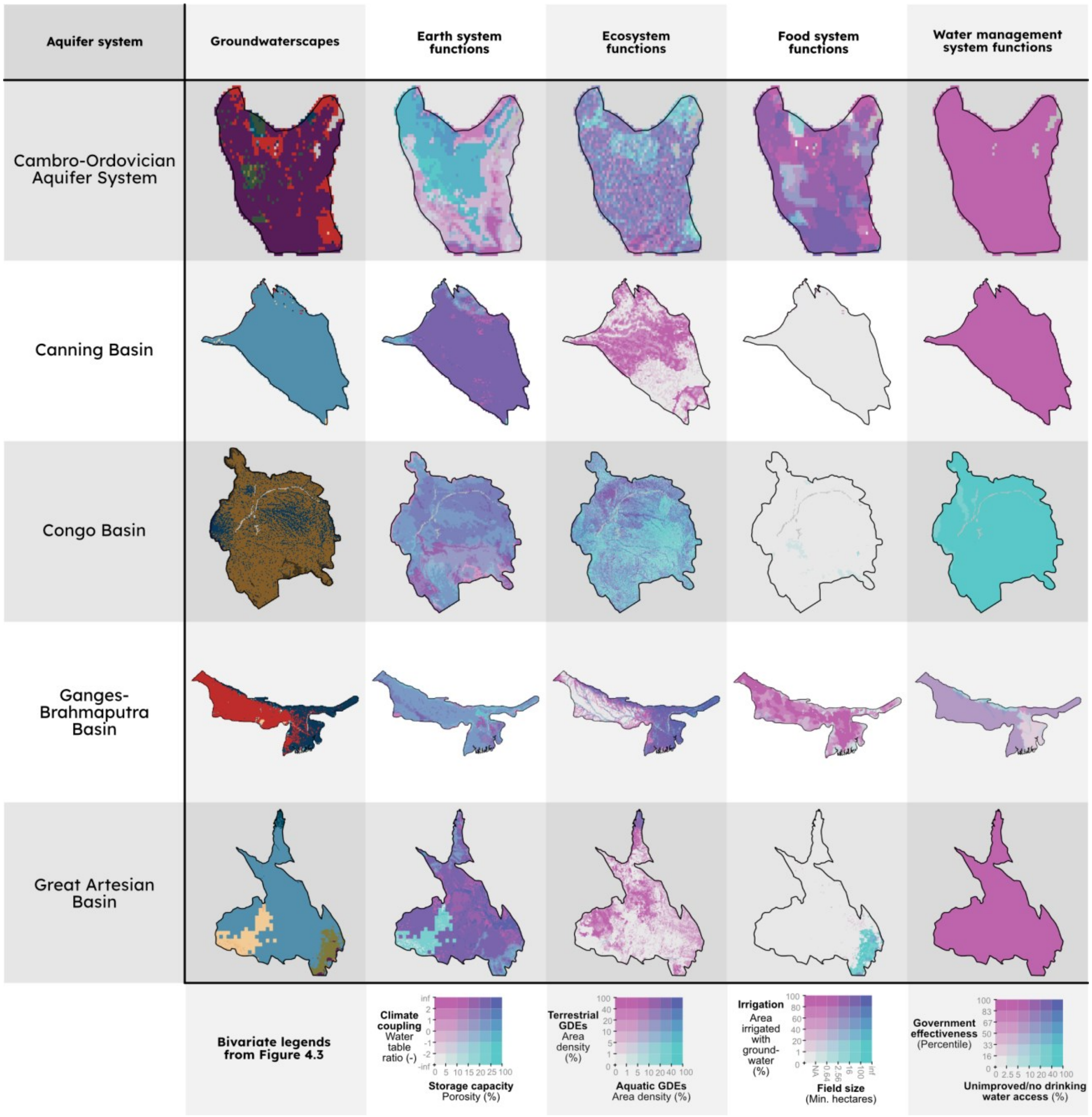


Figure IV.10. Groundwaterscape and function distributions within WHYMAP aquifers (Aquifer IDs 6-10)

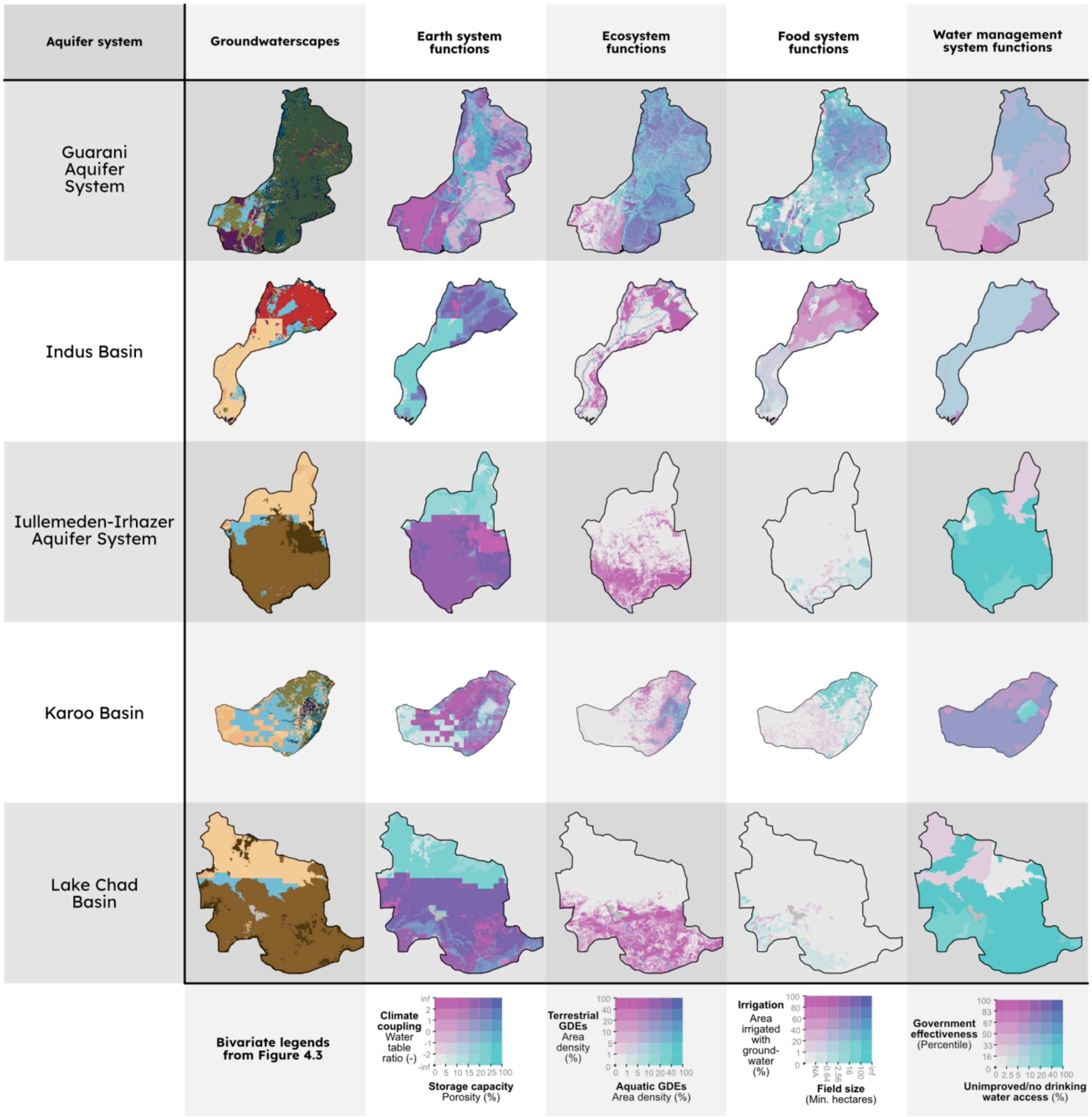


Figure IV.11. Groundwaterscape and function distributions within WHYMAP aquifers (Aquifer IDs 11-15)

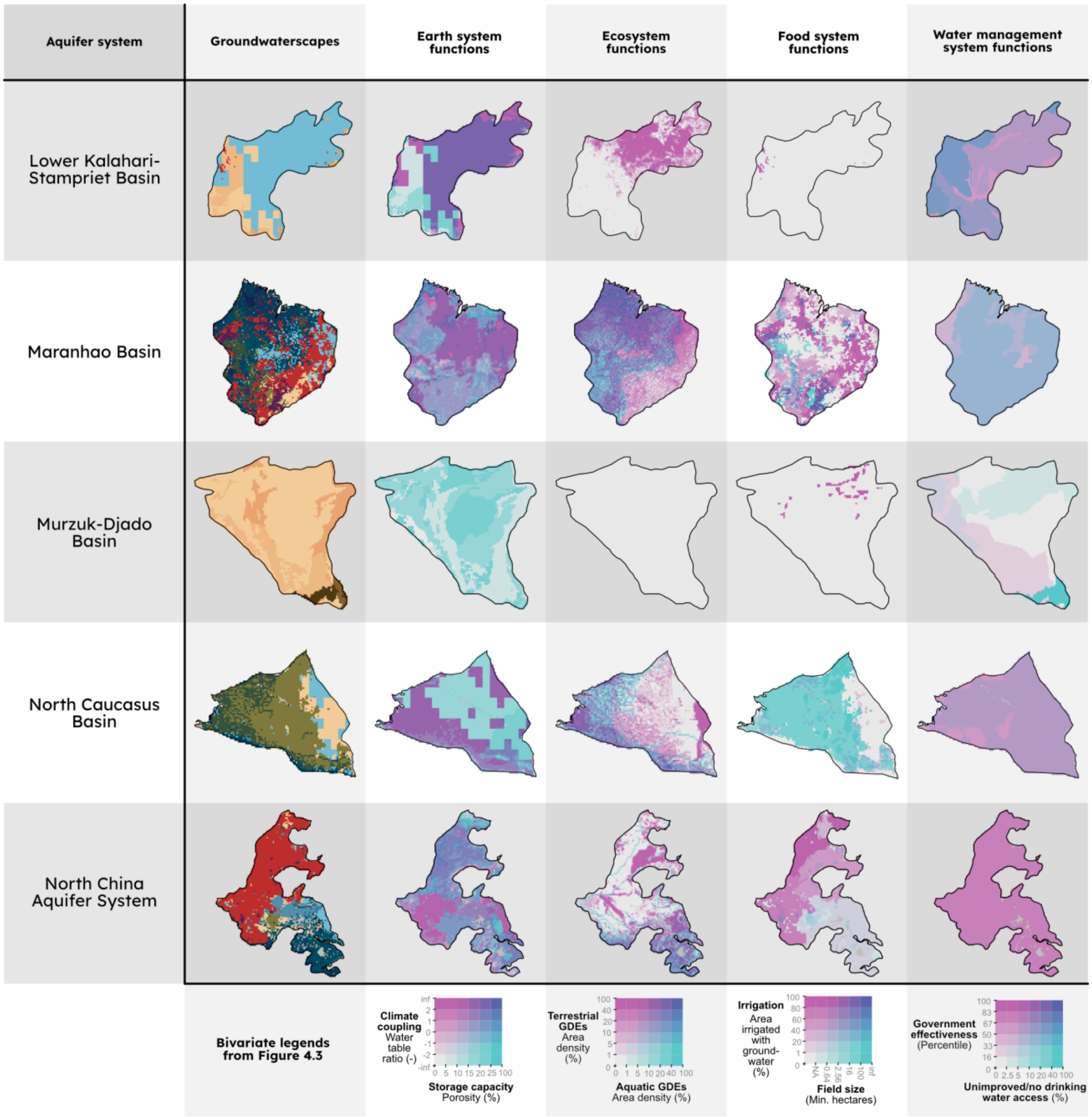


Figure IV.12. Groundwaterscape and function distributions within WHYMAP aquifers (Aquifer IDs 16-20)

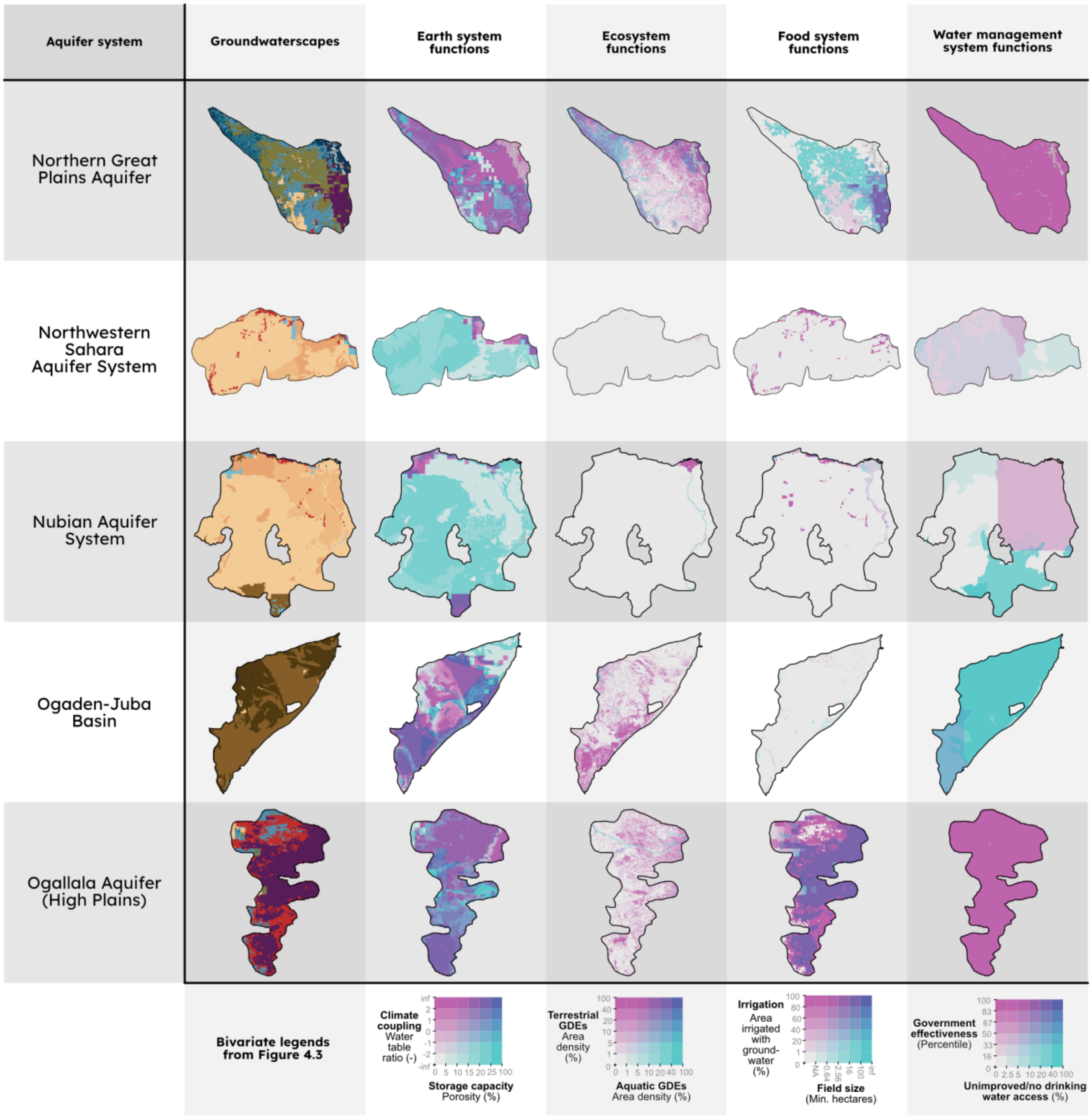


Figure IV.13. Groundwaterscape and function distributions within WHYMAP aquifers (Aquifer IDs 21-25)

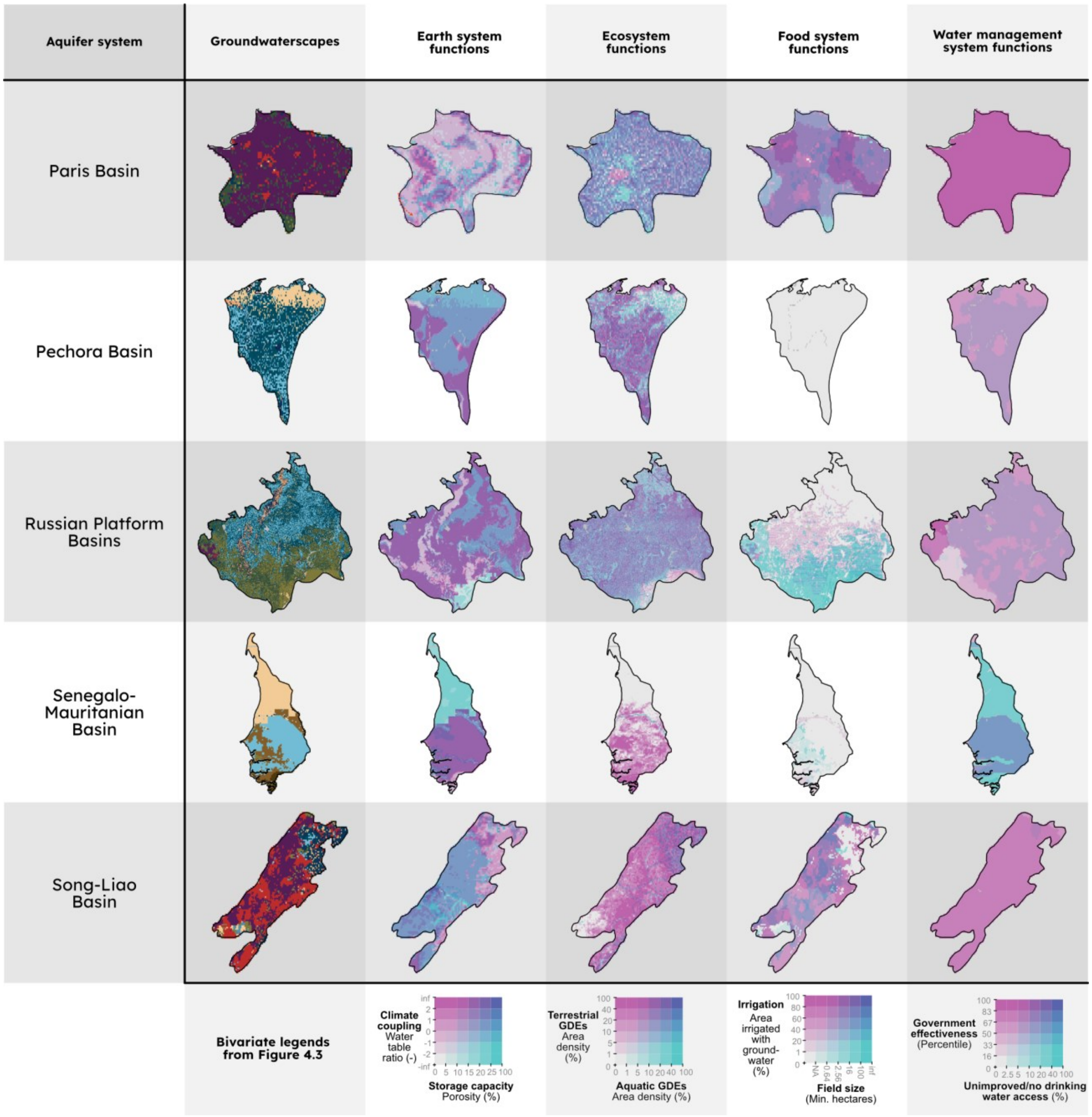


Figure IV.14. Groundwaterscape and function distributions within WHYMAP aquifers (Aquifer IDs 26-30)

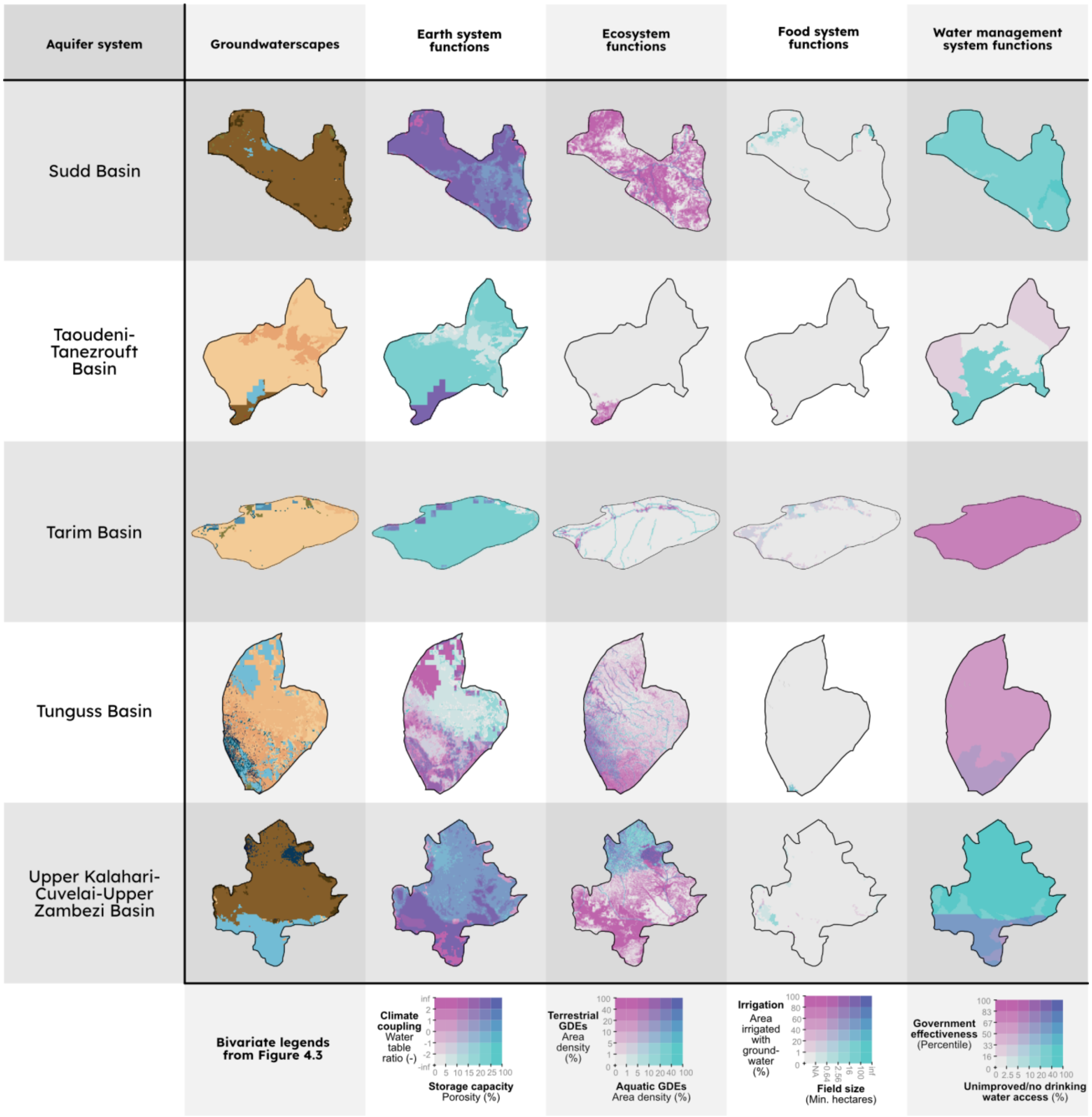


Figure IV.15. Groundwaterscape and function distributions within WHYMAP aquifers (Aquifer IDs 31-35)

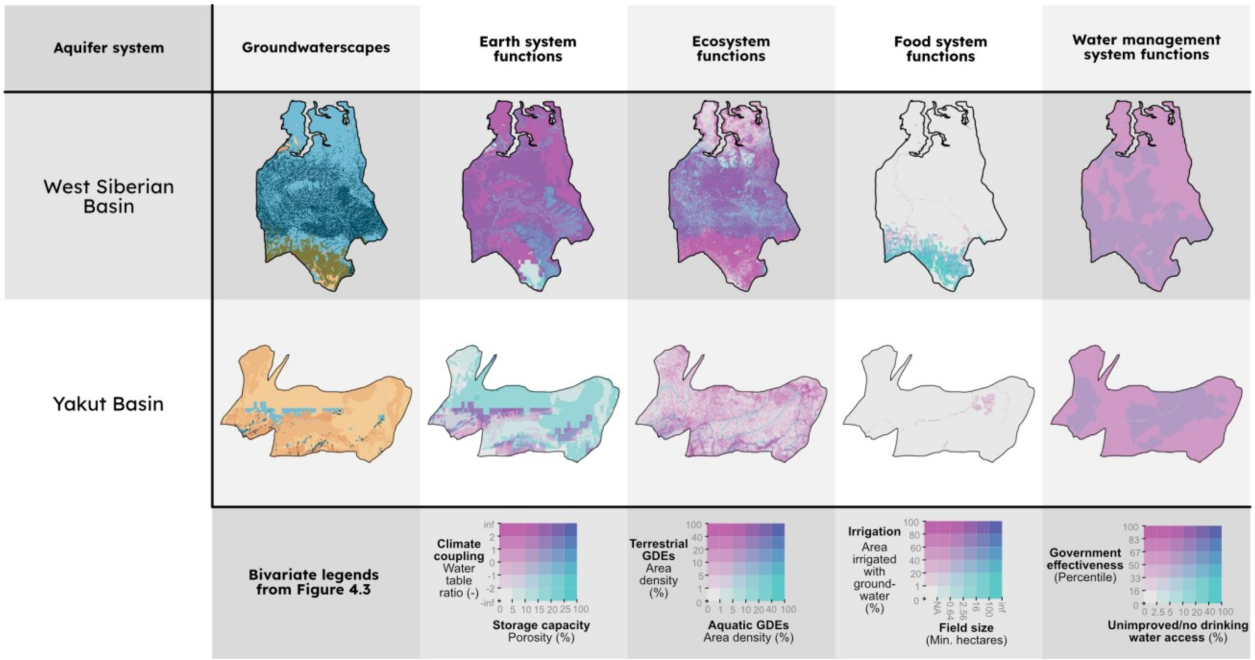


Figure IV.16. Groundwaterscape and function distributions within WHYMAP aquifers (Aquifer IDs 36-37)

Appendix V

SUPPLEMENTARY INFORMATION FOR CHAPTER 5

V.1 Supplementary figures and text

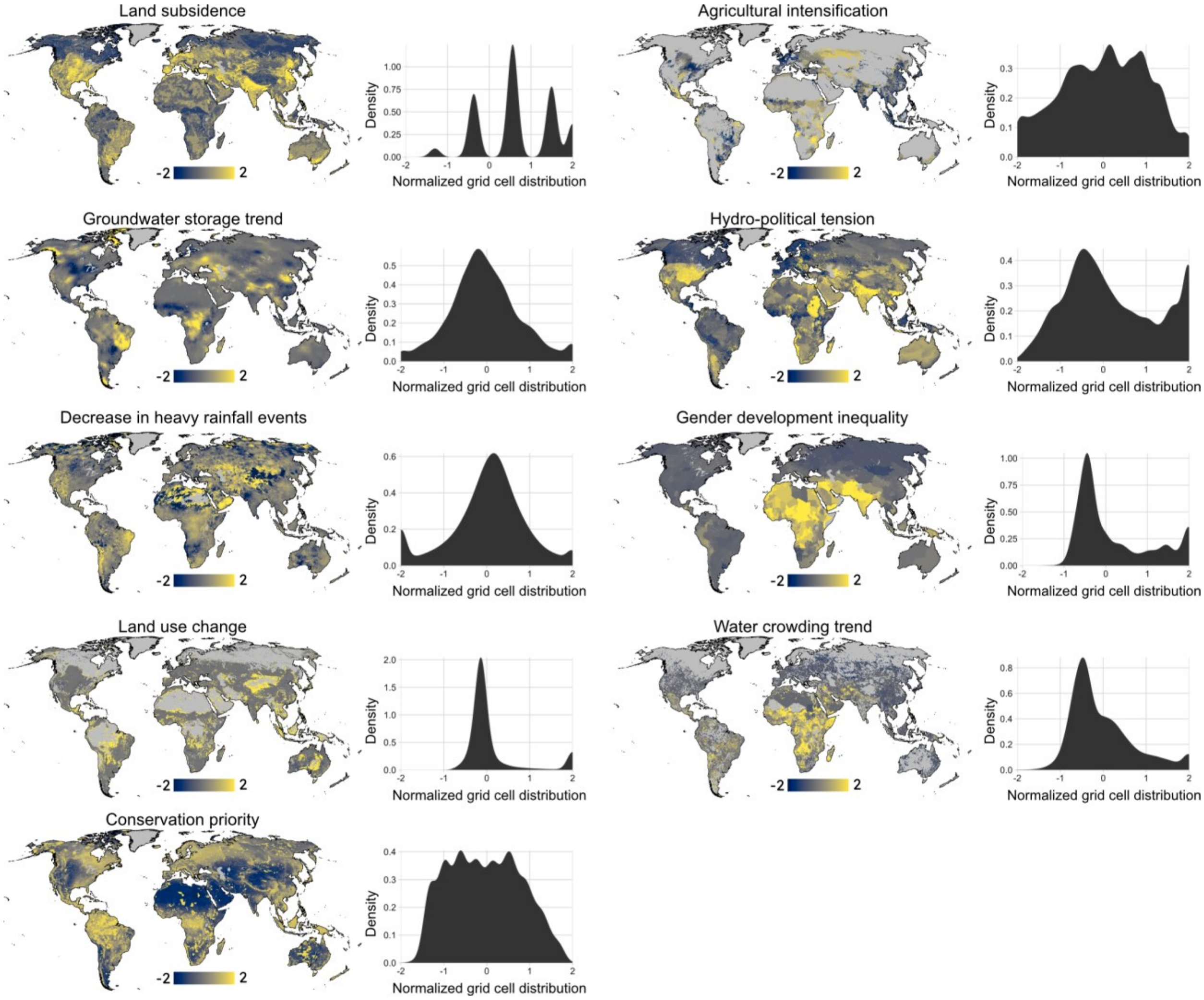


Figure V.1. Input risk data after preprocessing and normalization.

Grid cell distributions (not area distributions) for each raster layer are shown to the right of each map.

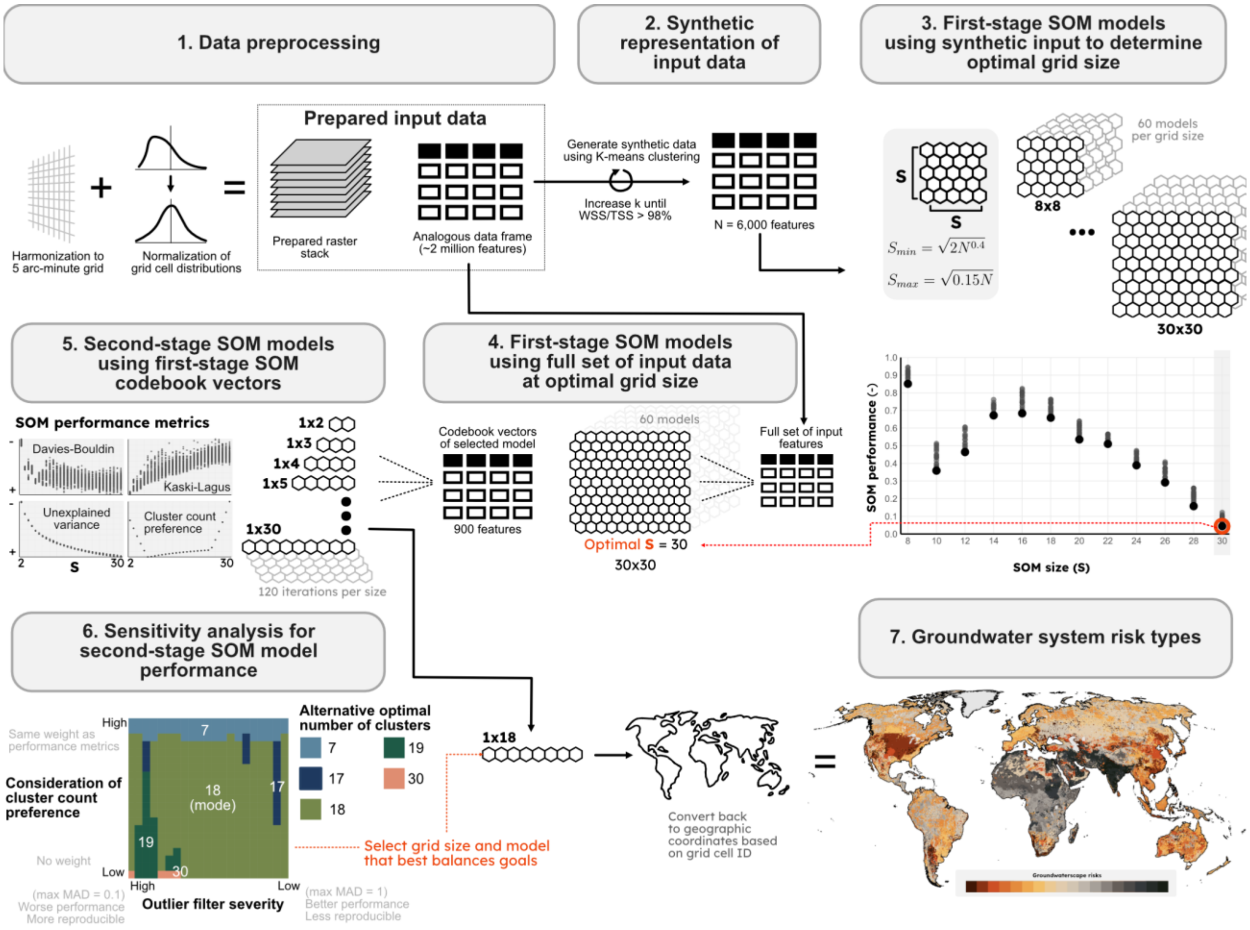


Figure V.2. Full groundwater risk type derivation methodology.

To support interpretation, follow the numbered boxes.

Accompanying text:

This figure provides an overview of the methodology to derive groundwater system risk types through a two-stage self-organizing map approach. This derivation method was only implemented for grid cells with full data coverage. Grid cells with partial input data coverage were assigned to a risk type based on the grid cell's nearest feature in the normalized input feature space (Euclidean distance) based on the available input features.

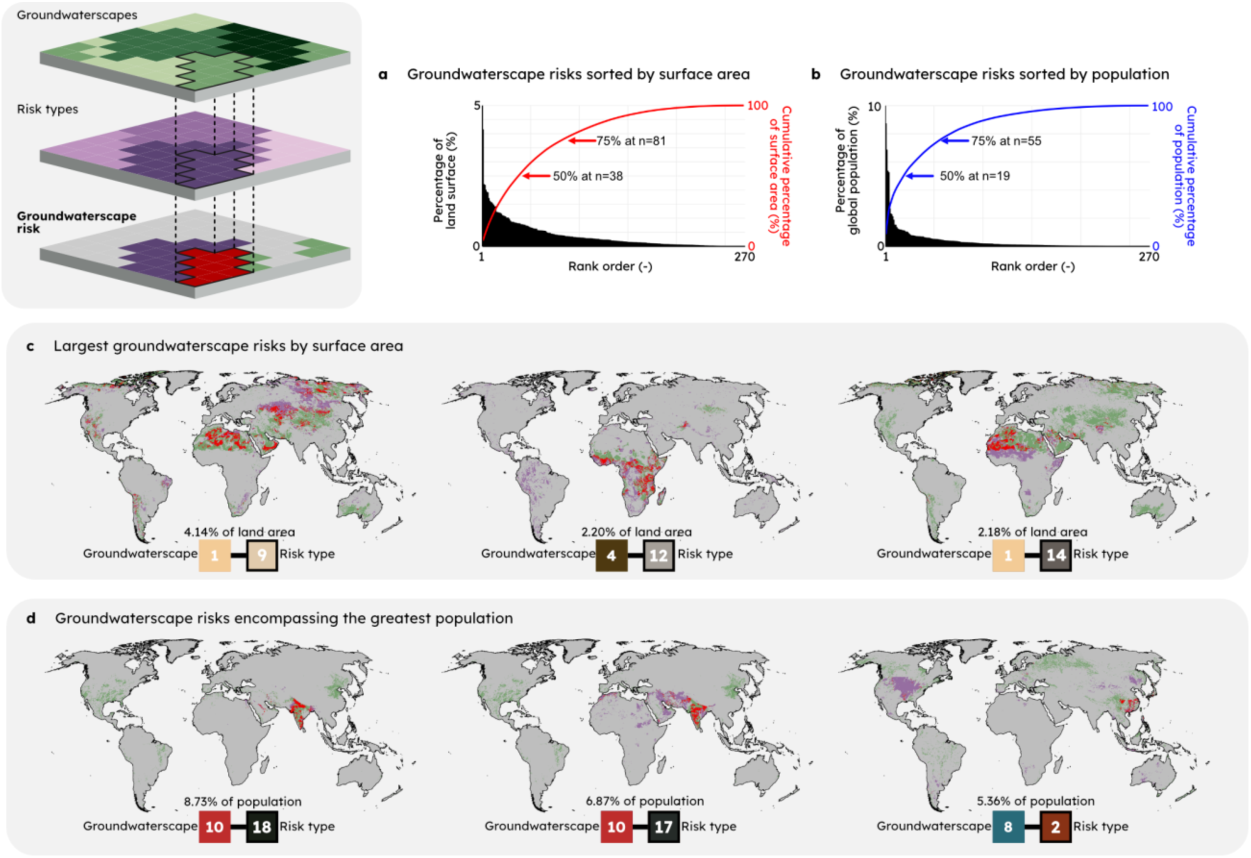


Figure V.3. Groundwaterscape risk distribution, ranked by encompassed population.

(a) Groundwaterscape risk cumulative area distribution. (b) Groundwaterscape risk cumulative population distribution. (c) Extent of the three largest groundwaterscape risks by area. (d) Extent of the groundwaterscape risks encompassing the largest proportion of the global population.

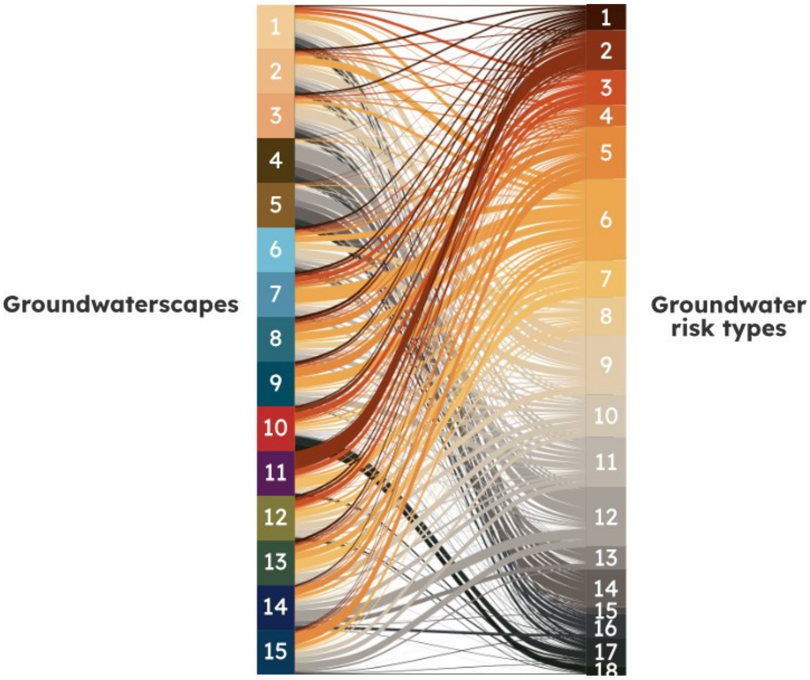


Figure V.4. Alluvial plot revealing the area distribution of groundwater risk types across groundwaterscapes.

<u>Accompanying text:</u>

This figure highlights the uneven distribution of groundwater risk types across groundwaterscapes. The plot is normalized by groundwaterscape area, which is why the thickness of individual groundwaterscapes (left side) is uniform while the thickness of groundwater risk types (right side) is variable (i.e., reflect the risk type area distribution).

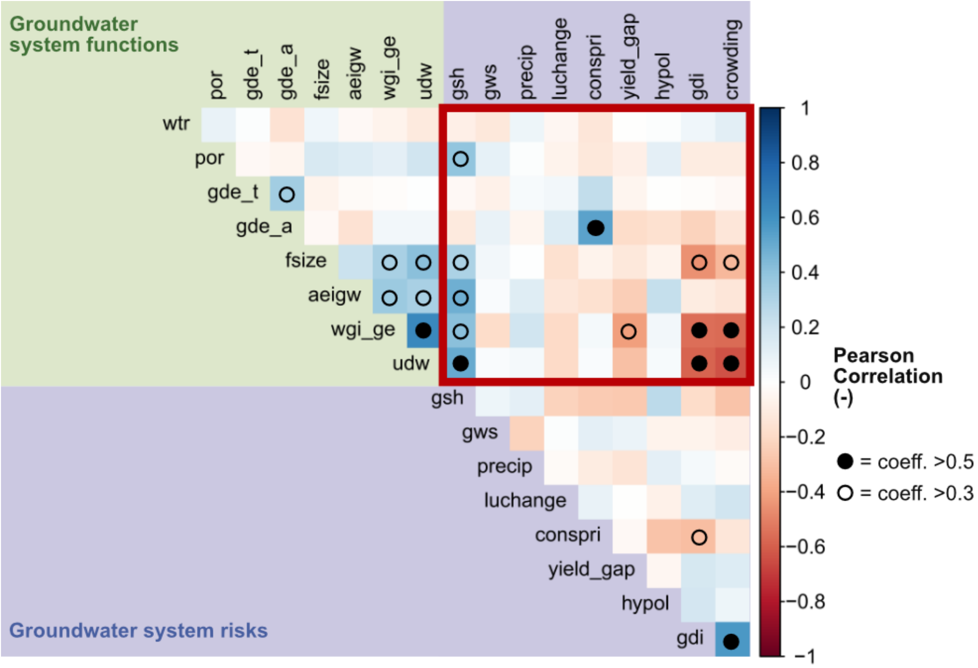


Figure V.5. Correlation coefficients between individual input datasets across both groundwaterscape and groundwater system risk type classification studies.

<u>Accompanying text:</u>

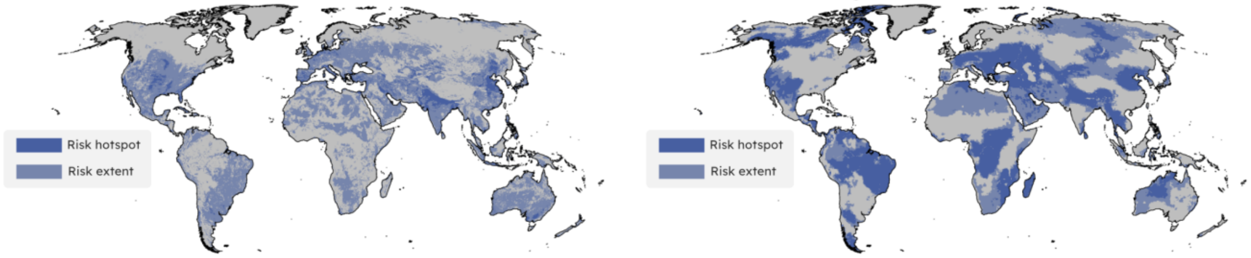
This figure shows the correlation between individual groundwater system functions (green) and groundwater system risks (purple). Highlighted in the red box are the correlations between the system functions and risks. Pairwise correlations above $|0.5|$ are annotated by a full circle while correlations above $|0.3|$ are annotated with a hollow circle. Please see text below for interpretation of short-form names included in the plot:

<u>Groundwater system functions</u>

wtr: Water table ratio
por: Storage capacity
gde_t: Terrestrial GDE area density
gde_a: Aquatic GDE area density
fsize: Field size
aeigw: Groundwater irrigation area density
wgi_ge: Government effectiveness
udw: Unimproved/no drinking water

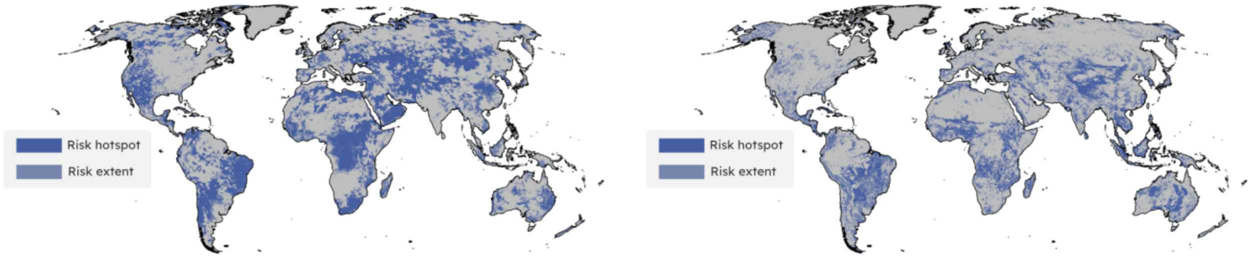
<u>Groundwater system risks</u>

gsh: Threat of land subsidence
gsw: Groundwater storage loss
precip: Change in heavy rainfall events
luchange: Land use change
conspri: Conservation priority needs
yield_gap: Yield gap
hypol: Hydro-political tension
gdi: Gender development inequality
crowding: Change in water crowding



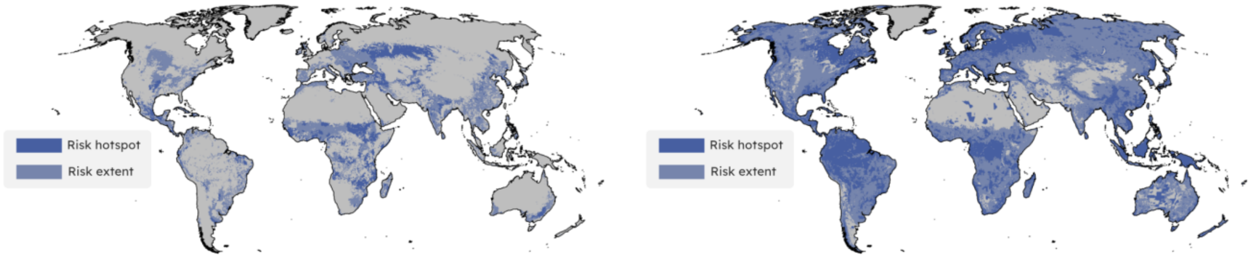
a) Land subsidence

b) Groundwater storage loss



c) Intensifying extreme precipitation

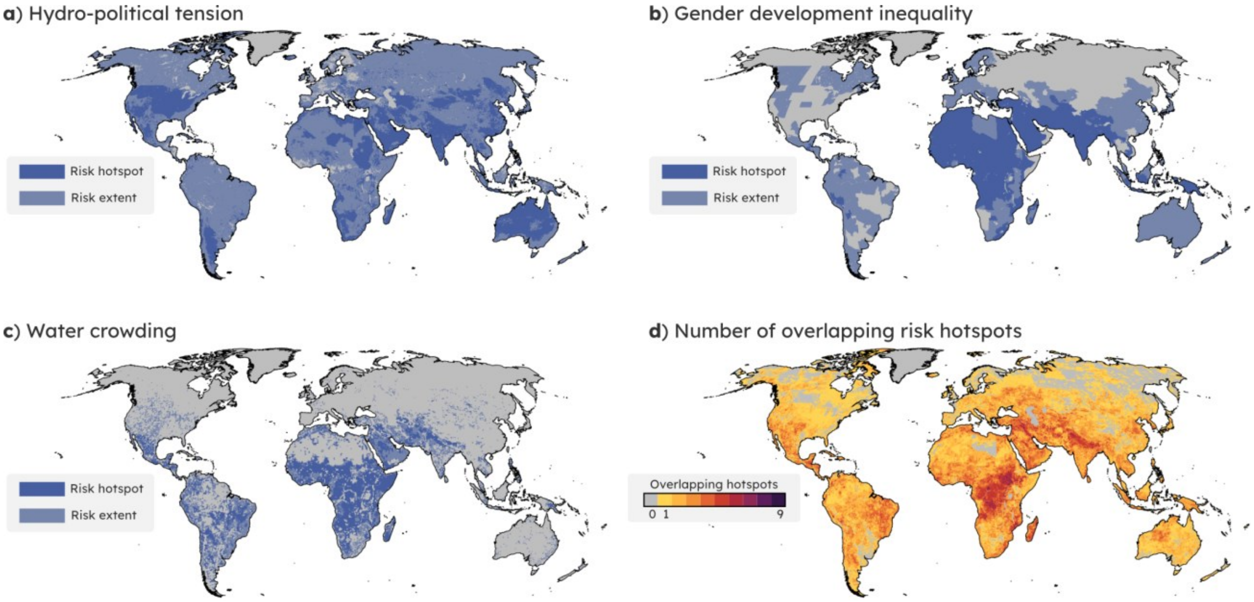
d) Land use change



e) Agricultural intensification

f) Ecological conservation needs

Figure V.6. Enlarged plots of individual groundwater system risks. [plot 1/2]



a) Hydro-political tension

b) Gender development inequality

c) Water crowding

d) Number of overlapping risk hotspots

Figure V.7. Enlarged plots of individual groundwater system risks. [plot 2/2]

V.2 Supplementary tables

Table V.1. Data sources and additional metadata.

Dataset	
Groundwater storage trends	**Data Source**: Herrera-García et al. (2021) **Persistent web link**: http://doi.org/10.6084/m9.figshare.13312070.v1 **Temporal Range**: 2010 **Spatial Resolution**: 30 arcsecond **Resolution Harmonization Method**: Maximum value aggregation
Groundwater storage trends	**Data Source**: Li et al. (2019) **Persistent web link**: doi.org/10.5067/IIU5JWU2AGRP **Temporal Range**: 2003-2021 **Spatial Resolu**tion: 0.25 **Resolution Harmonization Method**: Nearest neighbour resampling
Change in extreme precipitation	**Data Source**: Beck et al. (2019) **Persistent web link**: https://www.gloh2o.org/mswep/ **Temporal Range**: 1980–1989 relative to 2010-2019 **Spatial Resolution**: 0.1 decimal degrees **Resolution Harmonization Method**: Nearest neighbour resampling
Land use change	**Data Source**: Winkler et al. (2021) **Persistent web link**: https://doi.org/10.1594/PANGAEA.921846 **Temporal Range**: Land use in 2015 relative to 1960 **Spatial Resolution**: 0.01 **Resolution Harmonization Method**: Area-weighted fractions **Additional Preprocessing**: Urban, cropland, and pasture/rangeland calculated as proportion to total grid cell area. Thus forest, unmanaged grass/shrubland, sparse/no vegetation, and water land uses not considered.
Agricultural intensification	**Data Source**: Gerber et al. (2024) **Persistent web link**: https://doi.org/10.5281/zenodo.10234041 **Temporal Range**: circa 2010 **Spatial Resolution**: 5 arcminute **Resolution Harmonization Method**: None
Unmet conservation needs	**Data Source**: Jung et al. (2021) **Persistent web link**: https://doi.org/10.5281/zenodo.5006332 **Temporal Range**: Circa. present day **Spatial Resolution**: 10 km (Mollweide projection) **Resolution Harmonization Method**: Bilinear resampling
Hydropolitical conflict	**Data Source**: Farinosi et al. (2018) **Persistent web link**: Provided by author.

	Temporal Range: 1997–2012 **Spatial Resolution**: 0.25 **Resolution Harmonization Method**: Nearest neighbour resampling
Gender development inequalities	**Data Source**: Global Data Lab (Smits & Permanyer, 2019) **Persistent web link**: https://globaldatalab.org/shdi/ **Temporal Range**: 2020 **Spatial Resolution**: Subnational regions **Resolution Harmonization Method**: Rasterization at 5 arcminute
Change in water crowding (population)	**Data Source**: Wang et al. (2022) **Persistent web link**: https://doi.org/10.6084/m9.figshare.19608594.v2 **Temporal Range**: 2020, 2050 (SSP3) **Spatial Resolution**: 30 arcsecond **Resolution Harmonization Method**: Summation aggregation

V.3 Supplementary references

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Smits, J., & Permanyer, I. (2019). The Subnational Human Development Database. *Scientific Data*, *6*(1), 190038. https://doi.org/10.1038/sdata.2019.38

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Winkler, K., Fuchs, R., Rounsevell, M., & Herold, M. (2021). Global land use changes are four times greater than previously estimated. *Nature Communications*, *12*(1), 2501. https://doi.org/10.1038/s41467-021-22702-2

Appendix VI

SUPPLEMENTARY INFORMATION FOR CHAPTER 6

VI.1 Overview of general study approach

Conceptualizing and quantifying interactions between humans, ecosystems, and hydrological processes are critical to broad sustainability efforts across all scales (Xu et al., 2021; Falkenmark et al., 2021; Gleeson et al., 2020). To date, however, very few studies consider the complex interplay between humans, ecosystems, and hydrology at the global scale. To our knowledge, only one global study (Varis et al., 2019) integrates social, ecological, and hydrological considerations in a broad resilience framework. While the study of Varis et al. is highly relevant, and we implement their social adaptive capacity dataset in our study, our approach is specific to the potential impacts from co-occurring freshwater stress and storage loss on social and ecological systems. In contrast, Varis et al. evaluate the general ability of social adaptive capacity to offset broadly defined ecological vulnerability.

Motivated by the conceptual template of biodiversity hotspots (Whittaker et al., 2005), we sought to apply social-ecological system principles to identify the basins most vulnerable to freshwater stress and freshwater storage loss in hopes of steering policy agendas and scientific focus to the identified basins. As process-based knowledge of human-water system interactions at the global scale is characterized by deep uncertainty, we performed a parsimonious, spatial analysis to quantify basin vulnerability to freshwater stress and storage loss based on general social-ecological system principles.

We provide a conceptual model of our study in Figure VI.1, which visualizes the relationships of all elements considered in our methodology and discussion. These relationships are not intended to be exhaustive; rather, they are shown to clarify the core conceptual underpinnings of our study.

Noting the intertwined relationships between vulnerability, resilience, and adaptive capacity (Lei et al., 2014), we find it important to clarify our terminology and specifically why we describe this study as a vulnerability analysis. Whereas vulnerability and adaptive capacity can be generally considered as system states, resilience is a more dynamic property with an inherent temporal dimension. More simply, resilience and vulnerability may be best approached through dynamic and static frames of analysis, respectively, and are linked through the concept of adaptability (Lei et ai., 2014). As global social-ecological data is not readily available at sufficient temporal resolution or for particular processes that would enable resilience considerations to be addressed, we performed a static vulnerability analysis. Resilience concepts were invoked in our discussion, however, through the feedback mechanisms shown in the conceptual model. We dedicate the remainder of this section to providing addition details on the conceptual model in a zonal approach as specified by Figure VI.2.

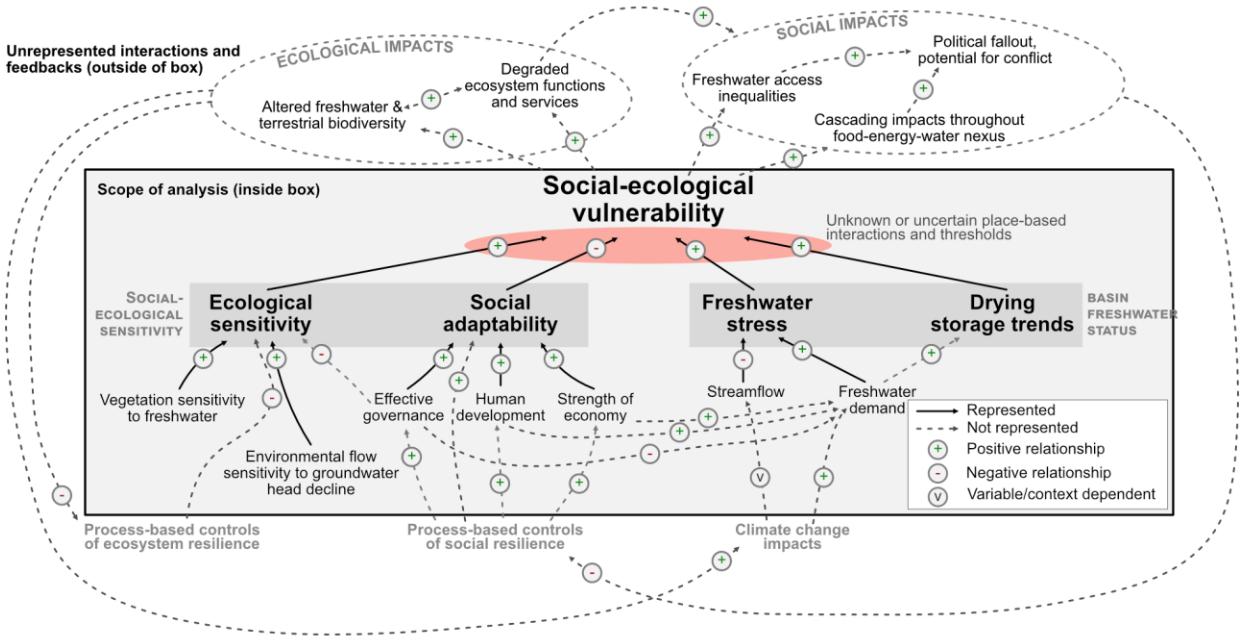


Figure VI.1. Conceptual model of the study.

Core variables and concepts incorporated in this study and our interpretation of the relationships between them. Elements represented in our analysis are shown by solid lines and are found within the black-outlined box (denoted by the ‘scope of analysis’ label). Elements and relationships that were not represented in our analysis but were discussed in the main text are found outside of this box and/or are shown by dashed lines. Positive (negative) relationships between two variables (first variable → second variable) represent relationships where we consider an increase in the first variable to lead to an increase (decrease) in the second variable.

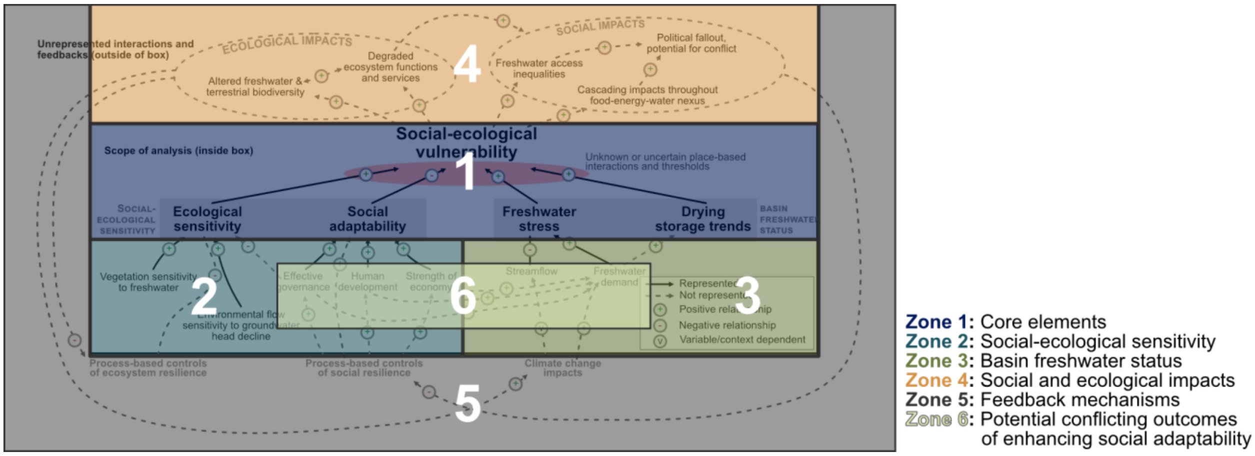


Figure VI.2. Zones of the conceptual model receiving supplemental description, individually.

VI.1.1 Zone 1: Core elements

Conceptually, we orient our vulnerability analysis around the broad definition of Turner et al. (2003) where "vulnerability is the degree to which a system … is likely to experience harm due to exposure to a hazard, either a perturbation or stress/stressor", which can be represented by the combined consideration of exposure to a hazard, system sensitivity, and adaptive capacity (Lei et al., 2014). The concept of sensitivity has many potential meanings in this study: it can imply the sensitivity of flow regimes to changes in freshwater storage, the sensitivity of ecosystems to changes in streamflow or in freshwater storage, or the sensitivity of human activities and well-being to hydrological perturbations. While we do not consider this first interpretation of sensitivity, it can be partially inferred through our combination of freshwater stress and storage trends. We do consider the latter two interpretations, which we refer to together as social-ecological sensitivity. Thus, we interpret sensitivity and adaptive capacity concepts from ecohydrological and sociohydrological perspectives, respectively, and not from a strictly hydrological perspective. (Note that this discussion on system sensitivity should not be confused with our sensitivity analysis in Section VI.4.)

Guiding our governing equation (equation 1 in Section 6.3), we considered social-ecological vulnerability to be a function of basin freshwater status (see Zone 3 description) and social-ecological sensitivity. While the input indicators for basin freshwater status and social-ecological sensitivity (i.e. freshwater stress, storage loss, ecological sensitivity, and social adaptability) all affect social-ecological vulnerability, specific threshold relationships between these four variables have yet to be identified or sufficiently studied.

VI.1.2 Zone 2: Social-ecological sensitivity

In our derivation of ecological sensitivity, we combined two global datasets that consider separate facets of ecological sensitivity to freshwater stress and storage trends: (1) vegetation sensitivity to anomalies in soil moisture and shallow groundwater storage, and (2) environmental flow sensitivity to simulated groundwater head decline. There are many other processes that contribute to ecosystem-wide sensitivity to freshwater stress and storage trends, though we are not aware of any others that have been considered in global analyses with accompanying open data products. For instance, there remains no global mapping of groundwater dependent ecosystems, or global analyses of place-based aquatic and terrestrial biodiversity sensitivity to environmental flow transgressions.

Our representation of social adaptive capacity was derived by Varis et al. (2019), who understood social adaptive capacity to represent the broad ability of the "social part of social-ecological systems" to both reactively and proactively adapt and increase resilience to ecological vulnerabilities. Varis et al. derived adaptive capacity from three input indicators: government effectiveness, strength of economy, and human development. We note that this is a relatively parsimonious conceptualization of adaptive capacity which may be challenged to represent critical dynamic properties of the social system, such as its absorbability or transformability (Mao et al., 2017).

VI.1.3 Zone 3: Basin freshwater status

We derived the basin freshwater status indicator to compress the bivariate relationship between freshwater stress and freshwater storage trends into a single dimension, which we discuss and describe in the main text and Methods. As freshwater demand is attributed as a driver of large-scale drying in several mid-latitude regions globally (Rodell et al., 2018), we link freshwater demand to drying trends in the conceptual model. However, as the storage trend data we used were derived from satellite observations (see Table VI.1) and not estimated from hydrological model with human activity representations, this relationship is shown by a dashed (i.e. unrepresented) line in the conceptual model.

VI.1.4 Zone 4: Social and ecological impacts

Despite the aforementioned knowledge gap on processes and thresholds of large-scale social-ecological systems in the context of freshwater stress and storage loss, our conceptual model assumes that increases in our derived indicator of social-ecological vulnerability leads to a greater likelihood that social and ecological impacts are experienced within the system. All ecological and social impacts shown in the conceptual model are discussed in the main text.

VI.1.5 Zone 5: Feedback mechanisms

Resilience is represented by the feedback mechanisms shown in the conceptual model that act on the underlying process-based controls of ecosystem and social functions. Ecological impacts, as identified in Zone 4, reduce ecosystem resilience by decreasing system diversity, connectivity, and altering 'slow variables' that regulate ecosystem functioning (Biggs et al., 2012). Ecological impacts can also feed back to affect freshwater stress and storage trends through altered ecosystem services, climate dynamics, and their interactions. Where regulating ecosystem services are degraded, the impacts of climate change may be exacerbated which may result in increasing rates of storage loss, increasing rates of freshwater demand, and decreasing or more-variable streamflow rates (Dai et al., 2016). Social impacts of freshwater stress and storage loss can feed back and deteriorate process-based controls of social resilience, though we note that literature is sparse on cascading impacts of hydrological hazards in large-scale social-ecological systems.

VI.1.6 Zone 6: Potential for conflicting outcomes when enhancing social adaptability

Considering the unique role of humans in driving the vulnerability of the social-ecological system, we identify a certain tension between the input variables used to derive social adaptive capacity and their impact on freshwater stress and storage trends. For example, human development and economic strength are identified as positive contributors to social adaptive capacity yet both are commonly associated with increases in freshwater demand that drive freshwater stress and storage loss. Thus, human development and economic growth may produce an offsetting effect on social-ecological vulnerability by simultaneously increasing freshwater stress and storage loss. There exists, therefore, a need for effective governance to balance these impacts, such as through linking Sustainable Development Goals, as we discuss in the Chapter 5.

VI.1.7 Limitations of the conceptual model and approach

Several important processes and considerations are absent from our conceptual model. These include the absence of global trade considerations, or the role of cross-basin water transfer infrastructure projects. We also performed a 'lumped' analysis of social-ecological vulnerability at the basin scale and thus sub-basin variability of all input variables was not considered. Thus, basins with particularly large variance in input parameters may be poorly represented by the basin average.

VI.2 Data selection and preprocessing

All data used in this study are described and justified in Table VI.1.

Table VI.1. Data sources, description, justification, and summary of any preprocessing applied.

Dataset	
Freshwater withdrawal and consumption rates	**Data Source:** Huang et al. (2018) **Persistent Web Link:** https://doi.org/10.5281/zenodo.1209296 **Temporal Range:** 1971 – 2010, monthly. **Spatial Resolution:** 0.5° **Resolution Harmonization Method:** N/A **Description and Justification:** Monthly withdrawal and consumption rates for six water use sectors: irrigation, domestic, electricity generation, livestock, mining, and manufacturing. Irrigation estimates are provided from four hydrological models: WaterGAP, H08, LPJmL, and PCR-GLOBWB. Non-irrigation sector estimates are spatially downscaled based on global population and livestock density maps. **Additional Preprocessing:** As this dataset was not available for our desired year of 2015, we aggregated all sectors over the most recent year available, 2010. To reconcile the four alternative irrigation estimates, we calculated the median of the four model alternatives at the individual grid cell and used these rates in addition to the other five sectors in our representation of total freshwater withdrawal and consumption rates. We used withdrawal rates data in our main analyses, but considered the impact of this decision by also deriving hotspot basins using consumption rates in our sensitivity analysis (Section VI.4).

Streamflow	**Data Source:** Global Streamflow Characteristics Dataset (Beck et al., 2013; 2015) and GRUN (Ghiggi et al., 2019) **Persistent Web Link:** http://www.gloh2o.org/gscd/ and https://doi.org/10.6084/m9.figshare.9228176 **Temporal Range:** N/A (reference) and 1902-2014, respectively **Spatial Resolution:** 0.5° **Resolution Harmonization Method:** N/A **Description and Justification:** The Global Streamflow Characteristics Dataset (GSCD) provides spatially-distributed, global maps of 17 streamflow characteristics derived from neural network ensembles trained with observed streamflow records, including mean annual flow (Q_{mean}). The Q_{mean} hydrological signature was trained over the record of available streamflow observations and does not correspond directly with a calendar year. GRUN is an alternate global, gridded streamflow dataset derived through a neural network trained with streamflow observations. Conversely to GSCD, GRUN reconstructs historical runoff at monthly time steps over 1902-2014. We used the GSCD dataset in our main analyses based on its more-frequent use as a reference dataset, and we considered the impact of this decision in our sensitivity analysis. **Additional Preprocessing:** We estimated Q_{mean} from GRUN, for use in the sensitivity analysis, by calculating average annual streamflow over 2000-2010.
Freshwater storage trends	**Data Source:** Rodell et al. (2018) **Persistent Web Link:** https://doi.org/10.1038/s41586-018-0123-1 **Temporal Range:** April 2002 – March 2016. **Spatial Resolution:** 0.5° **Resolution Harmonization Method:** N/A **Description and Justification:** Preprocessed annual trends in terrestrial water storage (TWS) from the Gravity Recovery and Climate Experiment (GRACE) satellite mission. TWS trends represent annual trends in combined groundwater, soil moisture, surface water, canopy water, ice and snow storages. GRACE observations are the only observation-based dataset of trends in global freshwater storage. While the ongoing GRACE Follow-On (GRACE-FO) Mission provides data to update the GRACE trends to present, they are not used in our study as they exceed our target year of 2015. **Additional Preprocessing:** N/A
Vegetation sensitivity	**Data Source:** Seddon et al. (2016) **Persistent Web Link:** https://doi.org/10.5287/bodleian:VY2PeyGX4 **Temporal Range:** 2000-2013 **Spatial Resolution:** 0.05°

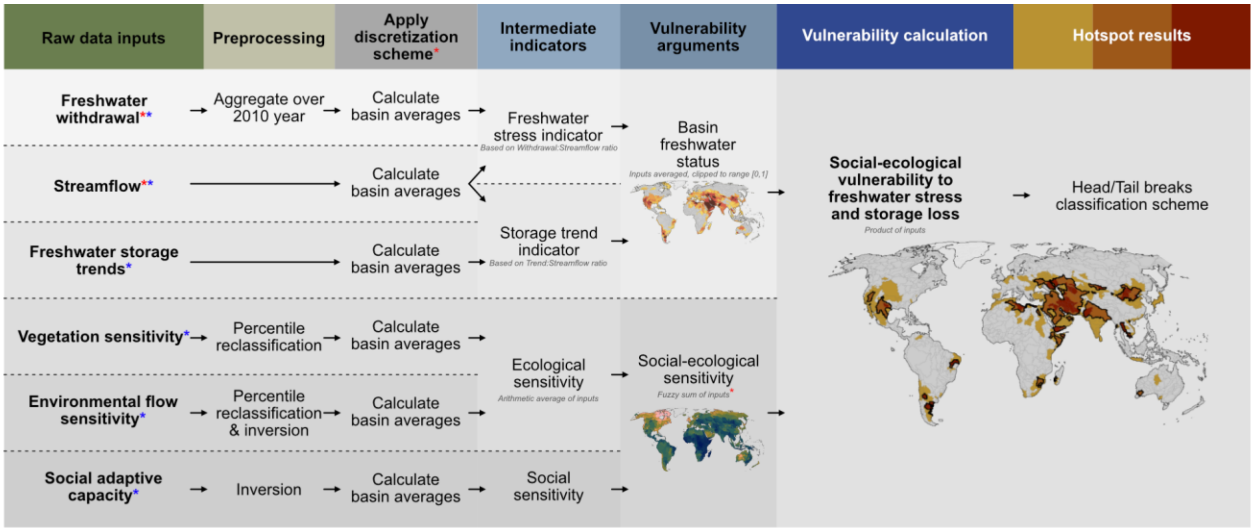
	Resolution Harmonization Method: Average resampling to 0.5° (arithmetic mean of all contributing raw pixels, n = 100) **Description and Justification:** Seddon et al. derived a vegetation sensitivity index (VSI) to identify ecosystem sensitivity to three climate variables: air temperature, water availability, and cloud cover. The VSI is derived from a seasonally detrended time series of the enhanced vegetation index (EVI) and the three climate variables. The VSI is calculated as the $\log_{10}$-transformed ratio of EVI to climate variance weighted based on the relative importance of each climate variable. The ratio of actual evapotranspiration to potential evapotranspiration is used as the proxy indicator of water availability. As the VSI is a composite indicator across the three climate variables, we used only the contribution of water availability to the overall VSI. **Additional Preprocessing:** N/A
Environmental flow sensitivity	**Data Source:** de Graaf et al. (2019) **Persistent Web Link:** https://doi.org/10.5683/SP2/D7I7CC **Temporal Range:** 1960-2100 **Spatial Resolution:** 0.0833° (5 arcminute) **Resolution Harmonization Method:** Average resampling to 0.5° (arithmetic mean of all contributing raw pixels, n = 36) **Description and Justification:** de Graaf et al. modelled the groundwater head decline at which a presumptive environmental flow limit is transgressed for all basins where groundwater pumping occurs. The model used in the study was PCR-GLOBWB coupled to a two-layer groundwater model that simulated lateral flow[13]. To date, no other global study has quantified the sensitivity of environmental flow transgression to groundwater pumping. **Additional Preprocessing:** N/A
Social adaptive capacity	**Data Source:** Varis et al. (2019) **Persistent Web Link:** https://doi.org/10.5061/dryad.h2v2398 **Temporal Range:** 2015 **Spatial Resolution:** 0.0833° (5 arcminute) **Resolution Harmonization Method:** Average resampling to 0.5° (arithmetic mean of all contributing raw pixels, n = 36) **Description and Justification:** Varis et al. derived an indicator of social adaptive capacity through an equal-weighted composite of governance, economic (GDP per capita) and human development indicators. To our knowledge, no other spatially distributed dataset exists that represents the general capacity of the social system to respond to broad environmental disturbances. **Additional Preprocessing:** N/A

Population count	**Data Source:** Gridded Population of the World – United Nations World Population Prospects version 4 (GPW UN-WPP Adjusted v4.11) (CIESIN, 2018; Doxsey-Whitfield et al., 2015) **Persistent Web Link:** https://doi.org/10.7927/H4PN93PB **Temporal Range:** 2015 **Spatial Resolution:** 0.5° **Resolution Harmonization Method:** N/A **Description and Justification:** Global gridded population count adjusted to match UN-WPP country totals. This dataset is the recommended GPW dataset for global analyses. This dataset is interchangeable with the Global Human Settlement Population (GHS-POP) dataset at coarse resolutions (i.e. 0.5°) as GHS-POP is a further downscaled product of GPW. **Additional Preprocessing:** N/A
Food crop production	**Data Source:** Kummu et al. (2021), IFPRI (2019) **Persistent Web Link:** N/A see code repository associated with above reference: https://github.com/matheino/holdridge **Temporal Range:** 2010 **Spatial Resolution:** 0.0833° (5 arcminute) **Resolution Harmonization Method:** Aggregated (summed) to 0.5° **Description and Justification:** Kummu et al. (2021) converted crop production data from the SPAM dataset into combined kilocalories for 27 major food crops for the year 2010. While more recent gridded agricultural products exist, none have been converted and combined into total kilocalorie production and would require additional processing. Thus, the Kummu et al. dataset represents the most ready-for-use dataset for our purposes, despite not being aligned with our desired year of 2015. **Additional Preprocessing:** N/A
Gross Domestic Product	**Data Source:** Kummu et al. (2018; 2019) **Persistent Web Link:** https://doi.org/10.5061/dryad.dk1j0 **Temporal Range:** 2015 **Spatial Resolution:** 0.00833° (30 arcsecond) **Resolution Harmonization Method:** Aggregated (summed) to 0.5° **Description and Justification:** Gross domestic product (GDP) in 2011 international USD. While an alternative UNEP/GRID Geneva gridded 2010 GDP dataset exists, Kummu et al.'s alternative is more aligned with FAIR (findable, accessible, interoperable, and reusable) data principles as well as with our desired year of 2015. **Additional Preprocessing:** N/A

Amphibian species richness	**Data Source:** IUCN amphibian richness grids (IUCN, 2015) **Persistent Web Link:** https://doi.org/10.7927/H4RR1W66 **Temporal Range:** 2015 **Spatial Resolution:** 0.00833° (30 arcsecond) **Resolution Harmonization Method:** Maximum resampling to 0.5° (maximum of contributing raw pixels, n = 60) **Description and Justification:** The number of amphibian species present per grid cell. We opted for these richness grids rather than the Global Freshwater Biodiversity Atlas as they are globally available in distributed grid format rather than at the basin scale with sub-global coverage as is provided by the Biodiversity Atlas. We selected amphibian species richness grids rather than other taxa following Tisseuil et al. (2013) who recommend that amphibians be used as a surrogate candidate for global freshwater conservation planning. **Additional Preprocessing:** N/A
Priority wetlands for conservation	**Data Source:** Ramsar List of wetlands of international importance (Ramsar Convention Secretariat, 2013) **Persistent Web Link:** https://rsis.ramsar.org/ **Temporal Range:** Last accessed 20 March 2021 **Spatial Resolution:** Point data **Resolution Harmonization Method:** N/A **Description and Justification:** A list of wetlands of international importance "on account of their international significance in terms of ecology, botany, zoology, limnology, or hydrology" (Ramsar Convention Secretariat, 2013). which must satisfy at least one of nine criteria. There are currently over 2,400 Ramsar Sites. **Additional Preprocessing:** N/A
Integrated water resources management implementation	**Data Source:** IWRM Data Portal (UNEP-DHI, 2021) **Persistent Web Link:** http://iwrmdataportal.unepdhi.org/countrydatabase **Temporal Range:** 2017, 2020 **Spatial Resolution:** National boundaries **Resolution Harmonization Method:** Minimum value polygon rasterization to 0.5° **Description and Justification:** The IWRM Data Portal records and presents "the global status and progress on SDG 6.5.1", which is "[the] degree of integrated water resources implementation". The Data Portal tracks SDG 6.5.1 progress through 33 indicators, across four IWRM components: an enabling environment, institutions and participation, management instruments, and financing. Baseline data is provided for the year 2017 and an updated dataset is provided for the year 2020.

	Additional Preprocessing: We primarily used the more recent (2020) dataset. However, we filled any gaps in the 2020 dataset if nations received IWRM scores in 2017 but did not participate in the 2020 update.
Basins	**Data Source:** HydroBASINS (Lehner & Grill, 2013; 2014). **Persistent Web Link:** https://www.hydrosheds.org/page/hydrobasins **Temporal Resolution:** N/A **Spatial Resolution:** 15-arcsecond **Resolution Harmonization Method:** Borders simplified to a 0.5° grid. **Description and Justification:** HydroBASINS are the global standard in basin discretization schemes. We used HydroBASINS level 4 in our main analyses, however we considered the impact of this decision by including levels 3 and 5 in our sensitivity analysis. **Additional Preprocessing:** N/A

VI.3 Supplementary methodology figures



★ Considered in uncertainty analysis
★ Considered in sensitivity analysis

Figure VI.3. Overview schematic of the hotspot basin derivation process.

To be viewed from left to right: (i) raw data inputs were preprocessed if necessary, then (ii) these inputs were summarized to the selected basin scheme, (iii) combined into intermediate indicators, and subsequently (iv) combined into vulnerability function arguments. The final vulnerability dataset, a product of both vulnerability arguments, was classified into vulnerability classes and hotspot basins using the Head/Tail breaks classification scheme. Data resolution harmonization methods are listed in Table VI.1 and are not shown in the 'preprocessing' column. Blue asterisks indicate the variables we considered in our uncertainty analysis (Figures VI.8 and VI.9). Red asterisks indicate the methodological elements we considered in our sensitivity analysis (Figure VI.10).

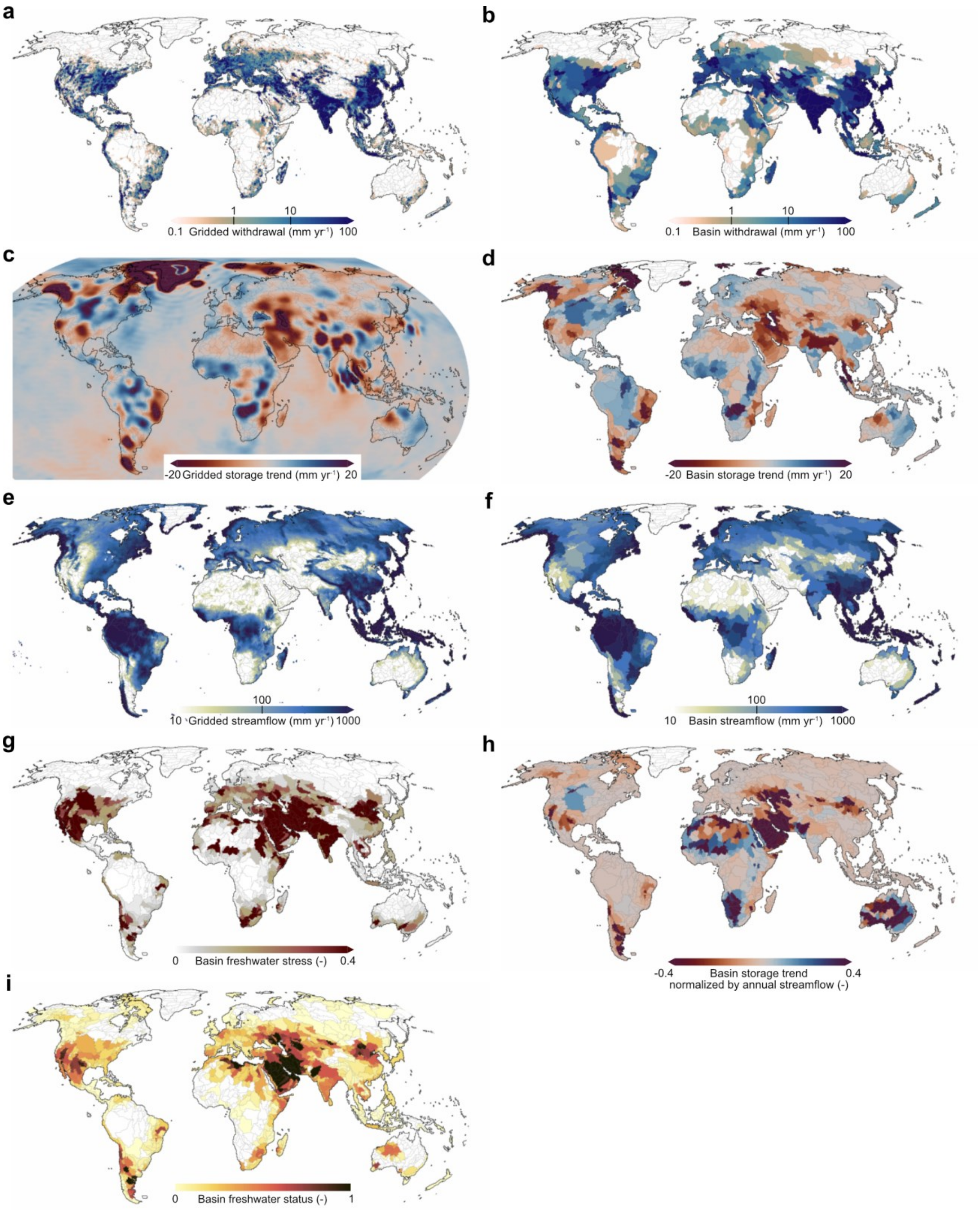
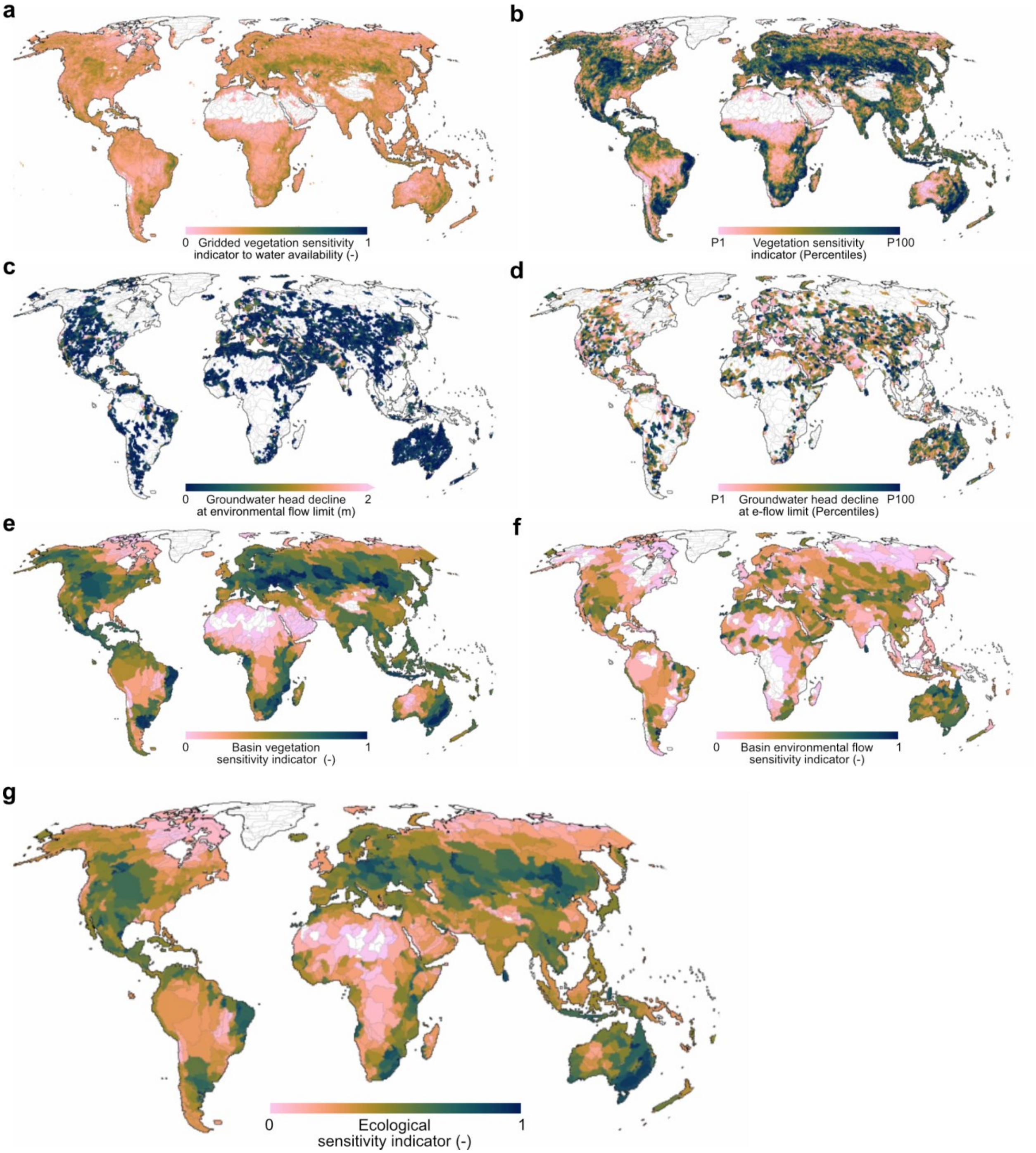


Figure VI.4. Data inputs and derivation steps of basin freshwater status.

(a) Gridded annual freshwater withdrawals. (b) Average annual freshwater withdrawals per basin. (c) Gridded trends in freshwater storage. (d) Trends in freshwater storage per basin. (e) Gridded annual streamflow. (f) Annual streamflow per basin. (g) Basin freshwater stress, calculated as withdrawals divided by streamflow. (h) Basin freshwater storage trend normalized by streamflow, calculated as storage trends divided by streamflow. (i) Basin freshwater status, derived from normalized inputs of panels (g) and (h), as described in the Methods.

Figure VI.5. Data inputs and derivation of the ecological sensitivity indicator.

(a) Vegetation sensitivity to water anomalies from the Vegetation Sensitivity Index (Seddon et al., 2016). (b) Percentile reclassification of panel (a). (c) Groundwater head decline at which environmental flow limits are transgressed (de Graaf et al., 2019). (d) Inverted percentile reclassification of panel (c). This dataset was inverted as threshold transgressions at smaller head declines represent greater sensitivity. (e) Basin average vegetation sensitivity percentiles (shown in b). (f) Basin average environmental flow sensitivity percentiles (shown in d). (g) Combined ecological sensitivity indicator, calculated as the average of maps (e) and (f), and normalised by the maximum basin value.


a
0 Gridded vegetation sensitivity 1
indicator to water availability (-)
b
P1 Vegetation sensitivity P100
indicator (Percentiles)
c
0 Groundwater head decline 2
at environmental flow limit (m)
d
P1 Groundwater head decline P100
at e-flow limit (Percentiles)
e
0 Basin vegetation 1
sensitivity indicator (-)
f
0 Basin environmental flow 1
sensitivity indicator (-)
g
0 Ecological 1
sensitivity indicator (-)

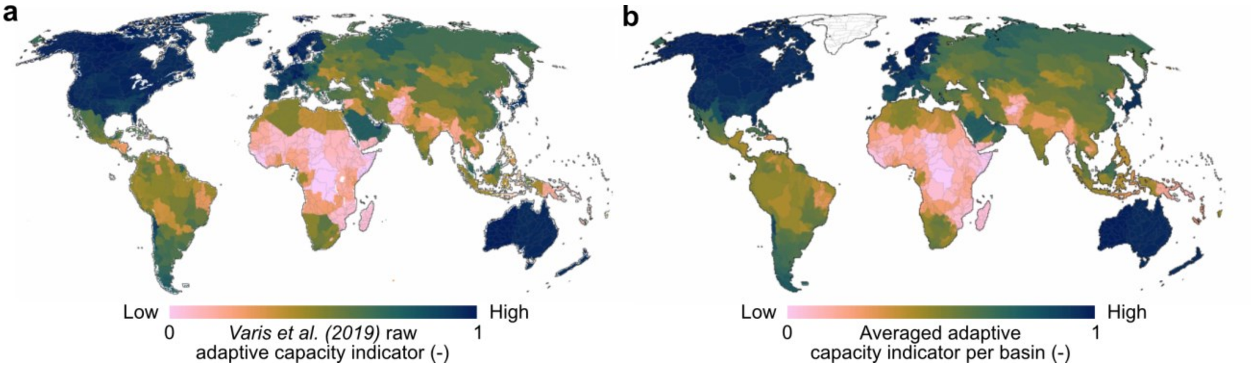


Figure VI.6. Adaptive capacity.

(a) Social adaptive capacity, as derived by Varis et al. (2019). (b) Social adaptive capacity per basin, as used in our analysis.

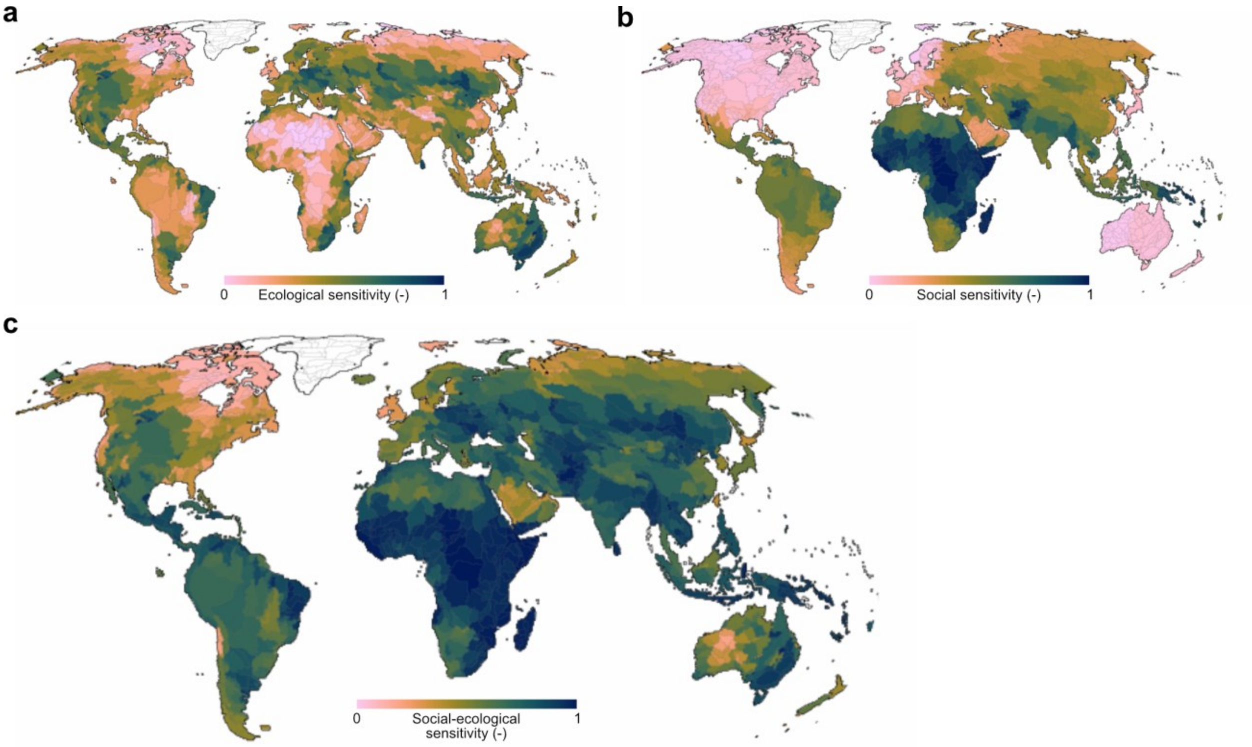


Figure VI.7. Derivation of the social-ecological sensitivity indicator.

(a) Ecological sensitivity indicator (as shown in Figure VI.5g). (b) Social sensitivity indicator (inverted from Figure VI.6b). (c) Social-ecological sensitivity, calculated as the fuzzy sum of (a) and (b).

VI.4 Uncertainty and sensitivity analysis methods and results

As our multidimensional input datasets possess either spatially variable or unquantified uncertainty, we opted for a parsimonious perturbation-based approach to explore the impact of input data uncertainty on hotspot basin results. We considered two forms of data uncertainty: systematic under- or over-estimation of individual parameters (hereafter referred to as spatially uniform uncertainty), and spatially variable uncertainty. We also conducted a separate sensitivity analysis that explores the impact of subjective decisions made in our methodology on our hotspot results.

To explore potential spatially uniform uncertainty, we simultaneously varied the basin-level input data for all six input variables to our hotspot analysis: freshwater withdrawal rates (mm yr^{-1}), streamflow (mm yr^{-1}), storage trends (mm yr^{-1}), social adaptability (-), percentile-reclassified vegetation sensitivity (-), and percentile-reclassified environmental flow sensitivity (-). For each perturbation set, all basins are perturbed by the same magnitude for each variable. The perturbations between variables and between sets were uncorrelated, where the perturbation applied per variable per set was randomly sampled from a normal distribution N($\mu = 0$, $\sigma = 10\%$ of the global σ of the variable). We repeated the hotspot analysis for 10,000 perturbation sets. Figure VI.8 reveals the results of our analysis of potential spatially uniform uncertainty. We found that the number of identified hotspot basins increased as withdrawal rates increased, and storage trends became more negative. We found interesting patterns in streamflow perturbations, where a bimodal distribution was found for negative perturbations while no clear effect was identified for positive perturbations. No trend was visible in perturbations applied to social adaptive capacity, vegetation sensitivity, or environmental flow sensitivity; however bimodal hotspot distributions were found in each. The lack of a trend for these variables likely derives from the percentile-based social-ecological sensitivity approach, as applying globally uniform perturbations across all basins does not change the rank-order of basin sensitivity (social or ecological) and thus yields minimal impact on our analysis. We found hotspot basins to be very consistent throughout this spatially uniform uncertainty analysis. Across the 10,000 perturbation sets, 98% of the transitional or hotspot basins were identified as so in over 50% of the perturbation sets, while 85% of the hotspot basins (exclusively) were identified as so in over 50% of the perturbation sets. Further, only 28 basins were identified as transitional or hotspot basins in over 50% of the perturbation sets but were not identified as so in our main analysis. Only 18 basins were identified as hotspot basins (exclusively) in over 50% of the perturbation sets but were not identified as so in our main analysis.

We found that there was very little impact of systematic under- and/or over-estimation of input variable datasets on our transitional and hotspot basin results, and slightly greater impact on our hotspot basin results. Thus, it appears that the impact of systematic over- or under-estimation of individual variables could minorly impact our differentiation of hotspot basins from transitional basins, but does not significantly impact our differentiation of transitional basins from non-vulnerable basins.

To explore potential spatially variable uncertainty, we similarly varied the basin-level input data for all six input variables simultaneously to our vulnerability analysis. Rather than perturbing all basins by the same magnitude for each variable per perturbation set, we perturbed basins individually based on random sampling from a normal distribution. The perturbations between basins, between variables, and between sets were uncorrelated, as the perturbation applied per basin, per variable, per set was randomly sampled from a normal distribution where we randomly determined the distribution's variance from a uniform distribution, i.e. N(μ = 0, σ ~ U(0, 20% of the global σ of the variable)). We repeated the hotspot analysis for 10,000 perturbation sets. Figure VI.9 reveals the results of our analysis of potential spatially variable uncertainty. We found that the number of identified hotspot basins decreased as the variance in perturbations applied to streamflow rates increased. These spatially random perturbations to all other input variables showed little to no impact on hotspot basin counts. We found our hotspot basins to be very consistent throughout this spatially variable uncertainty analysis. Across the 10,000 perturbation sets, 97% of the transitional or hotspot basins were identified as so in over 50% of the perturbation sets, while 83% of the hotspot basins (exclusively) were identified as so in over 50% of the perturbation sets. Further, only 19 basins were identified as transitional or hotspot basins in over 50% of the perturbation sets but were not identified as so in our main analysis. Only 1 basin was identified as a hotspot basin in over 50% of the perturbation sets but was not identified as so in our main analysis. Thus, we similarly found that potential spatially variable uncertainty in our input data has limited impact on our derived hotspot basins.

We also considered the impact of our subjective methodological decisions on our hotspot basin results. To account for the subjective decisions central to this study, we performed the hotspot derivation process for alternative combinations of basin scale, input data where suitable data alternatives exist, and indicator aggregation methods. While we implemented HydroBASINS level 4 as our basin discretization scheme for our main analysis, we considered here the alternative scales of HydroBASINS level 3 and 5 to address the potential impact of the modifiable areal unit problem on our results (Dark & Bram, 2007). There are also competing definitions of freshwater stress that vary in their consideration of withdrawals versus consumption rates. Another subjective decision we made was in regard to streamflow, as multiple gridded streamflow datasets exist though we used only one in our main analysis. Finally, we aggregated social and ecological sensitivity using the fuzzy sum operator, however arithmetic averaging is the more common approach in indicator-based studies. We thus computed hotspot basins for 24 alternative method configurations: scale (3 alternatives: HydroBASINS level 3, 4, 5), demand (2 alternatives: consumption, withdrawal), streamflow (2 alternatives: Global Streamflow Characteristics Dataset - GSCD, Gridded Runoff - GRUN), and sensitivity aggregation (2 alternatives: fuzzy sum, arithmetic mean). The frequency of individual grid cells being identified as transitional and hotspot basins through these alternative methodologies are shown in Figure VI.10. Similarly to our uncertainty analysis, we found that our subjective methodological decisions had minimal effect on transitional and hotspot basins and slightly more effect on the hotspot basin results. We found the hotspot basins in our main analysis to be identified as transitional or hotspot basins in the majority of the alternative configurations. While some regions were identified as transitional basins in alternative method configurations, they were only identified as so in a small minority of these combinations. Conversely, there were a few hotspot basins in our main analysis (e.g. southwestern Australia) that were only identified as hotspot basins in a minority of the alternative configurations. This finding indicates that the methodological configuration we selected and reported on in the main text was one of the few configurations that lead to these basins being identified as hotspots. However, these basins were still identified as transitional basins in several configurations. This analysis demonstrates one clear effect of subjective decision making on our overall hotspot results.

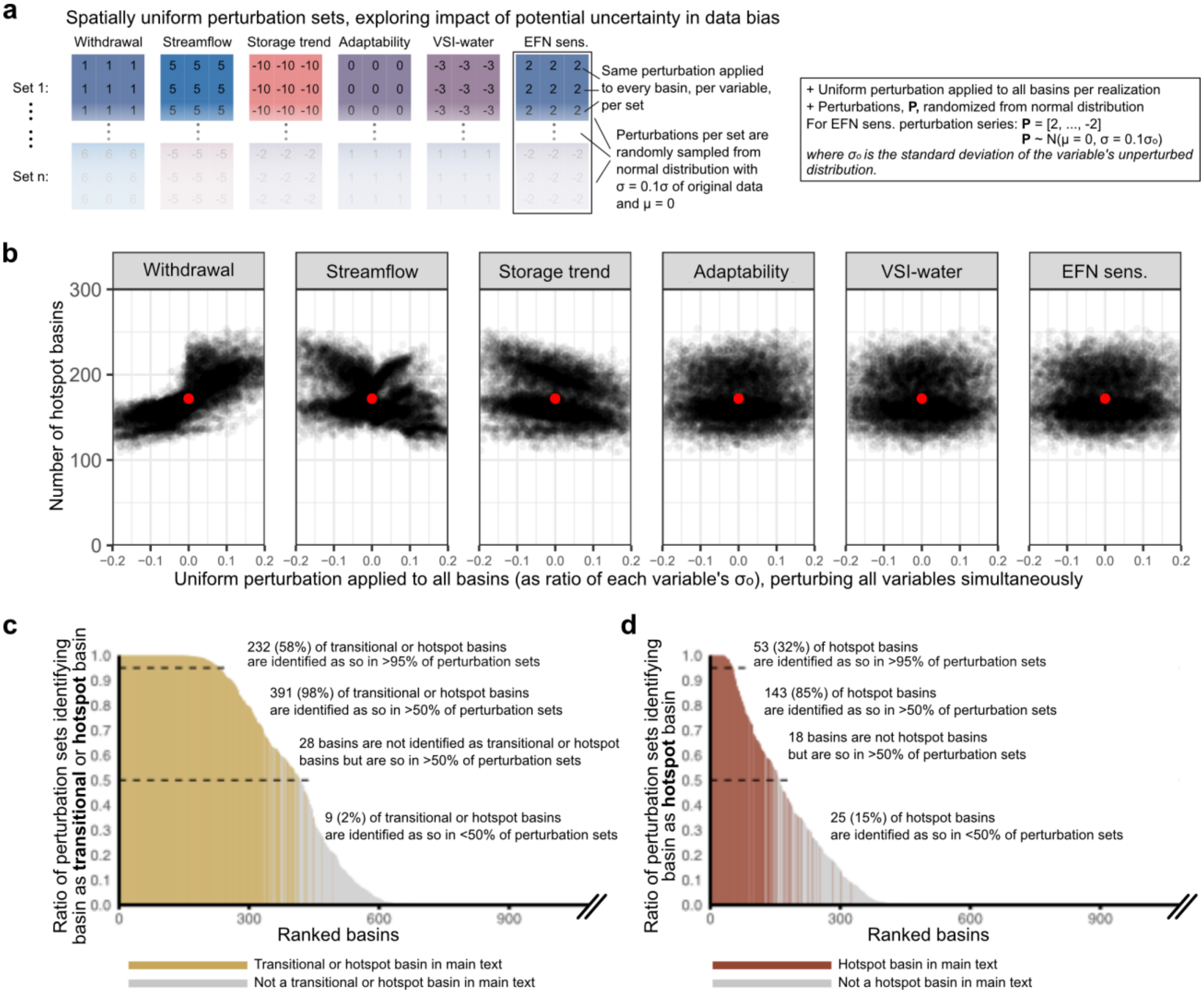


Figure VI.8. Analysis of potential spatially uniform data uncertainty.

(a) Schematic describing the uncertainty analysis approach. (b) Scatterplot of hotspot basin counts from 10,000 random perturbation sets, plotted against individual perturbation magnitudes per variable. The number of hotspot basins identified in the main text is shown by the red circle in each plot. (c) Frequency of basins being identified as either a transitional or hotspot basin over the 10,000 perturbation sets. Transitional or hotspot basins in the main text are coloured yellow. (d) Frequency of basins being identified as a hotspot basin over the 10,000 perturbation sets. Hotspot basins in the main text are coloured red.

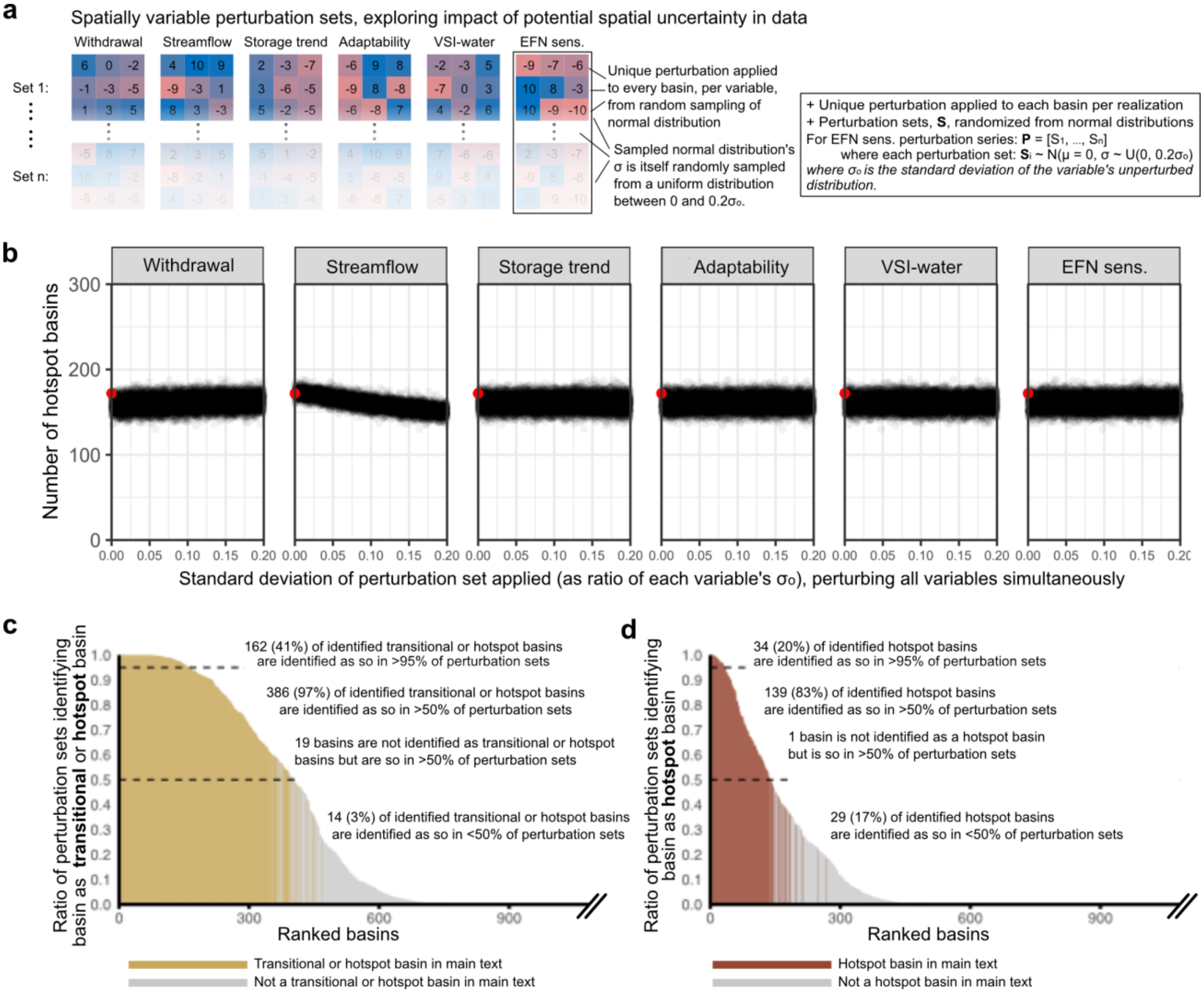


Figure VI.9. Analysis of potential spatially variable data uncertainty.

(a) Schematic describing the uncertainty analysis approach. (b) Scatterplot of hotspot basin counts from 10,000 random perturbation sets, plotted against individual perturbation magnitudes per variable. The number of hotspot basins identified in the main text is shown by the red circle in each plot. (c) Frequency of basins being identified as either a transitional or hotspot basin over the 10,000 perturbation sets. Transitional or hotspot basins in the main text are coloured yellow. (d) Frequency of basins being identified as a hotspot basin over the 10,000 perturbation sets. Hotspot basins in the main text are coloured red.

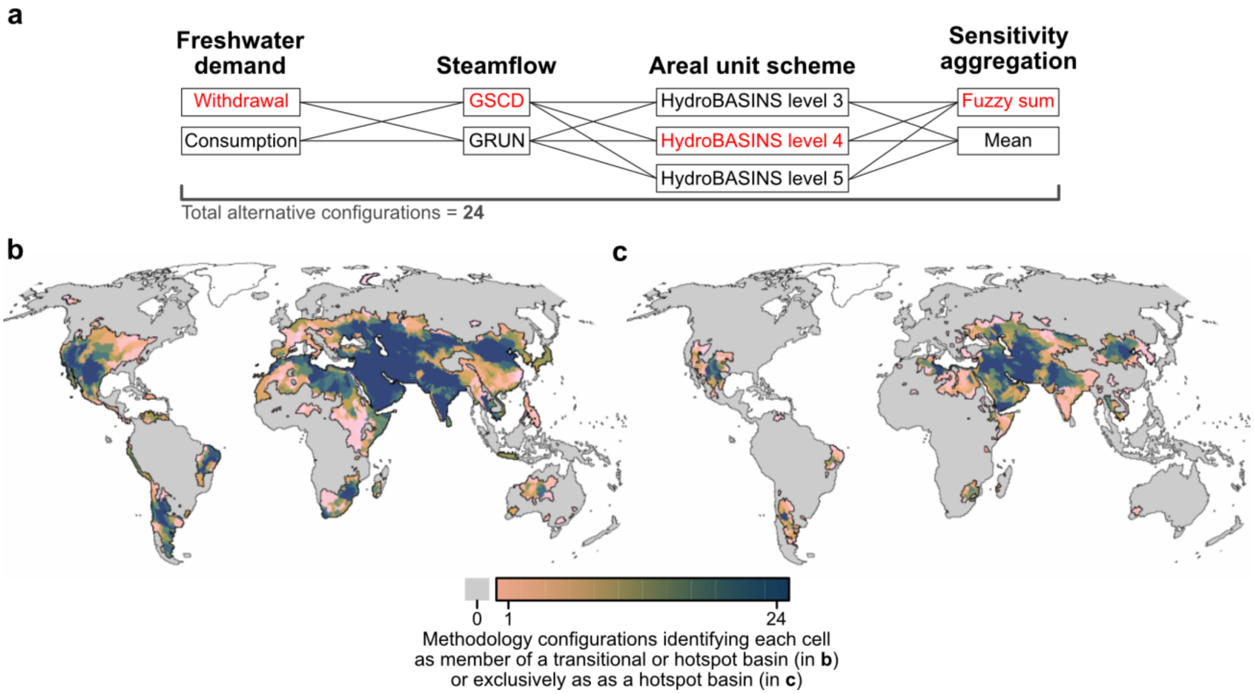


Figure VI.10. The sensitivity of hotspot basin results to subjective aspects of this study's methods.

(a) Schematic of the various methodological configurations evaluated in this sensitivity analysis. The configuration used in the main analysis is shown by the alternatives in red text. (b) The frequency of methodological configurations identifying each grid cell as at least a transitional basin. (c) The frequency of methodological configurations identifying each grid cell as a hotspot.

VI.5 Supplementary results

Table VI.2. Wetlands of international importance (Ramsar Sites) located in hotspot basins.

Ramsar Site No.	Site name	Latitude	Long-itude	Vulnerability class
36	Miankaleh Peninsula, Gorgan Bay and Lapoo-Zaghmarz Ab-bandan	36.83	53.70	Hotspot; High vulnerability
37	Lake Parishan and Dasht-e-Arjan	29.50	52.00	Hotspot; High vulnerability
38	Lake Urmia [or Orumiyeh]	37.50	45.50	Hotspot; High vulnerability
39	Neiriz Lakes & Kamjan Marshes	29.67	53.50	Hotspot; Very high vulnerability
40	Anzali Wetland Complex	37.42	49.47	Hotspot; High vulnerability
41	Shadegan Marshes & mudflats of Khor-al Amaya & Khor Musa	30.50	48.75	Hotspot; High vulnerability

42	Hamun-e-Saberi & Hamun-e-Helmand	31.24	61.25	Hotspot; Very high vulnerability
43	Lake Kobi	36.95	45.50	Hotspot; High vulnerability
44	Hamun-e-Puzak, south end	31.33	61.75	Hotspot; High vulnerability
45	Shurgol, Yadegarlu & Dorgeh Sangi Lakes	37.00	45.50	Hotspot; High vulnerability
46	Bujagh National Park	37.42	49.48	Hotspot; High vulnerability
48	Lake Gori	37.92	46.70	Hotspot; High vulnerability
49	Alagol, Ulmagol and Ajigol Lakes	37.35	54.58	Hotspot; Very high vulnerability
50	Khuran Straits	26.75	55.67	Hotspot; Very high vulnerability
51	Deltas of Rud-e-Shur, Rud-e-Shirin and Rud-e-Minab	27.09	56.75	Hotspot; High vulnerability
52	Deltas of Rud-e-Gaz and Rud-e-Hara	26.67	57.33	Hotspot; High vulnerability
53	Gavkhouni Lake and marshes of the lower Zaindeh Rud	32.33	52.78	Hotspot; Very high vulnerability
99	Kinjhar Lake	24.93	68.05	Hotspot; Very high vulnerability
108	Lakes of the lower Turgay and Irgiz	48.70	62.18	Hotspot; High vulnerability
111	Volga Delta	45.90	48.78	Hotspot; Very high vulnerability
135	Azraq Oasis	31.82	36.80	Hotspot; Very high vulnerability
230	Keoladeo National Park	27.22	77.53	Hotspot; High vulnerability
343	Blesbokspruit	-26.31	28.50	Hotspot; High vulnerability
347	Ash Meadows National Wildlife Refuge	36.42	-116.33	Hotspot; High vulnerability
380	Koshi Tappu	26.65	86.98	Hotspot; High vulnerability
409	Xuan Thuy Natural Wetland Reserve	20.17	106.33	Hotspot; High vulnerability
461	Wular Lake	34.27	74.55	Hotspot; High vulnerability
462	Harike Lake	31.22	75.20	Hotspot; High vulnerability
464	Sambhar Lake	27.00	75.00	Hotspot; High vulnerability

483	Toolibin Lake	-32.92	117.61	Hotspot; High vulnerability
560	Sundarbans Reserved Forest	22.03	89.52	Hotspot; High vulnerability
561	Mühlenberger Loch	53.53	9.80	Hotspot; High vulnerability
620	Lake Sevan	40.26	45.36	Hotspot; High vulnerability
621	Lake Arpi	41.06	43.64	Hotspot; High vulnerability
672	Veselovskoye Reservoir	46.92	41.03	Hotspot; Very high vulnerability
673	Lake Manych-Gudilo	44.60	42.83	Hotspot; Very high vulnerability
734	Área de Protección de Flora y Fauna Cuatrociénegas	26.85	-102.13	Hotspot; High vulnerability
868	Hula Nature Reserve	33.07	35.58	Hotspot; Very high vulnerability
887	Ndumo Game Reserve	-26.88	32.27	Hotspot; High vulnerability
920	Hawar Islands	25.67	50.83	Hotspot; High vulnerability
921	Tubli Bay	26.18	50.57	Hotspot; High vulnerability
952	Nylsvley Nature Reserve	-24.65	28.70	Hotspot; High vulnerability
998	Koh Kapik and Associated Islets	11.47	103.07	Hotspot; Very high vulnerability
1006	Govater Bay and Hur-e-Bahu	25.17	61.50	Hotspot; High vulnerability
1012	Lagunas de Guanacache, Desaguadero y del Bebedero	-33.00	-67.60	Hotspot; Very high vulnerability
1015	Sheedvar Island	26.80	53.40	Hotspot; Very high vulnerability
1026	Ain Elshakika	32.77	21.35	Hotspot; Very high vulnerability
1027	Ain Elzarga	32.78	22.35	Hotspot; Very high vulnerability
1054	Chott Merrouane et Oued Khrouf	33.89	6.18	Hotspot; High vulnerability
1066	Jiwani Coastal Wetland	25.08	61.80	Hotspot; High vulnerability
1067	Jubho Lagoon	24.33	68.67	Hotspot; Very high vulnerability
1069	Nurri Lagoon	24.50	68.78	Hotspot; Very high vulnerability

1075	Agh-Ghol	40.02	47.63	Hotspot; High vulnerability
1076	Ghizil-Agaj	39.12	48.98	Hotspot; High vulnerability
1083	Kayrakum Reservoir	40.33	70.17	Hotspot; High vulnerability
1108	Lake Dengizkul	39.12	64.17	Hotspot; High vulnerability
1109	Gomishan Lagoon	37.20	53.98	Hotspot; Very high vulnerability
1110	Verloren Valei Nature Reserve	-25.31	30.11	Hotspot; High vulnerability
1148	Eerduosi National Nature Reserve	39.80	109.58	Hotspot; Very high vulnerability
1160	Kanjli	31.42	75.37	Hotspot; High vulnerability
1161	Ropar	31.02	76.50	Hotspot; High vulnerability
1205	Bhitarkanika Mangroves	20.65	86.90	Hotspot; High vulnerability
1206	Bhoj Wetland	23.23	77.33	Hotspot; High vulnerability
1208	East Calcutta Wetlands	22.45	88.45	Hotspot; High vulnerability
1211	Pong Dam Lake	32.02	76.08	Hotspot; High vulnerability
1213	Tsomoriri	32.90	78.30	Hotspot; High vulnerability
1285	Runn of Kutch	24.38	70.08	Hotspot; Very high vulnerability
1296	Chott Melghir	34.25	6.51	Hotspot; High vulnerability
1313	Beeshazar and Associated Lakes	27.62	84.43	Hotspot; High vulnerability
1314	Ghodaghodi Lake Area	28.68	80.95	Hotspot; High vulnerability
1315	Jagadishpur Reservoir	27.58	83.08	Hotspot; High vulnerability
1326	Playa Tortuguera Rancho Nuevo	23.23	-97.77	Hotspot; High vulnerability
1362	Laguna Madre	24.73	-97.58	Hotspot; High vulnerability
1378	Lake Ganga and its surrounding wetlands	45.25	114.00	Hotspot; High vulnerability
1399	Dnipro-Oril Floodplains	48.53	34.75	Hotspot; High vulnerability

1417	Chott Sidi Slimane	33.29	6.05	Hotspot; High vulnerability
1439	Mapangyong Cuo	30.69	81.39	Hotspot; High vulnerability
1441	Shuangtai Estuary	40.91	121.76	Hotspot; High vulnerability
1569	Chandertal Wetland	32.48	77.60	Hotspot; High vulnerability
1570	Hokera Wetland	34.08	74.70	Hotspot; High vulnerability
1571	Renuka Wetland	31.62	77.45	Hotspot; High vulnerability
1573	Surinsar-Mansar Lakes	32.75	75.20	Hotspot; High vulnerability
1574	Upper Ganga River	28.55	78.20	Hotspot; High vulnerability
1588	Chatyr Kul	40.62	75.30	Hotspot; High vulnerability
1687	Makuleke Wetlands	-22.39	31.20	Hotspot; High vulnerability
1692	Gokyo and associated lakes	27.95	86.68	Hotspot; High vulnerability
1693	Gosaikunda and Associated Lakes	28.08	85.43	Hotspot; High vulnerability
1694	Phoksundo Lake	29.20	82.95	Hotspot; High vulnerability
1695	Rara Lake	29.50	82.08	Hotspot; High vulnerability
1697	Bahiret el Bibane	33.25	11.22	Hotspot; Very high vulnerability
1699	Chott El Jerid	33.70	8.40	Hotspot; High vulnerability
1700	Djerba Bin El Ouedian	33.67	10.92	Hotspot; Very high vulnerability
1701	Djerba Guellala	33.70	10.73	Hotspot; Very high vulnerability
1702	Djerba Ras Rmel	33.87	10.90	Hotspot; Very high vulnerability
1714	Zones humides oasiennes de Kebili	33.50	8.92	Hotspot; High vulnerability
1718	Hawizeh Marsh	31.42	47.63	Hotspot; Very high vulnerability
1762	Laguna de Babícora	29.33	-107.83	Hotspot; High vulnerability
1769	Río Sabinas	27.88	-101.15	Hotspot; High vulnerability

1822	Sistema de Humedales Remanentes del Delta del Río Colorado	32.32	-115.25	Hotspot; High vulnerability
1841	Aydar-Arnasay Lakes system	40.78	67.77	Hotspot; High vulnerability
1850	Mai Pokhari	27.00	87.92	Hotspot; High vulnerability
1855	Turkmenbashy Bay	39.78	53.35	Hotspot; High vulnerability
1863	Kulykol-Taldykol Lake System	51.38	61.87	Hotspot; High vulnerability
1872	Naurzum Lake System	51.49	64.30	Hotspot; High vulnerability
1873	Zharsor-Urkash Lake System	51.32	62.73	Hotspot; High vulnerability
1890	Lake Kuyucuk	40.75	43.45	Hotspot; High vulnerability
1892	Alakol-Sasykkol Lakes System	46.27	81.53	Hotspot; High vulnerability
1917	Roswell Artesian Wetlands	33.45	-104.38	Hotspot; High vulnerability
1939	Choghakhor Wetland	31.92	50.90	Hotspot; Very high vulnerability
1940	Kanibarazan Wetland	37.00	45.77	Hotspot; High vulnerability
1943	Son-Kol Lake	41.83	75.12	Hotspot; High vulnerability
1981	Baño de San Ignacio	24.87	-99.34	Hotspot; High vulnerability
1989	Khor Virap Marsh	39.89	44.57	Hotspot; High vulnerability
2005	Chott Elguetar	34.29	8.91	Hotspot; High vulnerability
2007	Marais d'eau douce Garaet Douza	34.47	8.48	Hotspot; High vulnerability
2008	Golfe de Boughrara	33.47	10.75	Hotspot; Very high vulnerability
2009	Les Gorges de Thelja	34.15	8.28	Hotspot; High vulnerability
2011	Oued Dekouk	32.34	10.61	Hotspot; Very high vulnerability
2047	Río San Pedro - Meoqui	28.28	-105.44	Hotspot; High vulnerability
2070	Humedales de Península Valdés	-42.46	-64.30	Hotspot; Very high vulnerability
2083	Lesser Aral Sea and Delta of the Syrdarya River	46.35	61.00	Hotspot; High vulnerability

2088	Mui Ca Mau National Park	8.68	104.79	Hotspot; Very high vulnerability
2100	Complexe des zones humides de Sebkhet Oum Ez-Zessar et Sebkhet El Grine	33.65	10.52	Hotspot; Very high vulnerability
2123	Van Eck Dam	-26.77	31.92	Hotspot; High vulnerability
2187	Shandong Yellow River Delta Wetland	37.77	119.09	Hotspot; Very high vulnerability
2201	Manantiales Geotermales de Julimes	28.41	-105.43	Hotspot; High vulnerability
2203	Con Dao National Park	8.71	106.64	Hotspot; Very high vulnerability
2206	Laguna La Juanota	26.49	-106.47	Hotspot; High vulnerability
2224	Complexe des lacs Ambondro et Sirave (CLAS)	-20.91	43.94	Hotspot; High vulnerability
2228	U Minh Thuong National Park	9.59	105.10	Hotspot; Very high vulnerability
2239	Mubarak Al-Kabeer Reserve	29.90	48.13	Hotspot; High vulnerability
2240	Sawa Lake	31.31	45.01	Hotspot; Very high vulnerability
2241	Central Marshes	31.18	46.98	Hotspot; Very high vulnerability
2242	Hammar Marsh	30.81	47.02	Hotspot; Very high vulnerability
2257	Lake Cluster of Pokhara Valley	28.21	83.98	Hotspot; High vulnerability
2282	Archipelago Velyki and Mali Kuchugury	47.56	35.20	Hotspot; High vulnerability
2294	Fifa Nature Reserve	30.96	35.44	Hotspot; Very high vulnerability
2302	Mangroves de Tsiribihina	-19.74	44.46	Hotspot; High vulnerability
2360	Van Long Wetland Nature Reserve	20.39	105.85	Hotspot; High vulnerability
2369	Zarivar	35.54	46.13	Hotspot; High vulnerability
2370	Sundarban Wetland	21.77	88.71	Hotspot; High vulnerability
2407	Nangal Wildlife Sanctuary	31.40	76.37	Hotspot; High vulnerability
2408	Beas Conservation Reserve	31.39	75.19	Hotspot; High vulnerability
2409	Sandi Bird Sanctuary	27.31	79.97	Hotspot; High vulnerability

2411	Sarsai Nawar Jheel	26.97	79.25	Hotspot; High vulnerability
2412	Nawabganj Bird Sanctuary	26.61	80.65	Hotspot; High vulnerability
2413	Saman Bird Sanctuary	27.02	79.18	Hotspot; High vulnerability
2414	Keshopur-Miani Community Reserve	32.09	75.39	Hotspot; High vulnerability
2415	Samaspur Bird Sanctuary	26.00	81.39	Hotspot; High vulnerability
2416	Parvati Arga Bird Sanctuary	26.94	82.16	Hotspot; High vulnerability
2425	Tianjin Beidagang Wetlands	38.79	117.36	Hotspot; Very high vulnerability
2433	Tudakul and Kuymazar Water Reservoirs	39.85	64.83	Hotspot; High vulnerability
2434	Bugdasheni Lake	41.20	43.68	Hotspot; High vulnerability
2435	Madatapa Lake	41.18	43.78	Hotspot; High vulnerability
2436	Kabartal Wetland	25.62	86.14	Hotspot; High vulnerability
2437	Asan Conservation Reserve	30.43	77.68	Hotspot; High vulnerability
2440	Sur Sarovar	27.25	77.84	Hotspot; High vulnerability

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Appendix VII

SUPPLEMENTARY INFORMATION FOR CHAPTER 7

VII.1 A simple delineation algorithm

Our derivation of groundwatersheds differs from existing approaches to map groundwater catchments, groundwater basins, and groundwater divides that use numerical or analytical models under assumptions of idealized conditions (Haitjema, 1995; Parker et al., 2016; Tiedeman et al., 1998). Though recent advances in computational power have extended these approaches to the continental scale (e.g., Maxwell et al., 2016), current limitations in data availability and computational resources make these numerical approaches not feasible at the global scale. Thus, we derived groundwatersheds using solely the topography of the water table as input to a common and simple watershed delineation methodology based on the D8 flow direction method (O'Callaghan & Mark, 1984). Importantly, this approach rendered our derived groundwatersheds to be largely consistent with localized groundwater flow systems but not with subregional and regional groundwater flow systems that are the product of regional gradients in the potentiometric surface (Figure VII.1). Simultaneously, the D8 flow direction method is very sensitive to the water table data, and thus our mapped groundwatershed results are conditional to the water table data we used (Table VII.1). See Section VII.5 below for further discussion on bias in the water table data used and its impacts on study outcomes.

We use groundwater-dependent ecosystems inside protected areas as the outlet features for our groundwatershed delineation. While these groundwater-dependent ecosystems contain lotic ecosystems, such as perennial rivers and streams, we do not include groundwater-dependent ecosystems (GDEs) that are located upstream of protected areas (i.e., we do not delineate groundwatersheds based on routed surface flow). We do this for two reasons. One, the groundwatersheds we derive represent local groundwater flow systems and including groundwatersheds of potentially distant surface water features that flow through a protected area diverges from our focus on direct, local impacts. Secondly, including groundwatersheds of upstream surface water features requires methodological decisions on scales of surface water system representations. Addressing differences between surface watersheds and groundwatersheds is a non-trivial endeavour, which we discuss further in Section VII.3 below.

We also do not differentiate between hydrologically connected and disconnected river reaches in our determination of groundwater-dependent ecosystems. We do this because data on groundwater-stream interactions is poorly represented in currently available global data. While we only consider perennial stream and river reaches, which are more likely to be reliant on groundwater discharge to sustain streamflow during dry reasons and are less likely to be hydrologically disconnected, we note that groundwater-surface water interactions in ephemeral streams are understudied in the literature (Quichimbo et al., 2020). We also do not remove losing river and stream reaches as surrounding water table levels regulate the hydraulic gradient across groundwater-surface water interactions.

Our analysis is a first-order, global scoping analysis and proof-of-concept for the application of groundwatershed mapping globally. Our simple approach provides space for future advances in mapping groundwatersheds globally, which will be possible once the above-mentioned challenges regarding computational resources, data availability and data quality are addressed.

VII.2 Comparison to surface watersheds

A similar analysis to ours that investigates the size and protected coverage of surface watersheds would complement this study and support a more comprehensive assessment of hydrological connections to protected areas. Yet, there are important differences in surface and groundwatershed delineation that make this comparison non-trivial and beyond the scope of our study:

1. **Outlet features**: In our derivation of groundwatersheds, we use three forms of groundwater-dependent ecosystems as outlet features: lotic ecosystems (i.e., perennial rivers and streams), lentic ecosystems (i.e., groundwater-dependent wetlands and lakes), and terrestrial ecosystems (where root zones intersect the water table). However, the full set of these groundwater-dependent ecosystems are not natural outlets for surface watersheds. For instance, it is not intuitive how or why a terrestrial ecosystem would be used as an outlet feature for a surface watershed delineation method. Furthermore, whereas a surface watershed typically has one outlet feature, our derivation of groundwatersheds uses as many outlet features as there are groundwater-dependent ecosystem grid cells inside each protected area.

2. **Scale dependence**: Surface watersheds are nested, hierarchical systems. Thus, delineating a surface watershed depends on a selection of scale (Lehner & Grill., 2013). Groundwater flow systems are also nested (Condon et al., 2020), yet due to data availability, and computational limits that lead to our use of the D8 flow direction method, we exclusively focus on shallow, local groundwater systems.
3. **Spatial resolution**: The water table data we base our groundwatershed delineation process on is made available at the spatial resolution of 30 arcsecond (~1 km). Yet, surface watersheds are typically derived at much higher spatial resolutions. For example, HydroBASINS are derived using a 3 arcsecond (~90 m) land surface elevation model (Lehner & Grill, 2013). Thus, our groundwatersheds are derived at a spatial resolution 10 times coarser (i.e., one 30 arcsecond grid cell to one-hundred 3 arcsecond grid cells) than the global standard for surface watersheds.

Despite these limiting differences, we performed a simple, scoping analysis to highlight the scale dependence of comparing to surface watersheds and to reveal general principles underpinning the characteristic differences between groundwatershed and surface watershed delineation.

In this simplistic, illustrative analysis (Figure VII.5), we calculated a rough approximation of the underprotected surface watershed ratio (i.e., a surface water analogue to the underprotected groundwatershed ratio, UGR).

We do not calculate a surface water analogue to the relative groundwatershed size (RGS), which is calculated as the ratio of groundwatershed size to the size of groundwater-dependent ecosystems within the protected area. RGS is similar to the conventional watershed property of drainage density (calculated as the ratio of watercourse length to basin size, with units of L^{-1}). However, RGS is inverted with respect to drainage density (i.e., basin size is in the numerator rather than the denominator and is replaced by groundwatershed size) and the length of watercourses inside the basin is replaced with the groundwater-dependent ecosystem area. Thus, an additional difference between RGS and drainage density is that RGS is a unitless metric. In terms of groundwatersheds, RGS is a useful metric to compare groundwater flow system size independent of protected area size and generally reveals where there are larger or smaller localized groundwater flow systems. Yet, the surface water analogue of RGS is not as intuitively useful. This is partially because the groundwater-dependent ecosystems we use are not reasonable outlet locations for a surface watershed, and because the surface area represented at by 30 arcsecond grid cells of surface watercourses overestimates surface water area inside the grid cell (see discussion on spatial resolution limitations in Section VII.5.3).

In this supplementary surface watershed analysis, we show the general principle that selecting a larger watershed scale will lead to a greater estimation of an underprotected surface watershed ratio (UGR analogue). We use nested HydroBASIN watersheds for this analysis at four scales: level 7 (large) to level 10 (small). HydroBASINS watershed levels range from level 1 (largest) to level 12 (smallest/most-nested), but we use only the range of levels 7 to 10 for illustration purposes. In general, we can observe that the underprotected ratio of surface watersheds increases as surface watershed scale increases. Yet, even in this simple demonstration, there are key differences in how this underprotected ratio is calculated when compared to groundwatersheds:

1. **Downstream basin area**: We do not remove portions of the surface watersheds that are downstream of the protected area. We do this because identifying downstream segments of the basin requires significant additional processing of land surface elevation, equivalent in scope to an analogous surface watershed delineation analysis which would be sufficient for a separate study. Thus, depending on where the protected area is located inside each basin (which also depends on the HydroBASIN level being used), there may be a substantial proportion of the basin that is downstream of the protected area yet is included in the underprotected ratio calculation in this analysis.
2. **Increasing extent of protection with scale**: Whereas groundwatersheds are individually assigned to protected areas, with overlapping groundwatershed areas assigned on an upgradient-to-downgradient prioritization basis, we do not do this for this surface watershed analysis. We do this for the same reason we do not remove downstream basin areas: it would require supplemental analysis on surface elevation-based flow direction and accumulation to determine the upstream-downstream order of protected areas inside the same HydroBASIN. Given this limitation, we calculate the underprotected ratio considering all protected areas inside a basin. As the basin scale increases, it becomes more likely that additional protected areas are found inside the basin.

The impact of these differences is two-fold. Firstly, as we do not remove downstream areas from basins, actual underprotected ratios in this surface watershed analysis would be lower than our preliminary results show (Figure VII.5e). Secondly, the increasing extent of protection at larger scales reduces the differences in underprotected ratios across scales. We hypothesize that if we isolated surface water basins per protected area, the differences between underprotected ratios across HydroBASIN levels 10 through 7 would be more pronounced. These limitations and methodological challenges hamper our ability to make explicit conclusions and comparisons between underprotected ratios for groundwatersheds and surface watersheds.

VII.3 Handling of overlapping groundwatersheds

Groundwatershed extents can overlap if an (upgradient) protected area exists inside the groundwatershed of another (downgradient) protected area. In these cases, the groundwatershed of the upgradient protected area is also part of the groundwatershed of the downgradient protected area. However, assigning total groundwatershed areas to both protected areas will lead to double counting the overlapping groundwatershed area. Our simple solution for this is to assign groundwatersheds based on an upgradient-to-downgradient prioritization scheme such that each protected area receives a groundwatershed whose extent is mutually exclusive of all other protected area groundwatershed extents. This approach is visualised in Figure VII.7.

This approach impacts the outcome of both metrics we use to understand patterns in groundwatershed results (i.e., the relative groundwatershed size, RGS; and the underprotected groundwatershed ratio, UGR). For RGS, which is calculated as the ratio of groundwatershed area to GDE area inside the groundwatershed, both arguments of this calculation can be affected. For example, using the example of the downgradient protected area as discussed above, the groundwatershed size will decrease (as only the non-overlapping area will remain) while the GDE surface area will decrease if there are GDE areas inside the overlapping groundwatershed area that are removed. Thus, the impact of this approach on the RGS metric is an increase in RGS if the overlapping groundwatershed area itself has a higher RGS than the remaining groundwatershed area, or a decrease in RGS in the inverse situation. For UGR, which is calculated as the ratio of underprotected groundwatershed area to total groundwatershed area, both arguments of the calculation can also be affected. For the downgradient protected area, both the total and underprotected groundwatershed areas will decrease. The impact on the UGR metric for the downgradient protected area will thus depend on if the proportion of underprotected-to-protected groundwatershed area is consistent for the mutually exclusive groundwatershed areas and inside the overlapping groundwatershed area.

VII.4 Uncertainty analysis

While groundwatershed extents exhibit month-to-month variation, we found that the total area of groundwatersheds for the world's protected areas fluctuates little throughout the year (Figure VII.8 and Figure VII.9). These small fluctuations in total groundwatershed size indicate the consistency of this first-order analysis of global groundwatershed mapping. However, these results simultaneously underscore the need for methodological advances to map groundwatersheds using improved process representation in future studies to refine these estimates and allow for consideration of regional and complex groundwater flow systems.

VII.5 Study limitations

VII.5.1 Representation of groundwater flow systems

As discussed above, our mapping of groundwatersheds is based solely on the gradient of the water table removes our ability to consider both regional flow systems and complex aquifers. Thus, our representation of local groundwater flow systems is a product of both the coarse spatial resolution of our analysis (30 arcsecond; ~1 km at the equator) and our assumption that the water table gradient is a sufficient proxy to represent the extents of localized groundwater flow systems. Furthermore, we used the rudimentary but computationally efficient D8 flow direction method.

VII.5.2 Geological simplifications

Although the D8 algorithm does not account for subsurface heterogeneity, the Fan et al. model does use heterogeneous hydraulic conductivity in the subsurface as soil hydraulic conductivity with an exponential decay function with depth (see Section S2.2. Hydraulic Conductivity in the Supplementary Information of Fan et al., 2013). Thus, since we are using their model output, we are implicitly incorporating subsurface permeability.

Furthermore, our water table-based groundwatershed delineation method operates on an implicit assumption that areas contributing to streams are unconfined surficial aquifers. In arid and high topographic gradient regions where water tables are deeper, our assumption of surficial aquifers with connections to streams is challenged. However, modifying our methodology to account for complex geologies is intractable at the global scale and would require a patchwork of methods depending on the geological setting. This limitation underscores our calls for more advanced, computationally intensive methods to understand the entire hydrological system, including deeper, nested groundwater flow systems at smaller scales.

VII.5.3 Spatial resolution and extent of groundwater-dependent ecosystems

Spatial resolution is a core parameter of any geospatial study (Dark & Bram, 2007). This study is conducted at the spatial resolution of 30 arcsecond (~1 km at the equator) which is a necessary precondition for interpretation of all results. We operated at 30 arcsecond as it is the resolution of Fan et al.'s (2017) water table depth model output, the foundational dataset of this study.

Other datasets used in this study that were not produced at 30 arcsecond (see Table VII.1) required spatially harmonization. For raster data, this only included groundwater-dependent wetlands (or, as referred to in Tootchi et al. (2019), groundwater-driven wetlands) which are provided at 15 arcseconds. For this data, we performed maximum value aggregation, meaning that a 30 arcsecond grid cell would represent the presence of a groundwater-dependent wetland if at least one of the four nested 15 arcsecond grid cells were identified as a groundwater-dependent wetland. While the Tootchi et al. groundwater-dependent wetland study itself notes that its estimated coverage of groundwater-driven wetlands is among the highest reported estimates in the literature, we subsequently overrepresent these wetlands through this spatial harmonization method.

Furthermore, when harmonizing raw vector data to 30 arcsecond raster format, we set rasterization arguments so that all grid cells that touch a vector line are included in the raster output. In the case of perennial streams and rivers, this method overrepresents the spatial extent of these water bodies as the vast majority of global streams and rivers have widths less than 1 km (Downing et al., 2012). The same principle applies to borders of lakes which do not entirely overlap a grid cell. Additionally, our use of maximum rooting depth per grid cell also overestimates terrestrial groundwater-dependent ecosystem extent as it is unlikely that the maximum rooting depth per grid cell is found across the entire grid cell.

Thus, the groundwater-dependent ecosystem grid cells we use in this study are not a comprehensive and refined global dataset but rather an indication of where these ecosystems can potentially occur. Conducting this study at 30 arcsecond is a pragmatic and robust approach given current data availability, yet we emphasize that this spatial resolution is necessary context for interpretation of all results, including foremost the global extent of groundwatersheds.

VII.6 Supplementary figures

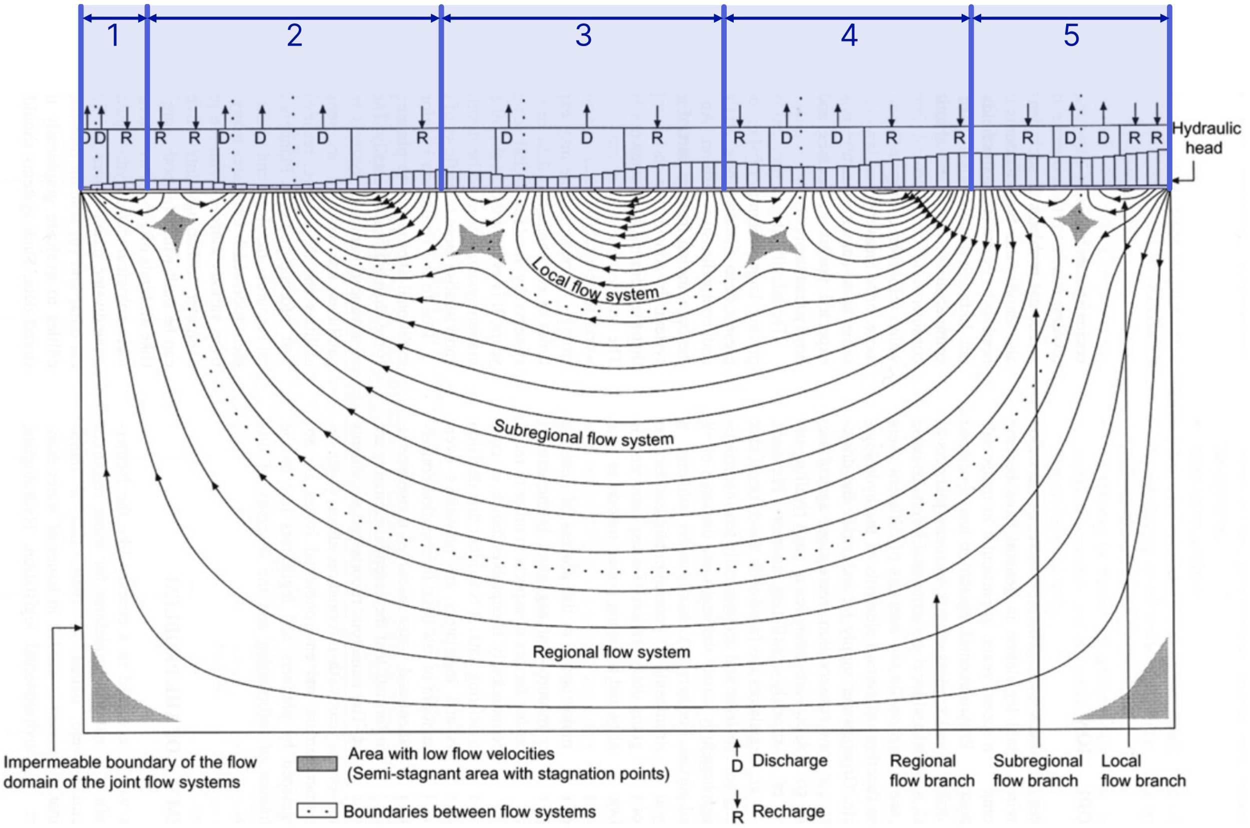


Figure VII.1. Groundwatersheds as representations of local flow systems.

A nested groundwater flow systems diagram (from Zhou et al., 2011) with individual groundwatersheds annotated using labels 1-5. Groundwatersheds are derived based on the topography of the water table (hydraulic head in the diagram). These groundwatersheds correspond closely with the local flow systems.

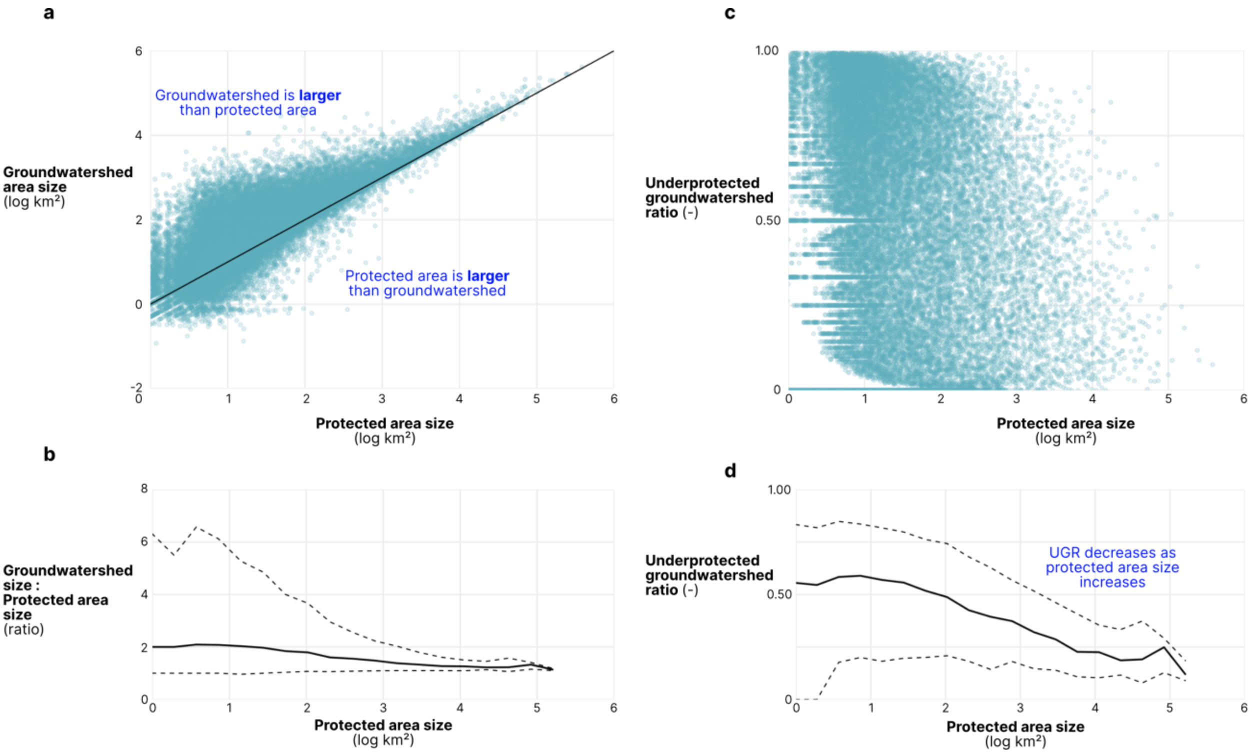


Figure VII.2. Relationships between protected area size, groundwatershed size, and groundwatershed protection.

(a) Scatter plot of protected area size and groundwatershed size (n = 32,490). The black line shows a 1:1 relationship. Both axes are plotted on a log10 scale. (b) Moving median (solid), and upper and lower quartile (dashed) ratios of groundwatershed size to protected area size as protected area increases and is shown on a semi-log plot. (c) Scatter plot protected area size and the underprotected groundwatershed ratio, shown on a semi-log plot (n = 32,490). (d) Moving median (solid), and upper and lower quartile (dashed) ratios of the underprotected groundwatershed ratio as protected area size increases.

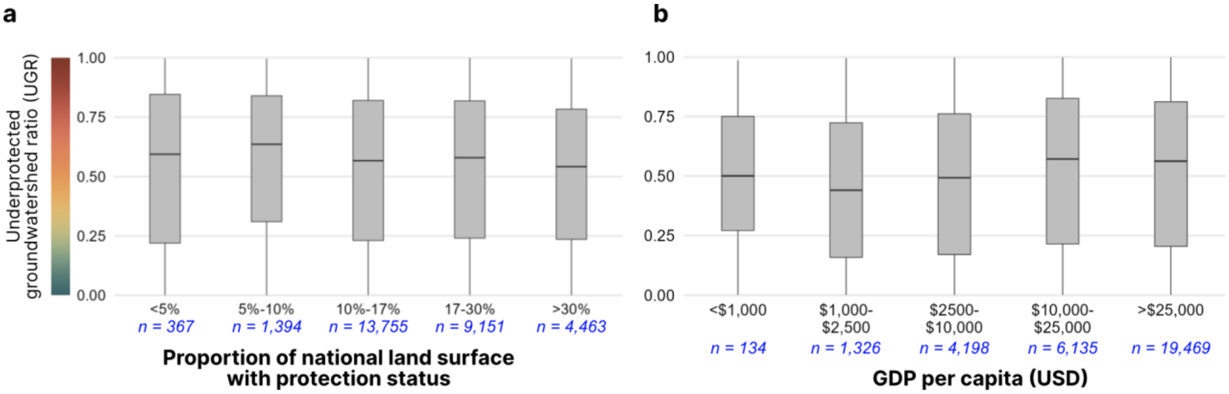


Figure VII.3. Exploratory plotting of the relationship between the underprotected groundwatershed ratio (UGR) and possible influences on groundwatershed protection.

(a) Boxplots representing the distribution of UGR against classes of national level terrestrial land area protection rates, for all groundwatersheds with available data on national land surface levels of protection (n = 29,230). (b) Boxplots representing the distribution of UGR against classes of GDP per capita, for all groundwatersheds with available data on GDP per capita (n = 31,362). Whiskers in all boxplots extend to minimum and maximum values per class; each box shows the interquartile range (i.e., the 25th and 75th percentiles); and the centre line represents the median value per class. Numbers in blue below the x-axis state the number of groundwatersheds in each class. Data for national land surface protection levels come from the World Bank Data Catalogue (2023), based on data from the World Database on Protected Areas. GDP per capita data come from Kummu et al. (2018).

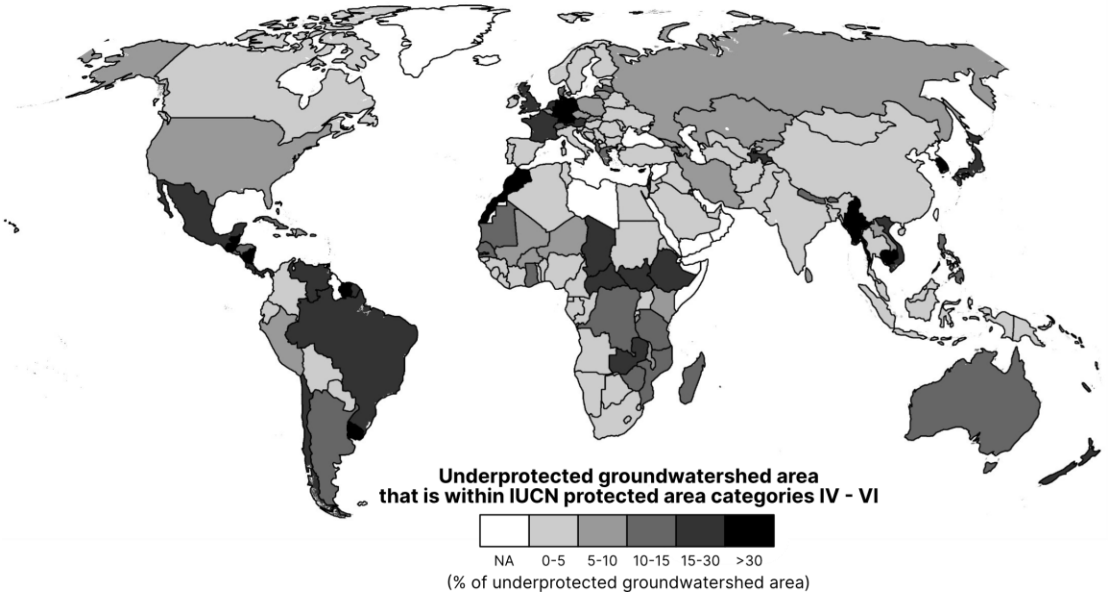


Figure VII.4. The proportion of underprotected groundwatershed surface area that is inside lower levels of protection (IUCN categories IV-VI) when summarized to the national scale.

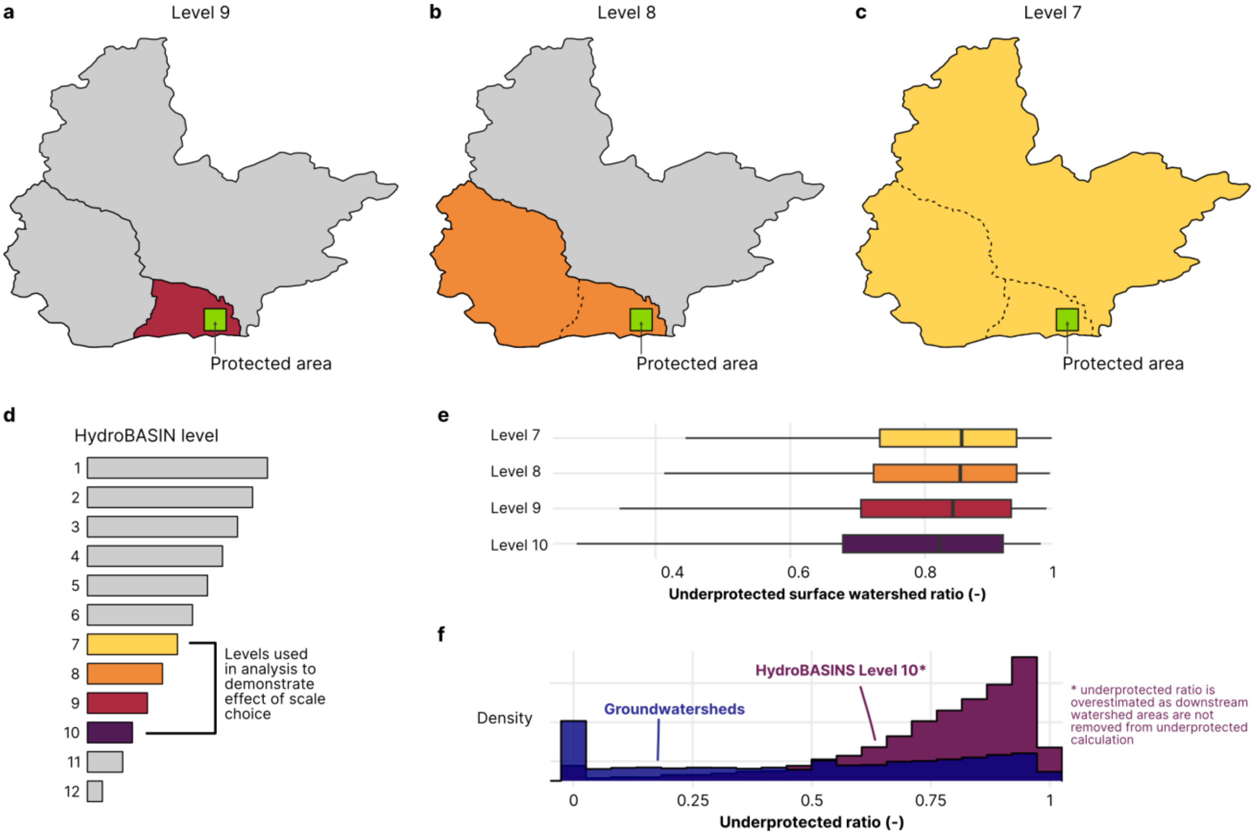
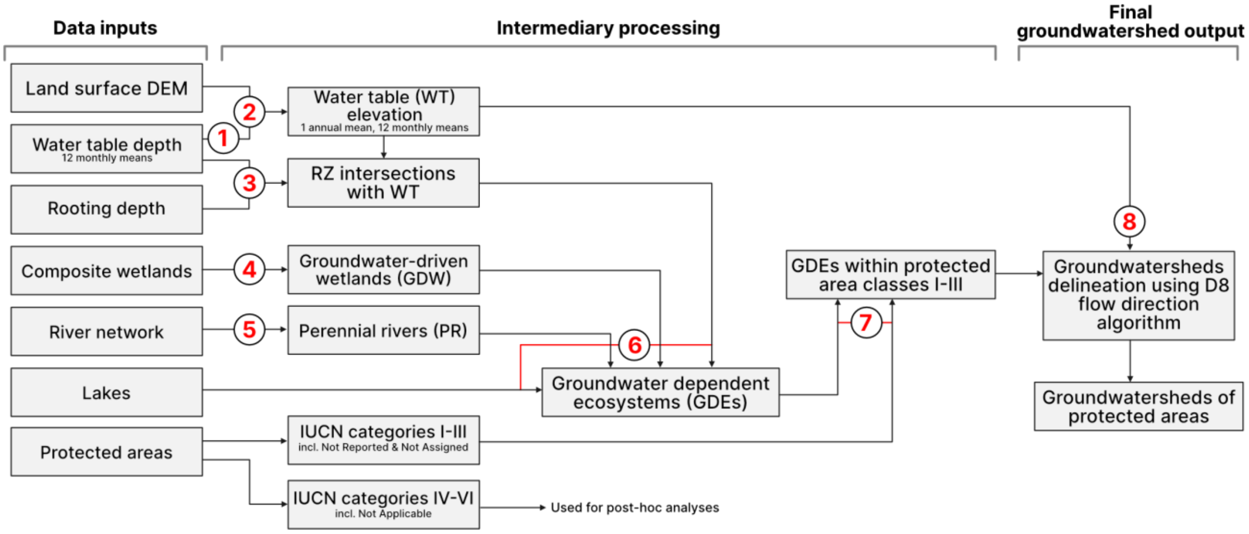


Figure VII.5. Preliminary analysis demonstrating the impact of scale in a comparative analysis with surface watersheds.

(a-c) Hypothetical HydroBASIN levels 9, 8, and 7, which are nested from higher levels to lower levels, as shown in (d). (e) Underprotected surface watershed ratio for global protected areas, calculated for HydroBASIN levels 7 through 10. Boxplots show 5th (lower whisker terminus), 25th (lower box bound), 50th (box centre), 75th (upper box bound), and 95th (upper whisker terminus) percentiles of the distribution of underprotected surface watershed ratios for the global set of protected areas with groundwatersheds in our analysis (n = 32,490).



Notes:

① Annual water table derived by calculating time-weighted mean across mean monthly water table data

② Water table elevations derived by subtracting water table depth from the land surface elevation model

③ Root zone intersection identified if the maximum rooting depth intersects the water table for a minimum of one month per year

④ Groundwater-driven wetlands are isolated from a composite wetland map

⑤ Perennial rivers (i.e. reaches with <50% probability of ceasing to flow for min. 1 month per year) filtered from the global river network

⑥ Designated as GDE if any one of the four inputs (ie. RZ intersection, GDW, PR, or lakes) occur

⑦ IUCN classes Ia, Ib, II, and III are used to mask GDEs which are subsequently used as outlet features for groundwatershed delineation

⑧ Groundwatershed delineation is performed using WhiteboxTools's functions D8Pointer and Watershed, with the D8 flow direction raster being generated based on the annual mean water table (1) and for outlet features set to GDEs within protected areas (8)

Figure VII.6. Overview of this study's workflow to derive groundwatersheds for the world's protected areas.

See numbered annotations in red for additional information on select operations.

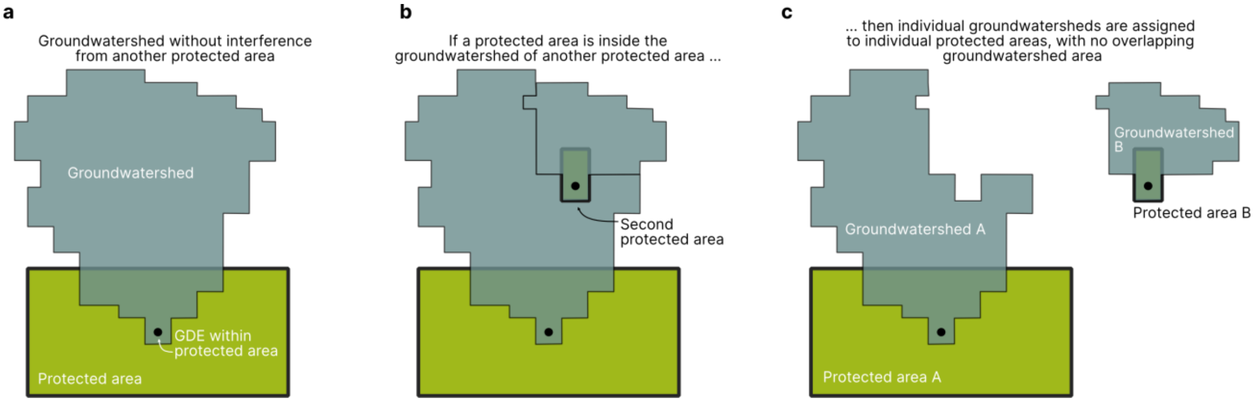


Figure VII.7. Groundwatershed assignment to protected areas when a protected area is located inside another protected area's groundwatershed.

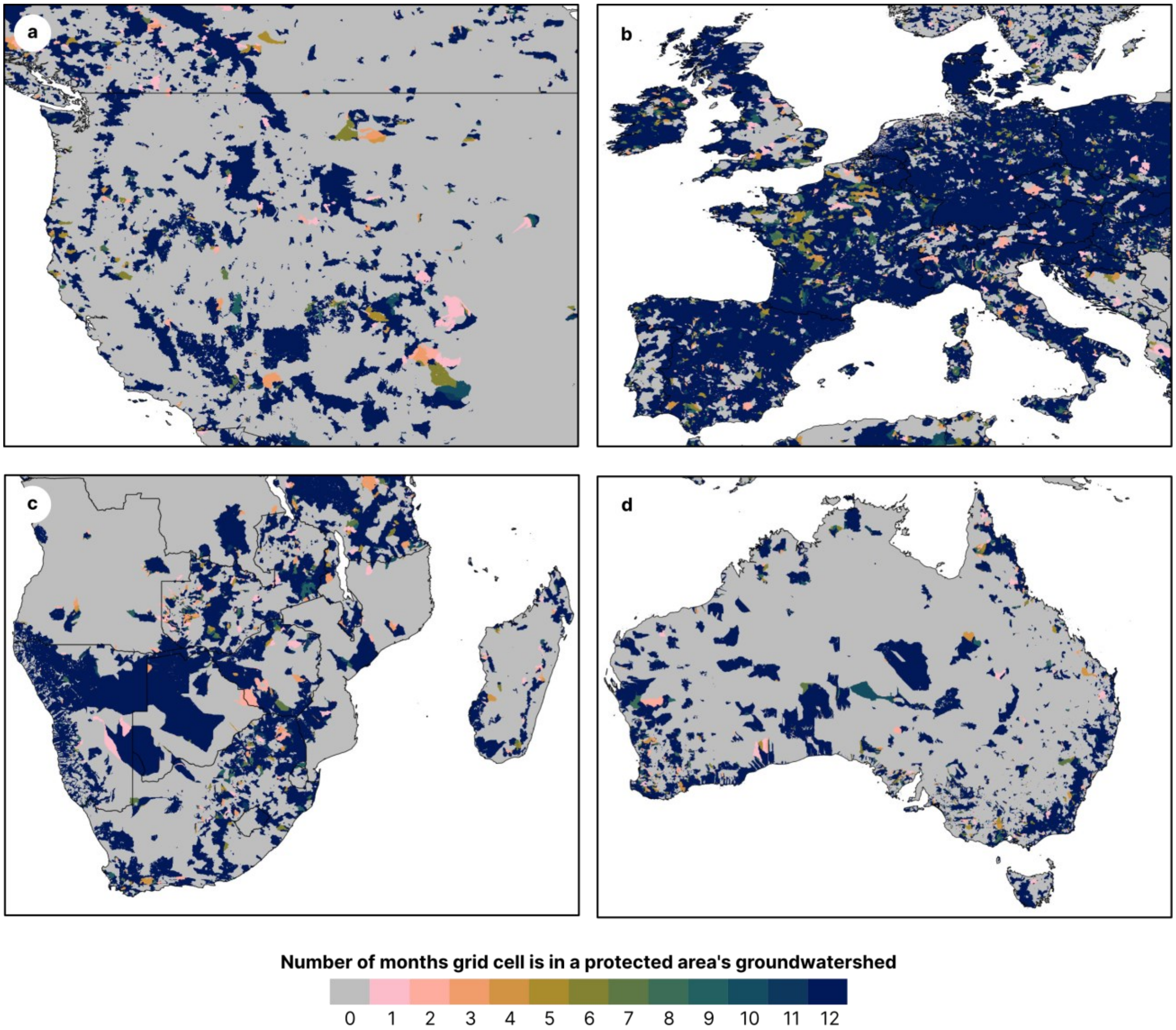


Figure VII.8. Sub-annual variability in groundwatershed extents shown for select regions of the world.

(a) Western North America. (b) Western Europe. (c) Southern Africa. (d) Australia. Colours represent the number of months per year that each grid cell is in a protected area's groundwatershed.

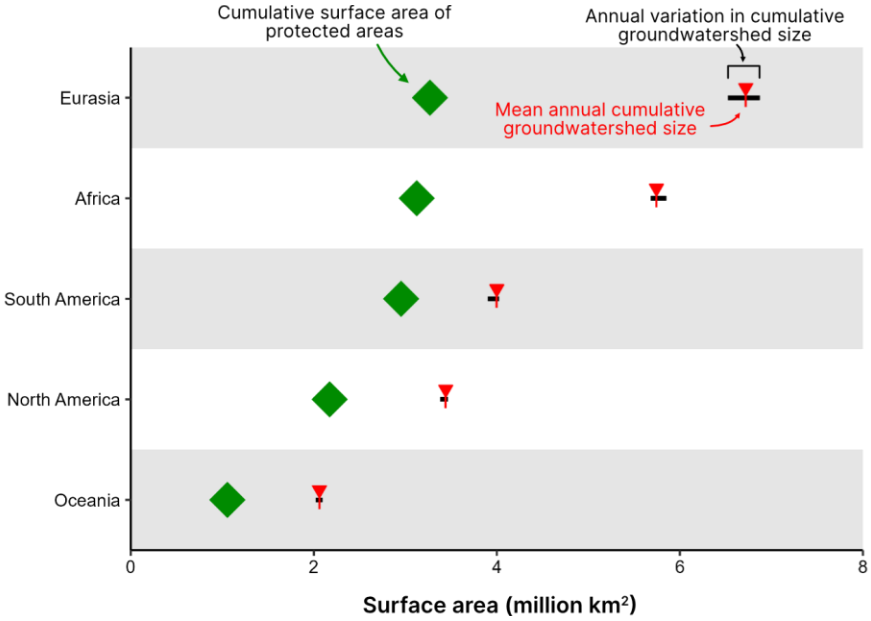


Figure VII.9. Sub-annual variation in cumulative groundwatershed size per continent. The distribution of groundwatershed sizes is calculated using mean monthly water tables for each month of the year. The cumulative surface area of protected areas per continent considered in our analysis, per continent, are shown by green diamonds. Month-to-month ranges (n = 12) in cumulative groundwatershed extent are shown by the bounds of the horizontal, thick black lines. Mean annual cumulative groundwatershed extent is shown by inverted red triangles.

VII.7 Supplementary tables

Table VII.1. Data sources, descriptions, justifications for inclusion, and preprocessing steps.

Dataset	
Protected areas	**Data source:** World database on protected areas (WDPA) (UNEP-WCMC & IUCN, 2021) **Persistent web-link:** https://www.protectedplanet.net **Temporal range:** Last accessed 7 June 2021. **Spatial resolution:** Vector data. **Description and justification.** The WDPA is the most extensive database of protected areas and is the standard data source for global studies considering protected areas. **Preprocessing:** Protected areas were filtered for their protected area class and reported surface area (see 3. Methodology), and subsequently rasterized in a binary representation of protected area presence/non-presence at 30 arcsecond (~1 km) resolution, including all grid cells touched by a protected area. All spatially contiguous protected areas were then identified and provided a unique ID for analysis.
Water table depth	**Data source:** Fan et al. (2017) **Persistent web-link:** http://thredds-gfnl.usc.es/thredds/catalog/GLOBALWTDFTP/catalog.html **Temporal range:** Mean monthly results over a 2004-2013 model run. **Spatial resolution:** 30 arcsecond (~1 km) **Description and justification:** The leading global water table depth dataset. We also select this dataset as it is produced in the same study as the maximum rooting depth data. **Preprocessing:** Water table depths were converted to water table elevations by subtracting the water table depth from the land surface elevation (see below).
Maximum rooting depth	**Data source:** Fan et al. (2017) **Persistent web-link:** https://wci.earth2observe.eu/thredds/catalog/usc/root-depth/catalog.html

	Temporal range: Average value of 2004-2013 model run. **Spatial resolution:** 30 arcsecond (~1 km) **Description and justification:** To our knowledge, this is the only spatially distributed dataset of maximum rooting depth with full global terrestrial surface area coverage. **Preprocessing:** None
Land surface elevation	**Data source:** Associated with Fan et al. (2017) **Persistent web-link:** Provided through direct author correspondence. **Temporal range:** N/A **Spatial resolution:** 30 arcsecond (~1 km) **Description and justification:** The land surface elevation data provided and used by the Fan et al. studies listed above. These studies mosaiced 30 arcsecond USGS HydroSHEDS digital elevation model, that hydrologically conditions NASA Shuttle Radar Topography Mission (SRTM) elevation data, for grids south of 60°N. For grids north of 60°N, the studies averaged 1 arcsecond NASA-JPL ASTER Global Digital Elevation Map within 30 arcsecond grid cells. **Preprocessing:** None
Perennial rivers	**Data source:** Messager et al. (2021) **Persistent web-link:** https://figshare.com/articles/dataset/Global_prevalence_of_non-perennial_rivers_and_streams/14633022 **Temporal range:** N/A **Spatial resolution:** Vector data **Description and justification:** A global prediction of river flow intermittence probability, using the river network of the global RiverATLAS database for all stream reaches with a mean annual flow of 0.1 m^3s^{-1}. **Preprocessing:** Rasterized all perennial rivers, which are identified at the individual river reach level, to a 30 arcsecond (~1 km) grid including all grid cells touched by a perennial river.
Groundwater-driven wetlands	**Data source:** Tootchi et al. (2019) **Persistent web-link:** https://doi.pangaea.de/10.1594/PANGAEA.892657

	Temporal range: N/A **Spatial resolution:** 15 arcsecond (~500 m) **Description and justification:** Global composite wetland maps that specify sub-classes of routinely flooded wetlands (RFW) and groundwater-driven wetlands (GWD). Though other global wetland maps exist, this is the only dataset to our knowledge that explicitly identifies groundwater-driven wetlands. **Preprocessing:** Groundwater-driven wetlands were isolated from the composite wetland maps and aggregated to 30 arcsecond (~1 km) resolution based on a binary evaluation of if a groundwater-driven wetland grid cell at the original resolution was contained within the grid cell at the aggregated resolution.
Lakes	**Data source:** Messager et al. (2016) **Persistent web-link:** https://www.hydrosheds.org/products/hydrolakes **Temporal range:** N/A **Spatial resolution:** Vector data. **Description and justification:** The leading global lakes dataset, which aims to include all lakes with a minimum surface area of 10 ha. **Preprocessing:** All lakes included in the database are rasterized to a 30 arcsecond (~1 km) grid, including all grid cells touched by a lake polygon.
Aridity	**Data source:** Trabucco and Zomer (2019) **Persistent web-link:** https://figshare.com/articles/dataset/Global_Aridity_Index_and_Potential_Evapotranspiration_ET0_Climate_Database_v2/7504448/3 **Temporal range:** 1970-2000 **Spatial resolution:** 30 arcsecond (~1 km) **Description and justification:** Aridity data from the Global Aridity Index and Potential Evapotranspiration Database that provides spatially distributed aridity index data based on the 1970-2000 period using the Penman-Monteith Reference Evapotranspiration equation. **Preprocessing:** None

Human modification gradient	**Data source:** Kennedy et al. (2019) **Persistent web-link:** https://figshare.com/articles/dataset/Global_Human_Modification/7283087 **Temporal range:** Median indicator year of 2016. **Spatial resolution:** 1 km **Description and justification:** A global representation of the degree of human modification made to terrestrial lands based on 13 stressors, that include human settlement, agriculture, transportation, mining, and energy datasets. To our knowledge, this is the most recent and comprehensive mapping of anthropogenic stressors to terrestrial lands available. **Preprocessing:** Reprojected from 1 km resolution in the Mollweide projection to WGS 84 and resampled at 30 arcsecond (~1 km) resolution using nearest neighbour cell value assignment.

Table VII.2. Filtering and preprocessing of protected areas.

WDPA full database (Downloaded: 7 June 2021)	
↙ 253,717 features ↘	
↓ WDPA features kept as **high levels of protection** *Criteria:* *IUCN class I-III or “Not Reported” or “Not Assigned” AND* *Reported area ≥1km^2*	↓ WDPA features grouped as **low levels of protection** *Criteria:* *IUCN class IV-VI or “Not Applicable” AND* *Reported area ≥1km^2*
54,600	48,017
↓ Contiguous protected areas (**High protection**) after conversion to 30 arcsecond format	↓ Contiguous protected areas (**Low protection**) after conversion to 30 arcsecond format
37,137	Not calculated as not needed
↓ Contiguous protected areas with **groundwater-dependent ecosystems**	↓ Contiguous protected areas with **groundwater-dependent ecosystems**
32,490 **The set of protected areas used for groundwatershed mapping**	Not calculated as not needed

Table VII.3. Metrics developed to assess patterns in groundwatersheds.

Indicator	Equation	Possible range
Relative groundwatershed size (**RGS**)	(Surface area of groundwatershed) / (Surface area of groundwater-dependent ecosystem within groundwatershed)	Minimum: 1 (groundwatershed area is completely aligned with ecologically connected area) Maximum: No maximum value
Underprotected groundwatershed ratio (**UGR**)	(Area of underprotected portion of groundwatershed) / (Total area of groundwatershed)	Minimum: 0 (groundwatershed is entirely within protected area) Maximum: 1^- (all but one grid cell of groundwatershed is underprotected)

VII.8 Supplementary references

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Appendix VIII

SUPPLEMENTARY INFORMATION FOR CHAPTER 8

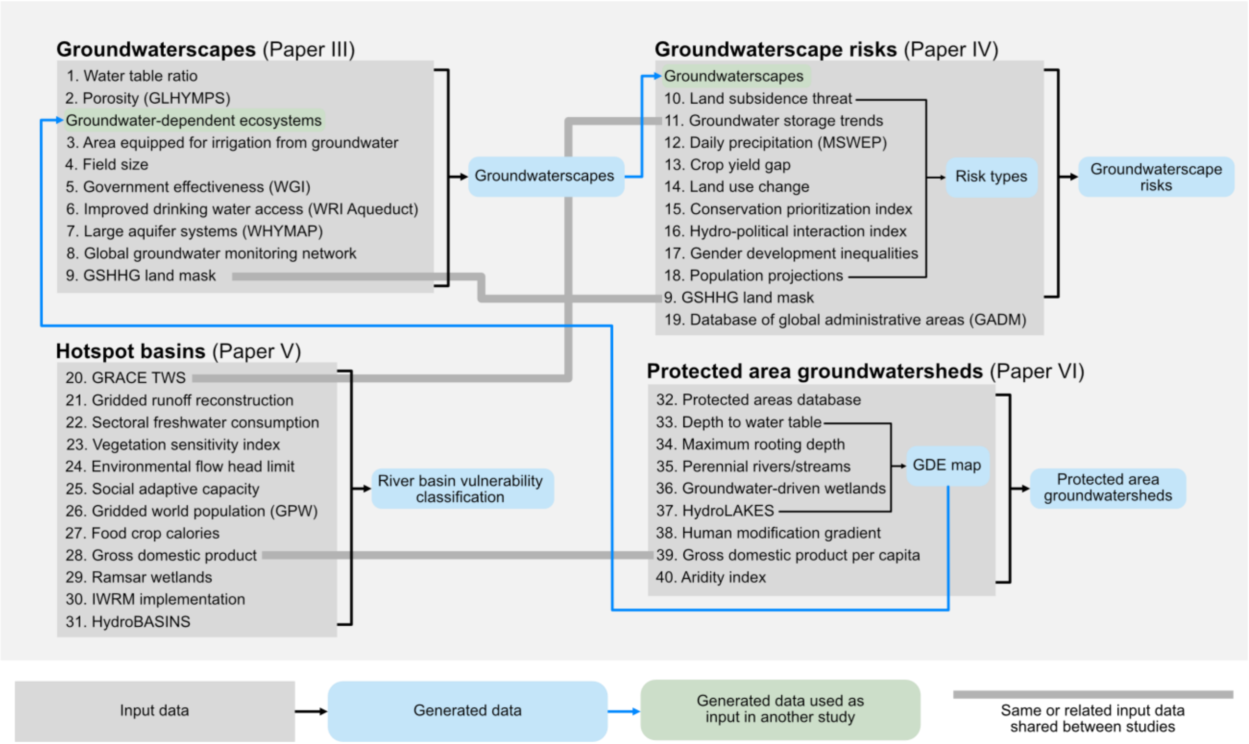

Figure VIII.1. Input and output datasets used and generated in the global-scale studies of this dissertation.