Part I

Integration Overview and Problem Settings

1 Integrated Groundwater Management: An Overview of Concepts and Challenges

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Abstract

Managing water is a grand challenge problem and has become one of humanity's foremost priorities. Surface water resources are typically societally managed and relatively well understood; groundwater resources, however, are often hidden and more difficult to conceptualize. Replenishment rates of groundwater cannot match past and current rates of depletion in many parts of the world. In addition, declining quality of the remaining groundwater commonly cannot support all agricultural, industrial and urban demands and ecosystem functioning, especially in the developed world. In the developing world, it can fail to even meet essential human needs. The issue is: how do we manage this crucial resource in

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an acceptable way, one that considers the sustainability of the resource for future generations and the socioeconomic and environmental impacts? In many cases this means restoring aquifers of concern to some sustainable equilibrium over a negotiated period of time, and seeking opportunities for better managing groundwater conjunctively with surface water and other resource uses. However, there are many, often-interrelated, dimensions to managing groundwater effectively. Effective groundwater management is underpinned by sound science (biophysical and social) that actively engages the wider community and relevant stakeholders in the decision making process. Generally, an integrated approach will mean “thinking beyond the aquifer”, a view which considers the wider context of surface water links, catchment management and cross-sectoral issues with economics, energy, climate, agriculture and the environment. The aim of the book is to document for the first time the dimensions and requirements of sound integrated groundwater management (IGM). The primary focus is on groundwater management within its system, but integrates linkages beyond the aquifer. The book provides an encompassing synthesis for researchers, practitioners and water resource managers on the concepts and tools required for defensible IGM, including how IGM can be applied to achieve more sustainable socioeconomic and environmental outcomes, and key challenges of IGM. The book is divided into five parts: integration overview and problem settings; governance; socioeconomics; biophysical aspects; and modelling and decision support. However, IGM is integrated by definition, thus these divisions should be considered a convenience for presenting the topics rather than hard and fast demarcations of the topic area.

1.1 Introduction

Managing groundwater has all the features of “wicked or messy” problems (Rittel and Webber 1973), which have multiple stakeholders and decision makers with competing goals, and where the systems of interest are complex, changing and multifaceted – having interactive social, economic, and ecological components – that are subject to a range of uncertainties caused by limited data, information and knowledge.

It is also a grand challenge problem in its severity, pervasiveness and importance. Stores of groundwater represent over 90 % of readily available freshwater on earth (UNEP 2008). However, historically, groundwater has been out of sight and thus underappreciated. Moreover, the time for groundwater system degradation to reach thresholds of concern, even if recognized, is typically longer than many timeframes used in societal decision making. As a result, despite its importance groundwater remains a minor player in water resources management. This relative inattention is changing. Groundwater usage surpasses surface water usage in many parts of the world, which is expected to increase further with advances in drilling and pumping. As well there is a growing awareness of the crucial connectedness of freshwater systems (Villhoth and Giordano 2007), and competition for all types of water has intensified across the globe, driven by the growing world population, and increased agriculture, industrial and economic development. Finally, the hidden nature of, and difficulty in characterizing, groundwater systems mean that once a groundwater system is degraded it is not quick, cheap, or easy to remedy. In this way a precautionary principle applies: an ounce of prevention truly may be worth a pound of cure.

The dependence of human and ecological communities on groundwater and their respective challenges varies substantially across the globe, but in no location is groundwater not utilized. The dependence of communities on groundwater can be seasonal or episodic; for example the resource may become critical to survival during severe drought when surface water resources run dry. There are countries, such as Belgium, Denmark, Saudi Arabia and Austria, where over 90 % of total water consumption is sourced from aquifers (Zektser and Everett 2004). However, on average, groundwater comprises approximately 20 % of the world's water use. In many humid regions, such as Japan and northern Europe, groundwater is mostly used for industrial and domestic purposes (Villhoth and Giordano 2007). In most countries outside the humid inter-tropical zone, groundwater is predominantly used for agricultural purposes, especially irrigation (Zektser and Everett 2004). Many large aquifers vital to agriculture, notably in India, Pakistan, Saudi Arabia, USA, China, Iran and Mexico, are under threat from overexploitation (Gleeson et al. 2012; Wada et al. 2012).

Where groundwater abstraction exceeds recharge over long periods and over extensive areas, the subsequent decline in watertable level affects natural groundwater discharge, which in turn may have harmful impacts on groundwater dependent streams, wetlands and ecosystems (Wada et al. 2010). Furthermore, lowered groundwater levels can reduce well yields and increase pumping costs, as well as lead to land subsidence on large scales (Konikow and Kendy 2005). The last can be particularly important. When sufficiently dewatered, accompanying aquifer compaction cannot be reversed, and no options are available to regain the lost aquifer storage. The groundwater in this case is truly "mined" and non-renewable. Partly due to its hidden nature, groundwater usage in many regions has been less monitored than surface water resources. Groundwater managers are typically "flying blind," especially in less advanced countries. Impacts of groundwater overexploitation and pollution can remain undetected for decades or even centuries, presenting further challenges for managing today's resource.

In addition to the poor scientific understanding of groundwater systems, other drivers of poor groundwater management practice have included suboptimal governance, short time horizons of management, and the resource being undervalued and underpriced. More practically, even seemingly small technology shortcomings such as the difficulty and lack of metering hinder implementation of integrated groundwater management. Declines in groundwater quality have also adversely affected use, reuse, and management efforts. As a result, the major threats to groundwater are multi-faceted. The wide range of interests that contribute to groundwater problems illustrates that groundwater issues are not a sector, state, or national issue, but a human issue. Given the complex nature of groundwater systems and their increasing importance as a source of water, there is broad consensus that an effective integrated approach to groundwater management is essential.

1.2 Integrated Groundwater Management

Integrated Groundwater Management (IGM) is viewed here as a structured process that promotes the coordinated management of groundwater and related resources (including conjunctive management with surface water), taking into account non-groundwater policy interactions, in order to achieve balanced economic, social welfare and ecosystem outcomes over space and time.

A valuable meta-discipline for such a process is that of integrated assessment (IA) (Risbey et al. 1996; Rotmans and van Asselt 1996; Rotmans 1998). IA is defined by The Integrated Assessment Society (www.tias-web.info) as "the scientific meta-discipline that integrates knowledge about a problem domain and makes it available for societal learning and decision making processes." Also "Public policy issues involving long-range and long-term environmental management are where the roots of integrated assessment can be found. However, today, IA is used to frame, study and solve issues at other scales. IA has been developed for acid rain, climate change, land degradation, water and air quality management, forest and fisheries management and public health. The field of Integrated Assessment engages stakeholders and scientists, often drawing these from many disciplines." In terms of water resource management, Jakeman and Letcher (2003) summarise key features and principles of IA (Table 1.1) and highlight the role of computer modelling in the process. The latter will be expanded upon in Part IV of this book. It is noteworthy that IA can bridge multiple topics; for example: although water and energy assessments are distinct threads in the IA literature, the meta-discipline offers a way forward to capture multiple issues and their interactions/interrelations.

Table 1.1 Common features of integrated assessment (Adapted from Jakeman and Letcher 2003)

A problem-focussed activity, needs driven; and likely project-based
An interactive, transparent framework; enhancing communication
A process enriched by stakeholder involvement and dedicated to adoption
Linking of research to policy
Connection of complexities between natural and human environment
Recognition of spatial dependencies, feedbacks, and impediments
An iterative, adaptive approach
A focus on key elements
Recognition of essential missing knowledge for inclusion
Team-shared objectives, norms and values; disciplinary equilibration
Science components not always new but intellectually challenging
Identification, characterisation and reduction of important uncertainties in predictions

To produce outputs that are useful for an intended purpose such as decision making, it is essential that IGM and IA address all important dimensions of integration. Below we discuss ten key dimensions of IGM based on a framework applied to Integrated Modelling proposed by Hamilton et al. (2015). These dimensions correspond to the integration of multiple, often disparate, topics: issues of concern; management options and governance arrangements; stakeholders; natural subsystems; human subsystems; spatial scales; temporal scales; disciplines; methods, models, tools and data; and sources and types of uncertainty. This book covers a wide range of challenges relating to groundwater management and the integration across and within the ten dimensions, as well as potential solutions to addressing such challenges.

1.2.1 Issues of Concern

IGM recognises that many issues are interrelated and thus cannot be solved in isolation. For instance, the modernisation of traditional gravity irrigation systems reduces groundwater recharge important for other uses; economic incentives (subsidies) provided by agricultural or energy policies can thus drive groundwater use. Similarly, policy interventions initially designed to solve a groundwater management problem may interfere (positively or negatively) with other policies or groundwater activity. For example, the enforcement of pumping restrictions to ensure that the sustainable use is not exceeded may lead to drastic changes in agricultural production and competiveness of a local agroindustry.

Clearly, addressing groundwater issues in isolation can inadvertently create or exacerbate other problems. Therefore, a joint assessment and treatment of issues across the policy sectors in Fig. 1.1 is important to avoid adversely offsetting actions. A holistic treatment of groundwater related issues is also needed to ensure that all stakeholder views are included and conflicts considered. The essence of IGM consists of clearly articulating and making trade-offs to limit adverse impacts and balance the needs and values associated with competing objectives. This process can involve selecting appropriate environmental, social and/or economic indicators as evaluation criteria, and using integrated assessment and modelling to assess the system performance under different scenarios (Hamilton et al. 2015).

1.2.2 Governance

The governance dimension of integration is ubiquitous yet is often a primary stumbling block to effective IGM. Groundwater governance comprises the promotion of responsible collective action to ensure control, protection and socially sustainable utilisation of groundwater resources and aquifer systems. This is facilitated by the legal and regulatory framework, shared knowledge and awareness of sustainability challenges, effective institutions, and policies, plans, finances and incentive structures aligned with society's goals (GEF et al. 2015). Governance can

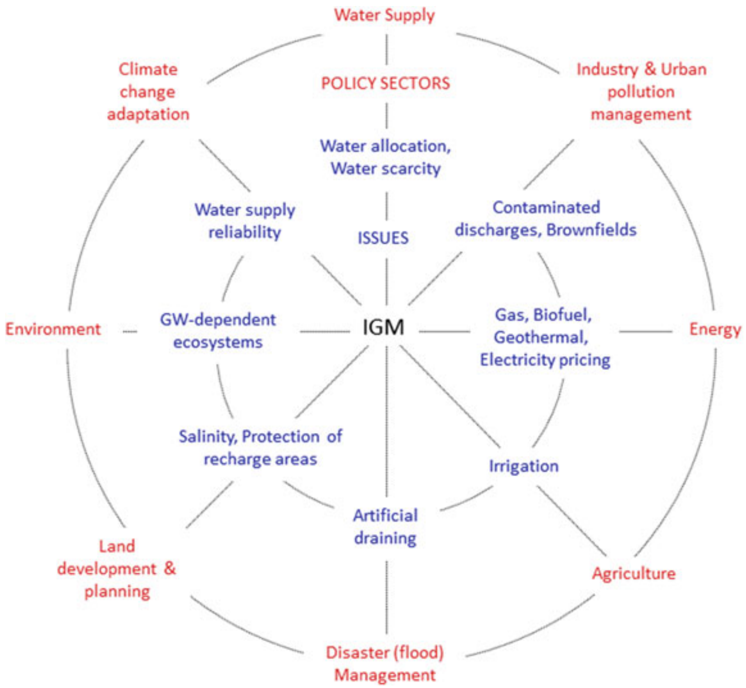

Fig. 1.1 Examples of diverse issues related to groundwater and their relevant policy sectors

be examined from various perspectives including institutional architecture, who is involved, and who is accountable for what to whom.

Such discussions include a mix of policy approaches, including the five types of instruments (Kaufmann-Hayoz et al. 2001):

- Command and control instruments such as regulatory standards, licences, and management zones; these tools aim to improve the behaviour of a target group through State intervention.
- Economic instruments such as taxes, subsidies or water markets, which influence micro-economic choices towards a desirable state, by influencing the costs and benefits of possible actions.
- Collaborative agreements which aim at strengthening cooperative behaviours between groundwater users, by enhancing non-economic motivations (altruism, reciprocity, trust, concerns for future generations)
- Communication and diffusion instruments, to distribute information aimed at influencing the knowledge, attitudes and/or motivations of individuals and their decision making (e.g. related to individual water consumption)
- Infrastructure instruments/investments, which describe the public sector investments intended to improve groundwater management such as those used to initiate managed aquifer recharge.

Ideally, decision makers should develop strategies and institutions that effectively combine these instruments to deliver acceptable environmental and socio-economic outcomes, and are also robust under potential changes to the natural and human settings (e.g., climate change, population increase). One of the main issues is ensuring the consistency of the interventions. Implementing one instrument may facilitate or inhibit the effectiveness of other instruments; it is important to consider possible synergies. IGM should provide a process for identifying intervention options and instruments and assessing their effectiveness under different scenarios. Groundwater governance is a complex process, where its effectiveness is influenced by challenges related to determining and implementing policies for groundwater allocation, and coordination of responsibilities across geographical, sectoral and jurisdictional boundaries.

1.2.3 Stakeholders

It is increasingly recognised that successful treatment of any wicked problem engages stakeholders appropriately. This particularly applies to groundwater management due to the invisible nature of the resource and the expense and related lack of high-quality information. Stakeholders are individuals or groups involved or interested in the problem – for example local/regional/national government, groundwater users, community groups, the water industry and those with relevant expertise (e.g. hydrologists, hydrogeologists, environmental modellers, agronomists, social scientists, ecologists, etc.). Though often avoided by groundwater scientists, the stakeholder engagement process is critical for effective IGM because it ensures that a broad range of interests, knowledge and perspectives are considered, shared and understood. Stakeholder engagement is also a valuable process in mutually educating, reducing conflict and building trust among researchers, decision makers and other stakeholders. Stakeholder engagement helps to develop a better understanding of demands on the resource and assimilates and publicizes scientific information used by managers. It also promotes mutual learning between users, managers, and policy makers in different domains (agriculture, water supply, energy, etc.). Perhaps most importantly, it can be considered as a necessary condition to gain acceptance of proposed management strategies needed for effective implementation by as many as hundreds or thousands of individual groundwater users. That is, those that are not included in the discussions about the groundwater resource are often those least likely to accept solutions proposed.

1.2.4 Human Setting

IGM operates within the human setting, including the social, political, cultural and economic characteristics of the stakeholders. One key role of groundwater managers is to make trade-offs between demand for water use and demand for groundwater sustainability. The demand for use is determined by prevailing market conditions and economic policies and to a lesser extent by societal values, including market conditions, policies and values concerning connected resources. The demand for groundwater sustainability and protection is determined by social drivers, including concerns for ecosystems and future generations. These drivers can in turn be influenced by the existing political context. Social drivers also shape the evolution of the institutional set-up, already described in the governance section above.

To effectively management groundwater systems it is necessary to understand how the human setting directly and indirectly relates to the groundwater system. This includes human responses to management interventions and other drivers like climate, and the socioeconomic impact of reduced access to groundwater or reduced groundwater quality. The human setting also underlies behavioural and socioeconomic factors that influence the adoption of better practices or new technologies identified by IGM.

1.2.5 Natural Setting

Most importantly, the natural setting forms the extent, limits, and service area of the natural resource from which all IGM must stem. This dimension relates to the integration and communication of the relevant scientific underpinnings and biophysical components of the system. The natural setting includes any substantive connection between aquifers and other natural features such as rivers, lakes, wetlands and springs. It also includes intra-aquifer connectivity within heterogeneous aquifers and inter-aquifer connectivity in multi-aquifer groundwater systems. The natural setting may encompass non-freshwater resources; the hydraulic connection between groundwater and the sea can be important as in estuary health and saltwater intrusion into pumping centres. IGM can also include joint consideration of groundwater and surface water systems with climate, vegetation, fauna and soils. It is increasingly being recognised that these compartments cannot operate or be managed in isolation, as demonstrated by the recent greater demand for conjunctive management of surface and groundwater resources.

1.2.6 Spatial Scales

The biophysical and socioeconomic processes related to groundwater systems occur at different spatial scales, ranging from global and regional scales (e.g. climate processes) down to the local scale (e.g. practices of individual farmers, endangered species restricted to a single spring, drinking water well protection zone). A single groundwater system can range from less than 10 km^2 to over 100,000 km^2 in size, and processes can operate at vastly different scales depending on the system. Biophysical processes can also operate at very different scales and boundaries than socioeconomic processes because groundwater flow is driven by gravity, not political boundaries. One of the key challenges of integrated assessment and modelling is accommodating the multiple spatial scales of system processes and interests. The stakeholders may also focus on scales that differ from the actual system processes, for example policy makers might have to develop strategies for groundwater management at a state or national level. Process upscaling/downscaling is commonly required to resolve potential mismatch of scales in integrated assessment frameworks. In many cases, mixed spatial scales are needed depending on what part of the system is represented.

1.2.7 Time Scales

Temporal aspects also operate at different scales – as might be expected when groundwater system processes typically occur over much longer time than human timeframes. The mismatch of temporal scales in IGM presents a considerable challenge in characterizing, understanding, and communicating aspects of groundwater systems, as well as how to manage them. Cause and effect may not be readily apparent due to substantial time lags between an action and its result; for example in some systems the effects of overexploitation of groundwater or poor land management may not be apparent in streamflow quantity or quality for several years or even decades. Similarly, even if extraction is reduced to sustainable limits, it may take decades before the effects are noticeable at land surface. Accurately attributing the effect of disturbance or management is further complicated by other confounding disturbances in the intervening period (e.g. extreme climate) and legacy effects from past practices (e.g., aquifers with low hydraulic diffusivity). The appropriate choice of time horizon (extent) and time step (resolution) is ultimately driven by the purpose of the IGM activity, and typically is selected to ensure important processes and responses can be captured.

1.2.8 Disciplines

To provide a holistic understanding of the system, IGM typically requires integration of knowledge and competencies from a broad range of paradigms (e.g. positivistic, interpretive) and disciplines (e.g., geology, hydrogeology, hydrology, hydrochemistry, engineering, ecology, law, economics, computer science, sociology, political science and psychology). Integrating disciplines involves challenges associated with incorporating divergent views and interests, theories, assumptions, types and formats of information, languages, research methodologies and tools (e.g. Hunt and Wilcox 2003; Hancock et al. 2009). IGM calls for a new breed of research, one focused on teams who are much more interdisciplinary and systems focused in their approach. Moreover, the interdisciplinary focus requires investments of time to communicate and understand points of view outside of one's field of expertise.

1.2.9 Methods, Models, Other Tools and Data

This dimension relates to the technical integration of different methods, models, tools and data from various disciplines and/or representing different processes or perspectives. There is a wide range of modelling and analytical tools that can be integrated to develop a comprehensive framework to facilitate IGM – both for the groundwater system itself as well as the socioeconomic drivers that act on the groundwater system. Integrated modelling is the common platform used for performing integrated assessment as it can support a systematic and transparent approach to integration (see Sect. 1.3 below). Combining diverse tools and data is a challenge in interfacing, interoperability, and appropriate distribution of limited available resources and effort. Such challenges have been the focus of work involving model and data standardisations and information exchange, work that is ongoing.

1.2.10 Uncertainty

No environmental system (natural and/or socioeconomic) can be perfectly characterized, especially when many of its key characteristics are inferred and imperfectly sampled. Handling the lack of detailed understanding of groundwater systems is one of the key challenges to their effective management. Uncertainty is embedded in all aspects of IGM, from our ability to represent the biophysical systems to the social systems in which they are embedded. Though the system cannot be perfectly characterized, the presence of uncertainty is well accepted and thus cannot be ignored. Effective IGM recognizes the source, nature and level of uncertainties associated with problem definition, social/political context, communication, and models and tools used in the assessment process. Due to the inherent and often large uncertainties associated with managing groundwater systems, there is a need to communicate decision making in the context of uncertainty and, when possible, develop robust management strategies that perform well under a range of plausible conditions.

1.3 Integrated Assessment, Modelling, and Other IGM Tools

Many tools can be used to support the development of policies in IGM. The development of conceptual models amongst stakeholders is a common starting point to frame the relevant issues, define outcomes, and manage complexity. A vital first step is to draw system boundaries wide enough to encompass the interacting influences, while keeping the conceptualisation only as complex as necessary to conduct useful analysis (Bazilian et al. 2011). Integrated models are generally considered the primary tool to articulate and test such conceptualisations because they can represent potential scenarios of policy interventions, uncontrollable drivers and uncertainties, and outputs that capture trade-offs or impacts of alternative actions. When properly constructed, they can also allow exploration of system feedbacks and linkages within a single framework. Because IGM encompasses a wide variety of drivers, feedbacks and spatio-temporal scales, integrated models that couple component models representing different system components (often from different paradigms) are often required (Kelly et al. 2013). For example, in exploring the socioeconomic and ecological impacts of reduced water allocations and adaptation options by farmers, Jakeman et al. (2014) developed an integrated model that couples surface-groundwater models with social Bayesian networks, crop metamodels, economic optimisation of production values, policy rule models, and ecological expert opinion. On the other hand, integrated models typically include one modelling methodology (e.g. Bayesian networks, system dynamics, agent-based models, expert systems) rather than a combination to represent the whole system. Including multiple methods is a topic of ongoing work.

The nature of integrated assessment, including the need to integrate perspectives from different disciplines and stakeholder groups, requires a process and modelling framework that is adaptive and facilitates participatory procedures. Often there is a flow of information from stakeholders on their knowledge of the system and preferences about the policy environment. This information, along with scientific knowledge, supports the conceptualisation, construction, and use of a model (Fig. 1.2). Model conceptualisation includes elements such as issue definition, specification of system boundaries and identification of measures, criteria, indicators and processes. The model, in turn, provides insight on the possible impacts and trade-offs under selected scenarios, which then flows back to inform stakeholder and policy preferences and system understanding. Scientists gain understanding from the modelling process as well through their interactions with stakeholders.

There are several important considerations handled when constructing integrated models. The purpose of the model drives the selection of system

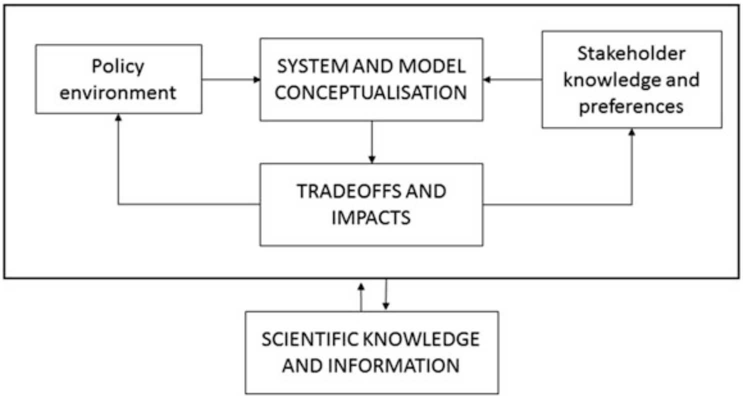

Fig. 1.2 Integrated modelling framework

processes, which in turn dictates the model structure that is applied and evaluated (Jakeman et al. 2006). Appropriate modelling takes into account the spatiotemporal detail required in the modelling, the nature of the data (qualitative and/or quantitative), the level of ability to represent uncertainty and feedbacks (Kelly et al. 2013). The choice of approach may also be dictated by human and computational resources. For example, Bayesian networks may be suitable when data is sparse or system understanding is limited but quickly interrogated; and process-based models may be suitable if system processes are understood and important for the IGM activity. The system dynamics approach may be appropriate when dynamic processes or system feedbacks are of interest, whereas agent-based models are appropriate when interactions between individuals are of interest (Kelly et al. 2013). Scenario analysis is useful when future conditions are difficult to estimate and underpin overarching uncertainty (e.g. climate change – See Anderson et al. 2015, Chap. 10). In summary, integrated assessment and modelling is often best supported by a suite of tools, with individual tools applied to leverage different information that is then compiled to provide an encompassing assessment of the system. The challenge is then ensuring effective communication between tools.

The outputs of integrated models are not a crystal ball defining one future. Rather, they are typically a heuristic tool that provides insights to support decision or policy making, a tool that articulates the trade-offs inherent to IGM. When properly used, these tools facilitate IGM through: (1) improving and articulating understanding (regarding potential impacts as well as system feedbacks and interactions); (2) educating scientists, decision- and policy-makers and other stakeholders; (3) limiting options explored to those that are feasible; and (4) building interaction and rapport between stakeholder groups, which can influence the range of policy changes considered.

1.4 Book Overview and Key Messages

The book is divided into five parts. An overview of each part and associated key messages are provided below.

1.4.1 Part I: Integration Overview and Problem Settings

This first part of the book provides a broad examination of integrated groundwater management and associated issues and challenges. As we have seen in Chap. 1, Integrated Groundwater Management is a grand societal challenge, perhaps the most urgent as many societies and ecosystems depend on the sustainability of their groundwater systems. Effective IGM considers the dimensions discussed in Sect. 1.2, and the effectively tailors the wealth of model platforms and tools available to support IGM to a specific problem context. Scientists and decision makers need to engage extensively with stakeholders and think and plan for the longer term inherent to all groundwater systems. Chapter 2 examines the international scale of groundwater issues, both in severity and extent. It points to the need for understanding the interconnections among aquifers, surface water, ecosystems, and human needs, especially given the complexities of social-ecological systems dependent on the resource. Chapter 3 discusses the interactions within components of groundwater-dependent and social-ecological systems, and proposes a conceptual framework to describe their complexity.

Chapters 4 and 5 examine the challenge of groundwater management under global change. Chapter 4 focuses on the water-energy-food nexus whereas Chap. 5 considers potential climate change impacts on groundwater, in addition to potential feedbacks of groundwater on the global climate system. Energy demand management measures have positive synergies in reducing consumption of water, but the impacts of new energy technologies on groundwater are mixed. The direct impacts of climate change on groundwater will vary with different combinations of soils/aquifer materials, vegetation, and climatic zone. Long-term monitoring of natural systems (groundwater, surface water, vegetation and land use patterns) provides a critical baseline to identify and evaluate effects of future change. Climate change mitigation and adaptation policies are expected to change, and in some cases (carbon sequestration in the landscape, some renewable energy technologies) exacerbate, the challenges associated with groundwater use and management.

1.4.2 Part II: Governance

Here six chapters deal with issues related to the governance of groundwater, focused on three case study regions: Australia, the European Union and the USA. It begins in Chap. 6 with a comparative study of groundwater governance in the three regions, classifying groundwater governance issues into the five blocks used in the Earth Systems Governance Framework. Strengths and weaknesses are elucidated as well as the governance difficulties and dilemmas faced in these three regions. A review of the fundamental legal principles relating to groundwater in the three regions, including the challenges of these legal frameworks in a cross-boundary context is discussed in Chap. 7. Australia, the western United States, and Europe display key differences in how they conceive of fundamental aspects of groundwater regulation, such as ownership and principles for permitting groundwater withdrawals. Yet they face very similar challenges in relation to integrating regulation of groundwater and surface water, groundwater and dependent environments, and groundwater across boundaries. Commonly, they deal with similar challenges in different ways, where a range of potential legal tools are used across the globe. In Chap. 8, groundwater challenges are examined through integrated management and planning approaches, with specific examples of policy frameworks for water management adopted in parts of the three study regions. From these examples, integrated groundwater management appears a “living” or iterated mechanism that is updated, refined and (if necessary) changed as new information and experience are gained. Chapter 9 explores the opportunities and challenges of delivering conjunctive management of water resources through collective action by governments and water users. Australia, Spain and the United States have made some progress in pursing conjunctive management through collective action, but their experiences have highlighted a number of practical and policy limitations. Conjunctive management through collective action is more likely where social and environmental crisis arise and where there is institutional recognition of hydrological connections (between groundwater and surface water), and where management tools are devolved to local water users.

Groundwater governance challenges, and associated potential social and environmental injustices, are addressed in Chap. 10, including how equity in water use is considered and how it has been translated into practice. The rationale for sharing or allocating groundwater is guided by the principle of *equitable and reasonable utilization*. Environmental justice is a useful lens in the arsenal of researchers, policy makers and natural resource managers that can be used to highlight the importance of a systems approach when dealing with common pool resources such as groundwater. In the last Chap. (11) of Part II, social justice and different groundwater allocation rules are contrasted in a French case study. It analyses the acceptability of rules for apportioning groundwater resources among agricultural users in over-used / over-allocated groundwater basins. The study highlights that acceptance of new water allocation rules is not only determined by how stakeholders perceive these rules in terms of distributive justice. Farmers' judgment is also influenced by their perception of the legitimacy (moral, pragmatic and cognitive) of the policy in which the question of allocation rule is embedded. Another determining factor is the perceived implementation difficulties that are expected to result from allocation rules.

1.4.3 Part III: Biophysical Aspects

The biophysical aspects of IGM are examined in Part III. It begins with a background to ecohydrology in Chap. 12, which considers how ecology and hydrology interactions are critical for determination of groundwater availability and sustainability, and once articulated, can be incorporated into effective groundwater management. In many cases, success of integrated groundwater management is measured by how well the interaction between ecology and hydrology aspects is articulated and addressed. Groundwater dependent ecosystems (GDEs), their structure and function, are reviewed in Chap. 13, and are discussed in terms of the potential threats resulting from over-extraction of groundwater. Defining the response function of ecosystems to groundwater extraction is a key research challenge for the future, with major implications to policy, legislation and sustainable management of GDEs and groundwater resources. Chapter 14 uses examples to illustrate how natural and anthropogenic water quality issues can drive IGM and its implementation – factors that can in some cases eclipse water quantity issues that may also exist. Water quality concerns can come from naturally occurring or human induced contaminants; moreover, such concerns are often based on public perception, which can limit the use and availability of groundwater. In this way, "acceptable" water quality is not a static definition, but changes with time with increasing analytical precision and increased knowledge on effects on human and environmental health. Chapter 15 examines the processes and issues around salinization and drainage in irrigation schemes. As the salinization of shallow aquifers is closely related to root-zone salinization, the two are considered together. A case study of root-zone salinization was taken from a developing country (Pakistan), whilst that of shallow aquifer salinization was taken from a developed country (Australia). Both case studies underscore how mitigation strategies to overcome groundwater salinization need to be integrated with policy.

In Chaps. 16 and 17 the promise and challenges of managed aquifer recharge (MAR) are explored, including opportunities to save excess water underground and reduce evaporation losses. MAR can augment groundwater with available surface water and can act alongside conjunctive use of surface waters and groundwater to sustain water supplies and achieve groundwater and surface water management objectives such as protection of ecosystems. Chapter 16 argues that specific local characteristics of each MAR site, precludes a single universal solution for all settings, suggesting existing legal frameworks must take this into account. Moreover, MAR function and the impacts on water availability, water quality, sustainability as well as on the local and downstream environment, need to be communicated to promote cost-effective implementation. Chapter 17 further describes the potential role of MAR in IGM for conserving surface water resources, improving groundwater quality and increasing groundwater availability. MAR may be used to replenish depleted aquifers, in association with demand management strategies to bring aquifers closer to hydrologic equilibrium needed for sustainable use. In suitable hydrogeologic locations, MAR options have been shown to be economic when compared to other sources such as seawater desalination.

1.4.4 Part IV: Socioeconomics

Part IV focuses on the social science and economic considerations of IGM. Chapter 18 examines groundwater management in modern-day China, which is facing unprecedented challenges that reflect many social, cultural and political drivers. The chapter examines how changes to the legislation system, institutional reforms and better management instruments can help China progress towards more integrated groundwater management. Chapter 19 explores the social dimensions of groundwater governance and how social sciences, including stakeholder engagement, social impact assessment and collaborative approaches, contribute to the IGM process. Difficult or 'wicked' natural resource management issues are often best addressed by engaging stakeholders in processes that involve dialogue, learning, and action to build and engage social and human capital. Human and social capital underpins much of the capacity of any community to respond to the challenges of sustainability. When conducting integrated research, it is critical for social researchers to be engaged from the outset in problem definition and setting research priorities.

In Chap. 20 the use of groundwater trading as a management strategy is investigated, where attempts in Australia and the USA to establish groundwater markets are used to frame important underlying issues. Before groundwater markets can successfully develop, institutions and regulations have to exist at some level. For fully efficient and effective policy, there is a need to invest in high quality economic and scientific research, where social concerns are not the sole important drivers for efficient and effective groundwater markets. In Chap. 21, assessment of the benefits of groundwater improvement and protection is addressed from an economic viewpoint of contingent valuation. Such economic analysis integrates benefits for present and future generations, and includes the “bequest” or “heritage” value, defined as the value of satisfaction from improving groundwater resources for future generations. Potential and limits to this approach are discussed using literature review and two case studies from France and Belgium. Chapter 22 evaluates strategies for groundwater management through economic instruments, current practices, challenges and innovative approaches. The last Chap. (23) of Part IV examines the expanding groundwater economy in North Africa, where aquifers have commonly been overexploited as a result of the short-term interests of private entities and the absence of effective governance.

1.4.5 Part V: Modelling and Decision Support

Lastly, Part V focuses on concepts of modelling, data management, and decision support for facilitating and informing IGM. Chapter 24 discusses the use of systems thinking, particularly soft- and critical-systems approaches, for incorporating human aspects (i.e. cognitive, social, cultural, and political) into groundwater management and research. It stresses the value of a multi-method approach to accommodate different perspectives using four international case studies, and suggests that practitioners and researchers need to be aware and explicit about their theoretical and methodological stance, but also creative about how they adapt and localise their approaches. Chapter 25 examines the use of decision support processes and models for articulating and improving groundwater management policies and trade-offs. Decision support systems (DSS) provide a means for water managers to evaluate complex data sets that include hydrogeologic, economic, legal and environmental elements. Although distributed groundwater modelling approaches are improving, examples of integrated groundwater DSS or participatory processes are not widespread. Nevertheless DSS are well suited for integrated groundwater problems because they can provide a set of applications, methodologies, and tools to communicate and cope with inherent complexity and uncertainty.

Chapter 26 discusses challenges that ripple to data management needed for IGM as new technologies in monitoring and computing, including data networks, are developed. Integrated studies typically have large data requirements, which not only need to be well stored, but also well described, easily discoverable and accessible, and in consistent form for use in integrated groundwater studies. Data networks are increasingly being used to provide access to large national data holdings in a consistent open standards based manner, which facilitates their use in integrated groundwater studies. Chapter 27 reviews the use of hydro-economic models as decision support tools for conjunctive management of surface and groundwater. It considers technical challenges involved in incorporating aquifer dynamics, stream-aquifer interactions, nonlinearities and multiple objectives into integrated frameworks. Hydroeconomic models can provide a useful insight into a more efficient operation of conjunctive use and the economic implications of different conjunctive use strategies. The final Chap. (28) relates IGM to uncertainty – uncertainty that resides in managing groundwater systems and in groundwater system models. A range of methods for exploring uncertainties and how they can be applied are discussed. Because no one approach is appropriate for all applications, techniques are often decided by the judgement of the modeller. As the scientific method cannot prove correctness, making predictions of uncertain outcomes needs to focus on eliminating the impossible and incorrect potential outcomes, and focus on elucidating alternative models and conclusions. One does not need to be able to use all possible alternatives, but it is important to be aware of alternatives that have not been used but could affect associated conclusions.

And perhaps one final message is warranted. Difficult problems and crises involving groundwater will only increase. Opportunities for IGM will then operate on two levels, the first being steadfast application of standard approaches to problems well recognized. Less predictable, come windows of opportunities for reform and more effective IGM. The challenge for all parties – decision-makers, water managers, scientists and other stakeholders – is to be prepared to seize opportunities to implement more sustainable and effective groundwater management. The aim of this book was to prepare the reader for such windows of opportunity by laying out the major disciplinary and interdisciplinary components, challenges, and opportunities, for integrated and sustainable management of groundwater.

References

Anderson MP, Woessner WW, Hunt RJ (2015) Applied groundwater modeling: simulation of flow and advective transport. Elsevier Academic Press, Amsterdam, 564 p. ISBN 9780120581030

Bazilian M, Rogner H, Howell M, Hermann S, Arent D, Gielen D, Steduto P, Mueller A, Komor P, Tol RSJ, Yumkella KY (2011) Considering the energy, water and food nexus: towards an integrated modelling approach. Energy Policy 39:7896–7906

GEF, World Bank, UNESCO-IHP, FAO and IAH (2015) Global groundwater governance a call to action: a shared vision for 2030. http://www.fao.org/fileadmin/user_upload/groundwater governance/docs/general/GWG_VISION.pdf. Accessed 11 June 2015

Gleeson T, Wada Y, Bierkens MFP, van Beek LPH (2012) Water balance of global aquifers revealed by groundwater footprint. Nature 488:197–200

Hamilton S, El Sawah S, Guillaume JHA, Jakeman AJ (2015) Integrated assessment and modelling: a review and synthesis of salient dimensions. Environ Model Software 64:215–229

Hancock PJ, Hunt RJ, Boulton AJ (2009) Preface: hydrogeoecology, the interdisciplinary study of groundwater dependent ecosystems. Hydrogeol J 17(1):1–4. doi:10.1007/s10040-008-0409-8

Hunt RJ, Wilcox DA (2003) Ecohydrology – why hydrologists should care. Ground Water 41(3): 289. doi:10.1111/j.1745-6584.2003.tb02592.x

Jakeman AJ, and 17 others (2014) Modelling for the complex issue of groundwater management. In: Obaidat MS et al. (eds) Simulation and modelling methodologies, technologies and applications, vol 256, Advances in intelligent systems and computing. Springer, London, pp 25–41. doi:10.1007/978-3-319-03581-9. ISSN 2194–5357, ISBN 978-3-319-03580-2

Jakeman AJ, Letcher RA (2003) Integrated assessment and modelling: features, principles and examples for catchment management. Environ Model Software 18:491–501

Jakeman AJ, Letcher RA, Norton JP (2006) Ten iterative steps in development and evaluation of environmental models. Environ Model Softw 21:602–614

Kaufmann-Hayoz R, Battig C, Bruppacher S, Defila R, Di Giulio A, Flury-Kleubler P, Friederich U, Garbely M, Gutscher H, Jaggi C, Jegen M, Mosier H-J, Müller A, North N, Ulli-Beer S, Wichtermann J (2001) A typology of tools for building sustainable strategies. In: Kaufmann-Hayoz R, Gutscher H (eds) Changing things – moving people. Birkhauser, Basel, p 33e108. Kelleher C, Wagener T, 2011

Kelly RA, Jakeman AJ and 11 others (2013) Selecting among five common modelling approaches for integrated environmental assessment and management. Environ Model Softw 47:159–181

Konikow LF, Kendy E (2005) Groundwater depletion: a global problem. Hydrogeol J 13:317–320

Risbey J, Kandlikar M, Patwardhan A (1996) Assessing integrated assessments. Clim Change 34: 369–395

Rittel HWJ, Webber MM (1973) Dilemmas in a general theory of planning. Policy Sci 4:155–169

Rotmans J (1998) Methods for IA: the challenges and opportunities ahead. Environ Model Assess 3:155–179

Rotmans J, van Asselt M (1996) Integrated assessment: a growing child on its way to maturity. Clim Change 34:327–336

UNEP (2008) Vital water graphics – an overview of the state of the world's fresh and marine waters, 2nd edn. UNEP, Nairobi. ISBN 92-807-2236-0

Villhoth KG, Giordano M (2007) Groundwater use in a global perspective – can it be managed? In: Giordano M, Villholth KG (eds) The agricultural groundwater revolution: opportunities and threats to development, vol 3, Comprehensive assessment in agriculture. CAB International, Wallingford, pp 393–402

Wada Y, van Beek LPH, van Kempen CM, Reckman JWTM, Vasak S, Bierkens MFP (2010) Global depletion of groundwater resources. Geophys Res Lett 37, L20402. doi:10.1029/2010GL044571

Wada Y, van Beek LPH, Bierkens MFP (2012) Nonsustainable groundwater sustaining irrigation: a global assessment. Water Resour Res 48:W00L06. doi:10.1029/2011WR010562

Zektser IS, Everett LG (eds) (2004) Groundwater resources of the world and their use, IHP-VI series on groundwater No 6. UNESCO, Paris

2 The International Scale of the Groundwater Issue

Michael N. Fienen and Muhammad Arshad

Abstract

Throughout history, and throughout the world, groundwater has been a major source of water for sustaining human life. Use of this resource has increased dramatically over the last century. In many areas of the world, the balance between human and ecosystem needs is difficult to maintain. Understanding the international scale of the groundwater issue requires metrics and analysis at a commensurate scale. Advances in remote sensing supplement older traditional direct measurement methods for understanding the magnitude of depletion, and all measurements motivate the need for common data standards to collect and share information. In addition to metrics of groundwater availability, four key international groundwater issues are depletion of water, degradation of water quality, the water-energy nexus, and transboundary water conflicts. This chapter is devoted to introducing these issues, which are also discussed in more detail in later chapters.

2.1 Introduction

Throughout history, groundwater has been a major source of water for sustaining human life. Because it is buffered from short-term variability in weather patterns, groundwater has often been considered a stable and reliable resource. With the advent of efficient pumps and rural electrification, global groundwater extraction increased from 312 km^3/year in the 1960s to 743 km^3/year in 2000 (Wada et al. 2010); approximately 70 % of this extraction is used for agriculture. About half of domestic human water consumption in urban areas is from groundwater

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(Giordano 2009). With increased water use comes a related possibility of local, regional, and international conflict over groundwater resources.

Groundwater, surface water, humans, and ecosystems are all interconnected in ways that necessitate an integrated approach to management. To manage in this way requires an understanding not only of the component aspects of the problem but also of the components' interconnections. See Chap. 1 for a comprehensive list and description of the dimensions of an integrated approach. Determining the scope of these issues, a first dimension, is challenging on a global scale, primarily because groundwater systems themselves are not all connected, and each system has its own characteristics; thus, any measurements of a specific system reflect specific local conditions, making extrapolation from data-rich to data-poor regions problematic. In contrast to measurements of streams which can integrate information over an entire watershed, point measurements of groundwater conditions commonly reflect a smaller land area, requiring more measurements to evaluate a comparable region. Remote sensing techniques such as the Gravity Recovery and Climate Experiment (GRACE) satellite provide information over larger areas, but they also require site-specific calibration information and are more accurate for determining changes than for assessing conditions at a certain time. A realistic picture of global conditions, then, must be based on aggregation of data from a variety of widely distributed organizations, many of them local in focus. These data must also be used with modeling techniques to obtain estimates of groundwater conditions.

Once information from observations and models is assembled, metrics that allow comparison among regions can be developed to guide management. These metrics are typically based on water balance computations, which in turn are based on estimates of human extraction and returns, removal from storage, water required for ecosystem services, and natural replenishment. The management challenge then becomes making the difficult choices regarding the level of sustainability required, because the relationship of humans to groundwater resources differs from place to place.

Four key international groundwater issues are depletion of water, degradation of water quality (see also the devoted coverage in Chap. 15), the water-energy nexus (Chap. 4), and transboundary water conflicts (Chap. 6). In the context of these issues, technical challenges abound in attempting to understand and quantify current impacts and resources, even more so in attempting to plot a way forward. Yet, some advances in understanding are being made, and common threads of challenges related to scale, governance, and the need for integrated data also provide opportunities to impact multiple issues with each advance.

Depletion is a major groundwater issue, but the definition of depletion is not completely obvious and has changed over time. Dating back to 1915, concepts of *safe yield* in relation to groundwater were proposed. Originally, a balance was sought between groundwater extraction and replenishment by recharge such that extraction could continue in equilibrium. This early definition did not incorporate transient conditions, nor did it consider ecosystem impacts (as covered in Sect. 2.3, Chaps. 12 and 13). The concept of depletion has since evolved into one that acknowledges sustainability and integrated water management, but a true accounting of depletion also must embrace socioeconomic considerations (as covered in

Sect. 2.4, and Chaps. 20 and 21). Depletion is still typically measured by decreases in groundwater levels and decreases in baseflow or levels in connected surface water bodies and degradation in water quality.

Degradation of water quality falls into two broad categories (Chap. 14): that due to natural conditions and that due to anthropogenic causes. Both forms of degradation can result from human extraction of groundwater. Extraction or changes to recharge can alter groundwater flow directions or expose aquifer material to air, allowing for previously clean water to encounter natural contaminants such as radium, salt, arsenic, and fluoride and resulting in poor water quality and associated health impacts. On the other hand, chemical and biological contaminants emanating from industry and agriculture also cause water quality degradation.

As expounded in Chap. 4, the water-energy nexus is an integrative issue with feedbacks among water extraction, water quality, and energy production/consumption. Declining water levels due to extensive extraction lead to increased lift required by pumps, thereby increasing the amount of energy required for irrigation and domestic use. Exploration for new energy sources—for example, shale gas—also has the potential to create groundwater contamination from various activities associated with its production, such as during hydraulic fracturing and deep disposal of drilling fluids.

Transboundary aquifers (Chap. 6) have often been cited as potential hotspots of global conflict. Many aquifers are bounded by the borders of a single country so, whereas internal conflicts arise and can be substantial, they are less likely to be violent than conflicts between nations. Exceptions include the Nubian Aquifer in North Africa and aquifers in the Israel/Palestine region. Conflicts less intense than war nonetheless occur within nations at scales ranging from individual ranches to larger regions. Dire predictions of wars over groundwater resources have been made for many years, and although some violence has occurred, extraordinary cooperation has sometimes been motivated by mutual need for groundwater resources. Uncertainties related to groundwater resources—in contrast with surface water systems—may increase the likelihood of future conflicts.

In this chapter, we explore each of these integrated issues more deeply. We also discuss technologies and techniques for better understanding them. The goal is to highlight the need for integrated management and to set a conceptual framework for the discussion and potential solutions described in more detail throughout the book.

2.2 The Concept of Groundwater Depletion

When evaluating the international scale of the groundwater issue, it is important to establish what makes groundwater an issue in the first place. Understanding concepts of sustainability, safe yield, and depletion are central to this. These concepts guide definitions of where groundwater stresses are important.

A parallel evolution in thinking has occurred in the last 100 years regarding (1) the connections between surface water and groundwater and (2) the importance of water provided to ecosystems. Despite previous misconceptions of "safe yield" (for example, using calculations of recharge as a basis for allotting an amount of water that can be "safely" extracted from a groundwater basin), it has become more widely accepted that discharge to streams, springs, etc., is often the limiting water balance element. With regard to ecosystem services, the concept of "safe yield" has evolved to "sustainability," augmenting consideration of undesirable economic impacts of depletion with the maintenance of discharge flows at levels that support ecosystem dependence on surface water and groundwater from aquifers.

As early as 1915, the term "safe yield" (Lee 1915) of a groundwater basin was used to define "the net annual supply which may be developed by pumping and Artesian flow without persistent lowering of the ground-water plane." Subsequent work (Todd 1959) made a more general definition as "the amount of water which can be withdrawn from it annually without producing an undesired result." Two important aspects of this definition warrant further scrutiny.

First, the specific source of water needs to be understood to evaluate whether withdrawals are balanced with sources. In Lee's original definition, the entire water balance was considered, and it was acknowledged that often the source of water to pumping wells is the interception of discharge to surface water bodies rather than the collection of recharge. The early workers (Lee 1915) stated that "It is obvious that water permanently extracted from an underground reservoir, by wells or other means, reduces by an equal quantity the volume of water passing from the basin by way of natural channels." Work by Theis (1940) and others also highlighted the importance of intercepted discharge to surface water or evapotranspiration as more significant than collection of recharge. However, over time, the importance of intercepted discharge was neglected and focus on balancing recharge with pumping became a popular definition of safe yield—including codification in legislation in some parts of the United States (Sophocleous 1997). This oversimplified concept has been called the "water budget myth" (Bredehoeft et al. 1982; Bredehoeft 1997; Sophocleous 1997).

Conservation of mass is a tenet of science, formally dating back to 1748 (Hockey et al. 2007), so the establishment of water budgets is a natural approach to assessing groundwater availability. Simply by accounting for inputs (through recharge and regional flow) and outputs (natural discharge to surface water, evapotranspiration, and anthropogenic extraction) and the change in storage, the amount of available groundwater can be established. Prior to pumping, the groundwater system is typically in dynamic equilibrium, with storage being constant and the sum of all inputs equal to the sum of all outputs. If a new stress acts on the system, either recharge must increase, discharge must decrease, or water must be removed from storage. It is uncommon for pumping to be accompanied by an increase in recharge from precipitation, so the change must result from some combination of a decrease in discharge or removal of water from storage. As water is removed from storage, the groundwater surface—the water table in unconfined aquifers or the potentiometric surface in confined aquifers—drops, which can increase the cost and difficulty of removing water through pumping. Through a dropping water surface, directly intercepted discharge, or a combination of those two effects, streams and springs can be reduced in flow or completely dried up. Removal of water from storage is referred to as "mining" or "overdraft," and some water is always mined before a new equilibrium is achieved after the addition of a stress such as human extraction through wells (Theis 1940). In the extreme, if all water is removed from storage, a groundwater basin could be, for practical purposes, depleted. A challenge for integrated groundwater management is to understand the sources of water where extraction is planned and to appropriately account for the deficiencies caused by extraction.

Second, in the 100 years since Lee's work, the concept of what is an undesired result has evolved significantly. Meinzer (1923), in the decade following Lee's work, in fact did not indicate specific undesired results, but rather defined safe yield as "...the rate at which water can be withdrawn from an aquifer for human use without depleting the supply to such an extent that withdrawal at this rate is no longer economically feasible." At that time, as noted by Reilly and coworkers (Reilly et al. 2008), indoor plumbing was not widespread in the United States and the population was dispersed. It was natural, then, that the feasibility of future *human* consumption would guide concepts of preserving future use. Another widespread attitude of those times was that water flowing to springs or lost to evapotranspiration was "wasted" (Lee 1915). More recently, ecosystem health has been recognized as an important consideration for current and future use, and the dialogue has shifted from a concept of "safe yield" to one of "sustainability" (Alley and Leake 2004). Sustainable development was coined as part of the development that "...meets the needs of the present without compromising the ability of future generations to meet their own needs" (World Commission on Environment and Development 1987). This broad definition is meant to encompass not only the economic needs of future generations but also the health of the ecosystems they depend upon. When viewed in this framework, groundwater use must be balanced not only with the ability of an aquifer to continue supplying water to wells for human consumption but also with the capacity to maintain discharge to surface water, phreatophytic vegetation, and other habitats that make up the ecosystems surrounding and connected to the groundwater system.

It is clear that managing groundwater in a way that does not deplete the source of water or displace water from *all* dependent ecosystems—including humans—is a technical challenge. Some impact is inherent in the disruption of natural equilibrium through human activity—the challenge is to establish an agreed-upon acceptable level of disruption. Pierce et al. (2012) propose a continuum approach that balances socioeconomic, ecosystem, and sustainability constraints. Recently, Werner et al. (2013b) evaluated and ranked occurrences of mega storage depletion worldwide in terms of physical processes and the importance of the resource. Such nuances in definition and approach can pose challenges in coming to agreement among stakeholders (Llamas 2004), but the result of concurring on a definition and approach is much better management of the resource, tailored to the specific environmental and socioeconomic needs of a specific area. Giordano (2009) highlights this complexity noting that groundwater mining in Libya and Saudi Arabia, although unsustainable by most strict definitions, may provide socioeconomic benefit with little or no ecological impact that outweighs the downside of acknowledged depletion that is taking place.

By taking into account water quality, aquifer salinization (Chap. 16), risk of sea water intrusion, and subsidence issues, Konikow and Kendy (2005) described depletion as a physical process that renders reduction in the total or usable volume of the resource. Thus, depletion leads to consequences realized or perceived to be negative for the current and future use of the resource. Consequences of depletions such as salinization can be substantial, because it commonly is very time and resource expensive to bring a degraded aquifer back to its natural state. Further, some impacts—such as subsidence—can be irreversible (Zektser et al. 2005).

Today's nuanced understanding of differences in source from recharge, discharge, and storage is generally well documented and supported in the scientific literature. Yet, the water budget myth persists where science meets policy because it is much simpler to use a single metric—"recharge"—to regulate how much water may be extracted from an aquifer without undesired consequences without regard to the importance of timescale (Harou and Lund 2008). Many of the metrics available to document depletion must pass over these nuances to apply at a large scale and still rely on balancing recharge with human extraction—including metrics referred to in this chapter. Although the concept of sustainability has made its way into the dialogue through acknowledgement that ecological flows should be maintained, the recommended solution still often seems to be regulating pumping rates at less than or equal to recharge rates (ASCE 2004; Beck 2000).

2.3 Groundwater Depletion Globally

Groundwater demands for consumptive and environmental uses are expected to grow, while supplies will remain constrained by unsustainable use of the aquifers. In the last five decades, economic gains from groundwater use have been substantial, but they have been realized at high social and environmental costs (Custodio 2002; Birol et al. 2010). Groundwater levels in many places have already dropped and are further dropping in response to excessive extraction. Adverse effects of overdraft have been observed in many places in the forms of reduced flows in streams and wetlands, stream-aquifer disconnection, water quality degradation through intrusion of saline or poor-quality surface or groundwater, reduced availability of groundwater for consumptive uses, land and aquifer subsidence, and increased costs of pumping. Recent studies have also quantified the contribution of groundwater depletion to sea level rise, accounting for as much as 13 % in recent years (Konikow 2011; Wada et al. 2012)

2.3.1 Global Estimates of Groundwater Extraction

Giordano (2009) reported global groundwater extraction in excess of 650 km^3 per year (Fig. 2.1), with India, the United States, China, Pakistan, Iran, Mexico,

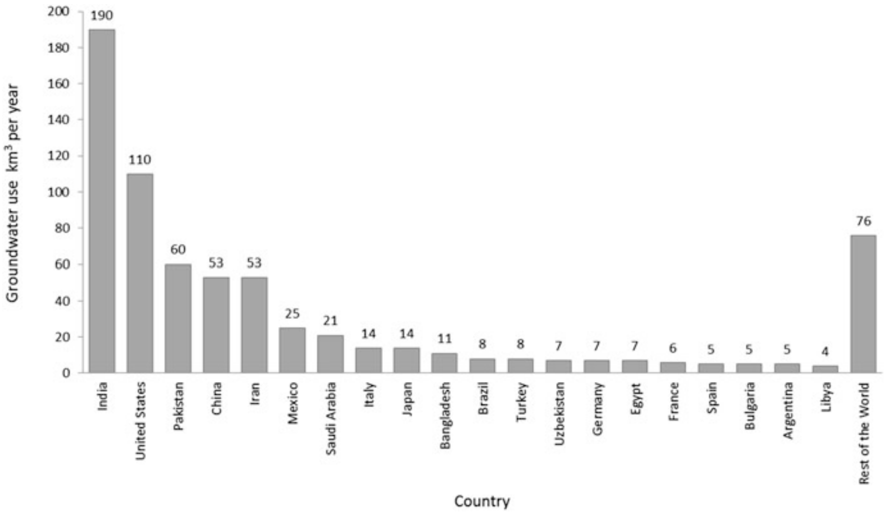

Fig. 2.1 Groundwater use by country, in cubic kilometers per year (Adapted from Giordano (2009))

and Saudi Arabia collectively accounting for 75 % of the global annual water extraction.

The GRACE analysis reports an approximate doubling of global groundwater extraction between 1960 and 2000. From 1960 to 2000, global groundwater annual extraction increased from 312 km^3 in 1960 to 734 km^3 in 2000. Major hot spots of depletion were observed in arid and semiarid parts of the world, mainly resulting from high population density, heavy reliance on groundwater, little and highly variable rainfall that generates quick runoff, and low rates of natural recharge. For subhumid and arid parts of the world Wada et al. (2010), prepared a global map of groundwater depletion by calculating the difference between global groundwater recharge and groundwater extraction for the year 2000. Hot spots of groundwater depletion were reported in the northwest of India, northeast of China, northeast of Pakistan, and in the High Plains and California Central Valley aquifers in the United States. Other countries where depletion was significant included parts of Iran, central Yemen, and southern Spain. The total global groundwater depletion in those areas was reported as 283 ($\pm$40) km^3 per year (Wada et al. 2010). Using an index based approach, Werner et al. (2013b) reported mega storage depletion cases around the world from more than 50 published sources. The largest depletion indices were reported for China, Spain, and the United States.

2.3.2 Global Depletion Examples

Depletion is typically measured by decreases in groundwater levels and decreases in baseflow in surface water bodies that are connected to aquifers. Regions where depletion has been significant as quantified through the best available scientific information include south and central parts of Asia, north China, the Middle East and North Africa, North America, parts of Australia, and many localized areas in southern Europe. In the United States, about 700–800 km^3 of groundwater has been depleted from aquifers in the last 100 years (Konikow and Kendy 2005). In the Fuyang River Basin in the North China Plain, the water surface has dropped from 8 to 50 m during 1967–2000 (Shah et al. 2000). In India, consumptive uses are depleting the groundwater reserves of Rajasthan, Punjab, and Haryana at a rate of or 17.7 ± 4.5 km^3/year. Similarly, a volume of 143.6 km^3 of groundwater was depleted during the period 2003 and 2009 in the in the north-central Middle East, including portions of the Tigris and Euphrates River Basins and western Iran (Voss et al. 2013).

Next we provide an overview of the major depletion examples; the cases discussed are representative and not an exhaustive inventory of the global depletion cases. More details of the global depletion cases can found in Konikow (2011), Morris et al. (2003), Wada et al. (2010) and Werner et al. (2013b).

2.3.2.1 High Plains Aquifer, United States

In the United States, 60 % of irrigation relies on groundwater. The High Plains (HP) aquifer is one of the largest freshwater groundwater systems in the world, covering eight states and encompassing over 450,000 km^2 in area. The HP aquifer is the most intensively used aquifer in the United States, responsible for nearly one-third of the total groundwater extraction, and it provides drinking water to nearly 2.3 million people residing within in the boundaries and vicinity of the aquifer system (Dennehy et al. 2002). Groundwater is considered as the major economic driver for the HP region, known as the “breadbasket of the United States” and annually contributing US $35 billion of the US $300 billion in national total agricultural production in 2007 (Scanlon et al. 2012a).

On the basis of groundwater monitoring data from 1950 to 2007 from 3600 wells, Scanlon et al. (2012a) estimated that 330 km^3 of groundwater was depleted from the HP aquifers. This storage decline in the HP aquifer accounts for nearly 36 % of the total groundwater depleted in the United States during 1900–2008 (Scanlon et al. 2012b). If the depletion were assumed to be uniform throughout the HP aquifer, the corresponding drop in water surface over the entire HP region would be 4 m.

The effects of depletion in terms of water surface decline are highly variable spatially. For example, recent groundwater monitoring from GRACE, (Scanlon et al. 2012d) indicates almost negligible depletion and water surface decline in the northern HP (Nebraska, 0.3 m), concurrent with much greater decline in the water surface in the central HP (Kansas, 7 m) and the southern HP (Texas, 11 m). In localized pockets of the southern HP and large areas of Kansas and Texas, a decline of more than 30 m was observed over 17,000 km^2, where the ratio of the rates of extraction to natural recharge was found to be 10 and greater. Large variation in the depletion is primarily due to a decrease in natural recharge from north to south but partially due to the amount of water pumped from the aquifer. A common view of the HP aquifer is that it contains old water that has been mined and depleted continuously since 1950s. Groundwater age dating indicates that some of the fossil water in the central and south HP aquifer was recharged as long ago as 13,000 years. Policy implemented to control groundwater use in the HP is described in Chap. 21.

2.3.2.2 Northwestern India

India has become the largest consumer of groundwater at the global scale with an estimated total annual consumption of 230 km^3 per year, or about one-fourth of the total global groundwater extraction annually. The annual replenishable groundwater resources of India are estimated as 433 km^3, with net availability of 399 km^3 (Chatterjee and Purohit 2009). India's apparent groundwater surplus can be misleading because of large variation across regions in terms of groundwater availability and extraction, as well as natural recharge. This imbalance of pumping and natural recharge has placed several aquifers in a state of overexploitation and many under semicritical and critical categories (Rodell et al. 2009).

In comparison to the only 20 million ha of land irrigated with surface storage, the irrigated area fed by groundwater now exceeds 45 million ha. Production returns from groundwater irrigation are almost twice those of surface water irrigation because of high reliability and cheaper access. About 70 % of India's agricultural production is generated through use of groundwater (Fishman et al. 2011). The economic value of groundwater irrigation in India in 2002 was estimated at US $8 billion per year (World Bank 2010). Groundwater is a primary source of drinking water supplies for rural villages and a growing number of urban areas. A major portion (85 %) of rural drinking water supply comes from groundwater.

The exploitation of groundwater in many states of India has expanded over the last five decades through installation of millions of irrigation wells (Shah 2009). And the scale of resource exploitation has accelerated in the last two decades. The number of tubewells was less than a million in 1980, jumped to 8 million in the mid-1990s, and exceeded 15 million by 2010 (Shah et al. 2012). In addition to cheaper pumps and low well installation costs, electric power subsidies to farms have played a pivotal role in the phenomenal growth of tubewells and overexploitation of groundwater in 16 major states of India. The flat power tariff reduced the marginal cost of pumping groundwater to near zero (Shah et al. 2012).

Because of the heavy reliance on groundwater for consumptive uses in India, the resource is now approaching its critical limit in some states. The national groundwater assessment in 2004 indicated one-third of India's aquifers fall in the overexploited, semicritical, or critical categories (Rodell et al. 2009). An increasing number of aquifers in northwestern India have reached unsustainable levels of exploitation. In the northern state of Punjab, groundwater in 75 % of the aquifers is overdrawn; in the western Rajasthan state, the corresponding fraction is 60 % (Rodell et al. 2009; World Bank 2010). The potential social and economic consequences of groundwater depletion are serious because aquifer depletion is concentrated in densely populated and economically productive areas. The implications can be serious for achieving food security and sustaining economic growth and environmental quality.

2.3.2.3 Northeastern China

In China, significant shifts toward groundwater dependency have occurred over the last 50 years (see Chap. 19 for a comprehensive overview of Integrated Groundwater Management in China). The installation and use of tubewells across China has increased dramatically, from 150,000 in 1965 in all of China to 4.7 million by the end of 2003 (Wang et al. 2007). In many parts of the country, groundwater levels have been falling at astonishing rates, often more than one to tens of meters per year. Overdraft occurs in more than 164 locations across 24 of China's 31 provinces, affecting more than 180,000 km^2 (Werner et al. 2013b).

Aquifers of the North China Plain (NCP) play a central role in China's food production. The region supplies nearly half of China's wheat and one-third of other cereal grains. The NCP covers 320,000 km^2 and is home to more than 200 million people. In the NCP, groundwater overexploitation for agricultural, industrial, and urban uses began in the early 1970s and became a serious problem after the 1980s with more intensive groundwater extraction. The negative impacts of overexploitation became evident during the 1990s in many parts of the NCP with rapid declines in water levels in both unconfined and confined aquifers. Cones of depression in the potentiometric surface have developed and expanded, with decreases in storage causing subsidence and water quality degradation associated with water surface declines. Groundwater depletion has led to seawater intrusion into the freshwater aquifer system; for example, in the coastal plain of Laizhou city, lateral sea water intrusion into the fresh aquifer system has increased from 50 m per year in 1976 to more than 404.5 m per year in 1988 (Changming et al. 2001). Groundwater depletion has salinized 44 % of the total area between the coastal plain and the city. The Chinese government has implemented a series of water-saving initiatives such as water efficiency in irrigation techniques, water pricing and groundwater licensing, and similar measures. However, the lack of information on volumetric groundwater extraction and limited groundwater monitoring networks make groundwater management challenging.

2.3.2.4 Middle East and North Africa (MENA)

From the standpoint of declining water availability, the Middle East and North Africa (MENA) region is considered by many to be the most water-scarce region of the world. The MENA countries possess annual renewable water resources of 1274 m^3 per capita—the lowest in the world—making the region the most water stressed globally by this metric. MENA is home to about 6 % of the world's population, consisting of 22 countries with 381 million people. And the population is projected to reach nearly 700 million by 2050 (Droogers et al. 2012). Population densities in MENA are largest where irrigation systems are present, including the Nile Delta in Egypt, the central part of Iraq, and Iran (Abu Zeid 2006).

Countries and small territories in the MENA region such as Bahrain, the Gaza Strip, Kuwait, Libya, Oman, Qatar, Saudi Arabia, the United Arab Emirates (UAE), and Yemen have few renewable water resources and heavily rely on groundwater and desalination for most of their supply. The region has some 2800 desalination plants that produce about 10 km^3 of freshwater annually, representing about 38 % of global desalination capacity.

Other countries in MENA such as Egypt, Iraq, Iran, Jordan, Lebanon, the West Bank, Sudan, and Syria get much of their water from river systems but at the same time depend on groundwater for supplemental use. Aquifers in MENA contain both renewable and fossil water. Many countries in the region are depleting groundwater at a rate that exceeds recharge. For example, the ratio of annual groundwater extraction to the estimated recharge exceeds 3.5 in Egypt, is about 8 in Libya, and is 9.54 in Saudi Arabia (Michel et al. 2012). GRACE data (Voss et al. 2013) show that a volume of 143.6 km^3 of groundwater was depleted during the period 2003–2009 in the north-central Middle East, including portions of the Tigris and Euphrates River Basins and western Iran.

In Chap. 24 of this book, the scale of the groundwater-dependent economy in Algeria, Morocco, and Tunisia is discussed. These three countries in North Africa have a high reliance on groundwater for irrigated agriculture, with more than 1.75 million ha of farmland and probably more than 500,000 farm holdings. Algeria's 88 %, Tunisia's 64 %, and Morocco's 42 % of irrigated land rely on groundwater resources. The official figures reported in Chap. 24 indicate that more than half the aquifers in Algeria and Morocco and about one-quarter of the aquifers in Tunisia are overexploited.

2.3.2.5 Australia

Groundwater resources are of great socioeconomic and environmental significance for Australia. The Great Artesian Basin (GAB) is the largest groundwater aquifer system in Australia and underlies 22 % of the Australian continent. The GAB includes considerable areas of the states of Queensland, New South Wales, the Northern Territory, and South Australia. Limited available information on the potential of the GAB resource indicates that nearly 60,000 km^3 of water is contained in the GAB. Groundwater in Australia is pumped mainly from unconfined aquifers, and there is increasing concern regarding the potential impact of groundwater depletion on the sustainability of the resource.

Because of limited and highly variable surface water availability, groundwater use for irrigation has substantially increased in Australia. From the National Land and Water Resources Audit (2001), Khan (2008) reported a 90 % increase in groundwater use across Australia between 1985 and 2000. At present, the volumes of water pumped from aquifers are much greater than natural recharge (Nevill 2009). In many parts of Australia, overdraft from the aquifers is resulting in falling groundwater levels in the shallow unconfined systems and decreasing groundwater pressures in the deep confined and semiconfined systems (MDBA 2012). Many aquifers in the Murray Darling Basin in particular are showing negative socioeconomic and environmental effects as a result of overdraft from aquifers. In many coastal aquifers, saline seawater has intruded to the fresh groundwater aquifers; thus, degradation of groundwater quality is further undermining use of the already scarce resource. A detailed account of saltwater intrusion in Australia and elsewhere is provided in Chap. 16.

2.3.2.6 Techniques for Assessing Groundwater Depletion

Data assimilation of water level fluctuation is the most direct and simplest method to estimate the volume of water depleted from an aquifer. The technique integrates head changes over the aquifer area and multiplies the obtained area by a representative aquifer storage factor to yield an estimate of storage depletion. Major challenges confronted by this simple technique are to establish large-scale monitoring networks and to collect water level data over large areas at regular time intervals. Maintaining a large-scale groundwater data base and keeping the data updated are costly and complex tasks. Community data integration—such as the Incorporated Research Institutions for Seismology (IRIS 2013)—combines centralized data serving with common data standards. Although Aquastat (FAO 2013) is an example of serving water information internationally, it does not include seamless data integration as does IRIS and has limited data on the spatial distribution of groundwater storage and water levels. Particularly in developing countries, advances in data integration will enable managers and researchers to work with more complete information to assess and manage groundwater resources. See Chap. 27 which is devoted to advances in integrated data management.

Even with the great advances in other techniques discussed in the following paragraphs, personal communication with various governmental agencies and ministries remains the most robust and definitive method of assessing groundwater levels and, thus, depletion. Efforts at personal communication can run up against cultural and language barriers—including the desire of some governments to treat water data as strategic and secret (Voss et al. 2013)—and can be very tedious and time consuming. Without organizing community efforts and common data standards, compiling data on the regional scale often requires many late-night phone calls and individual persistence (Fan Y (2013), Personal Communication). Such long-term individual effort can lead to a snapshot in time on conditions at the continental scale (Gleeson et al. 2011) and the global scale (Fan et al. 2013); but without a time series, depletion values cannot be easily obtained. This challenge is less acute for aquifers that fall under a single government's management authority but is exacerbated in transboundary aquifers.

In the United States and Canada, efforts have been made to adopt the Groundwater Markup Language (GWML, (Boisvert and Brodaric 2011)) to unify data among agencies and organizations within both countries. The First Groundwater Interoperability Experiment (Open Geospatial Consortium Inc. 2011) worked toward harmonizing groundwater data across the border between the two nations. In the Second Groundwater Interoperability Experiment (Open Geospatial Consortium Inc. 2013), Australia and Europe are joining the effort. This progress represents steps down a path toward consolidating data and enabling evaluation of conditions on a global scale, but large gaps of information still remain for many areas (Fan et al. 2013).

Even though direct regional groundwater depletion estimates can be integrated to provide global depletion estimates, groundwater data collection and data interpretation are subject to a high level of inconsistencies across countries and regions.

When groundwater data are of questionable quality, information generated through such data tends to be less reliable. This is why the magnitude of depletion is imperfectly assessed and poorly documented at the global scale (Giordano 2009). The water balance approach uses a number of scientific methods to estimate and account for various types of recharge and discharge processes to estimate groundwater storage differences and depletion over specific periods. Numerical simulation models based on water balance calculations have been helpful to estimate net groundwater removed from an aquifer. But the accuracy of the model to predict depletion depends on the quality of hydrogeological data provided as input to the model. Recent advances in the development of three-dimensional hydrogeological models have made it possible to provide better representation of the aquifers, underlying geological formations, and the processes that link the groundwater system both to surface water in general and ecological processes specifically. Examples include HydroGeoSphere (Therrien et al. 2012), GSFLOW (Markstrom et al. 2008), and MIKE SHE (DHI Software 2012). Three-dimensional modeling enables more detailed estimates of depletion and impact on surface water, but it remains limited by the data. At the continental and global scales, models of recharge processes and groundwater flow are typically data-driven, with relatively simple treatment of the physics integrated over coarse grids (Cao et al. 2013; Fan et al. 2013; Scanlon et al. 2006; Wood et al. 2011).

In practice, direct measurement of groundwater depletion at the global scale is imperfect. The imperfections arise because of insufficient groundwater monitoring data networks and inconsistent data collection and reporting standards. Another challenge arises when the depletion process is viewed from multiple dimensions, leading to different definitions of the depletion process and its estimation. Recently, satellite-based GRACE has been able to more confidently measure the changes in groundwater storage over large regions. GRACE measurements are made by measurement of the Earth's gravity, detected from the distance between two coordinated satellites that are generally separated by about 220 km (Tapley et al. 2004). Small changes in gravity on short timescales are generally a function of changes in water storage (underground, on the surface, and in the atmosphere), so quantification of gravity changes can be converted to estimates of water storage changes (Ramillien et al. 2008). Parsing of water content among groundwater, snow, the atmosphere, and surface water requires some processing that differs for various locations and scales (Scanlon et al. 2012c; Longuevergne et al. 2010). Although not a replacement for direct measurement of groundwater storage, GRACE observations have the potential to extend estimates of storage over time, although only back as far as the 2002 launch of the GRACE satellites. Rates of storage depletion in important groundwater-stressed regions have been made using GRACE, including the High Plains of the United States (Scanlon et al. 2012a), India (Rodell et al. 2009), and the Tigris, Euphrates, western Iran region in MENA (Voss et al. 2013).

2.4 Contamination of Groundwater

Water in nature, on the surface or underground, is never free from impurities and typically contains many dissolved and suspended constituents (salts, other inorganic and organic chemicals, sediments, and microorganisms). Contamination of a water body or an aquifer occurs when the concentration of one or more substances increase to a level such that the resulting water quality undermines the use of resource and, in some instances, becomes a hazard to the environment and a risk to human, animal, or plant life (Morris et al. 2003). The principal causes of groundwater contamination due to human activity can be classed as agricultural, industrial, and urban (Foster et al. 2002). Human activity can add salts, chemicals, and microorganisms (pathogens) that affect quality of groundwater.

This section provides an overview of major issues and concerns related to contamination of groundwater. See Chaps. 15 and 16 for a more detailed discussed of water quality.

Here, the significance of the widespread groundwater contamination problem is highlighted with relevant examples. Three groundwater contamination examples and their effects are summarily discussed: (i) land and aquifer salinization, (ii) contamination due to chemicals, and (iii) contamination due to microorganisms.

2.4.1 Land and Aquifer Salinization

Salinization of land and water is a widespread phenomenon that is an issue in more than 100 countries, including China, India, and the United States. Current global estimates indicate that over 1 billion ha are affected by various degrees of soil salinization (Shahid 2013). Globally 45 million ha (18 %) of the total 230 million ha of irrigated land are negatively affected by irrigation-related salinity (Ghassemi et al. 1995), which can result from a high water table, poor drainage conditions, and use of saline-brackish water for irrigation with insufficient drainage.

The Indus Basin of Pakistan is an example of a region severely affected by land and aquifer salinization problems that resulted from continuous irrigation without sufficient drainage. It is estimated that out of the total 16.3 million ha of irrigated land in Pakistan, about 6.2 million ha (38 %) have become waterlogged, with water table levels of <1.5 m below the surface; additionally, 2.3 million ha (14 %) have become saline, with soil ECe (soil saturated extract) >4 dS/m (Kahlown and Azam 2002).

Detail beyond the following overview of land and aquifer salinization process is given in Chap. 16.

2.4.1.1 Land Salinization

Salinization is a characteristic of soil and water which relates to their water-soluble salt content. Such salts predominantly include sodium chloride, but sulfates, carbonates, and magnesium may also be present. A saline soil is one which contains sufficient soluble salts to adversely affect plant growth and crop production. Waterlogging and salinity have been persistent problems in many irrigation regions of the world. Irrigation water normally contains salts in the range of 300–500 mg/l (IWMI 2007). A simple calculation shows that, in the absence of effective leaching, an annual irrigation of 1000 mm with good quality irrigation water and with salt content as low as 300 mg/l adds 300 kg of salts per hectare of irrigated land in a single year. Rainwater, which is considered a source of pure water, can also become a source of salt addition to aquifers and land. Raindrops, during their brief residence in the atmosphere, dissolve carbon dioxide to form a weak carbonic acid. During infiltration, the weak carbonic acid reacts with minerals and rocks in the soil to dissolve them more readily to become a source of salt in aquifers (Hillel 2000). Changes in properties of soil and water lead to the development of an environment which deteriorates soil and water quality.

Waterlogging, another major problem in irrigated land, is the saturation of soil particles with water that results from the rising of the water table due to over-irrigation, seepage, or inadequate drainage. Salinization, however, is a process that increases the concentration of salts in water or soil beyond a threshold limit; that is, mean electircal conductivity in the root zone (EC_e) in excess of 4 deci-siemens per meter (dS/m) at 25 °C (Hillel 2000). The processes of waterlogging and salinization, although different in their characteristics, usually occur together and adversely affect water quality and crop yield.

2.4.1.2 Aquifer Salinization

Mixing of saline water with freshwater is a frequent cause of aquifer salinization in many coastal regions (Werner et al. 2013a). Coastal aquifers are more vulnerable to groundwater extraction because of high population densities and predicted sea-level rise (Ferguson and Gleeson 2012). Coastal areas are the most densely populated areas in the world, with 8 of the 10 largest cities of the world located at coastlines. Nearly half of the world’s population resides in coastal areas (Post 2005), and coastal aquifers provide a water source for more than one billion people.

In most cases, coastal aquifers are hydraulically connected to seawater. Under natural conditions, the hydraulic gradient (in part, a function of the density variation of the seawater and freshwater systems) maintains net water flow from the freshwater aquifer toward the sea. However, the gradient is usually small, and any excessive groundwater pumping can alter the hydraulic balance and allow seawater to enter and replace the freshwater pumped out from the aquifer (Werner et al. 2013a). The quality of groundwater aquifers can also be adversely affected by pumping if interlink connections exist between brackish or saline water. Additionally, a low rate of natural groundwater recharge in combination with sea-level rise can introduce and accelerate movement of saltwater into freshwater aquifers, although Ferguson and Gleeson (2012) found that the impact of groundwater extraction on coastal aquifers was more significant than the impact of sea-level rise or changes in groundwater recharge.

The overall impact of saline water intrusion highly depends on the amount of extraction and natural groundwater recharge. Incorrect positioning of well fields can accelerate the problem. Climate change is expected to exacerbate many water resource problems, but the impact of seawater intrusion may be much more serious and widespread because many areas with moderate population densities and water demand are expected to experience saltwater intrusion.

Seawater intrusion has affected groundwater quality in major coastal irrigation regions around the globe where pumping has destabilized the hydraulic equilibrium of the aquifers. Coastal regions such as Queensland in Australia, Florida in the United States, the southern Atlantic coastline of Spain, and Lebanon are among the most highly visible and notable cases where saltwater has intruded into coastal aquifers. Other problem areas in the United States include Cape May County in New Jersey and in Monterey and Orange Counties in California (Barlow and Reichard 2010). Similarly, in the western State of Sonora in Mexico, seawater has intruded approximately 20–25 km inland, forcing the closure of irrigation wells. Likewise in Cyprus, Egypt and Israel, exploitation of groundwater resources for irrigation has lowered aquifers' hydraulic heads to allow seawater intrusion.

In the Burdekin coastal region of Queensland, Australia, more than 1800 wells are currently used for irrigation. The large volumes of groundwater extracted have at times lowered the regional water surfaces and made it challenging to control seawater intrusion (Narayan et al. 2007). To confront long droughts, future use of groundwater is likely to increase in Australia. This growing use of groundwater will stress the aquifers already in deficit. Thus, saltwater intrusion will likely become more challenging because of the extensive coastlines where the majority of the population resides.

2.4.2 Groundwater Contamination Due to Chemicals

Fertilizers, pesticides, and salts contained in irrigation water can be major agricultural contaminants. Excessive irrigation drives water from the root zone of crops to the groundwater below (Chowdary et al. 2005), carrying with them applied fertilizers and pesticides and their component nitrogen compounds, phosphorus, potassium and other minerals and chemical compounds (Langwaldt and Puhakka 2000). Because of the widespread areal extent of these contaminants, they are often referred to as "nonpoint-source" contaminants.

Industrial wastes contain a wide variety of heavy metals and solvents. A recent study by Dwivedi and Vankar (2014) reported contamination of groundwater potentially from industrial sources (tanning, textile, and several others) in the Kanpur-Unnao district of India. Concentrations of cadmium, cobalt, chromium, copper, mercury, nickel, lead, tin, and zinc were found to exceed the maximum permissible limit. When chemical releases occur at specific facilities, they are referred to as "point-source" contaminants.

The accidental spillage and leakage of industrial chemicals can also cause serious groundwater contamination (Foster and Chilton 2003a). Subsurface releases of MTBE (methyl tertiary-butyl ether) can be a source of groundwater contamination. MTBE is a gasoline fuel additive that can leak from gasoline underground storage tanks and contaminate aquifers and wells. In the United States alone, releases of gasoline fuels has been reported at more than 250,000 sites, putting over 9000 municipal water supply wells at risk of contamination with MTBE (Einarson and Mackay 2001). Synthetic microorganic compounds also known as emerging organic contaminants (EOCs) are another and new source of groundwater contamination reported across Europe and many other parts of the world (Lapworth et al. 2012). EOCs are used for a range of industrial purposes including food preservation, pharmaceuticals, and healthcare products (Lapworth et al. 2012). Public health and environmental impacts of EOCs in groundwater are currently under-researched areas.

Arsenic and nitrate are two major contaminants with serious public health impacts. High concentrations of arsenic in groundwater have been recognized as a major public health concern in several countries and often are the result of natural conditions rather than human activity. The contamination of groundwater by arsenic in Bangladesh has been called the largest poisoning of a human population in history (Smith et al. 2000). An estimated 36 million people in the Bengal Delta alone (Bangladesh and India) are at risk of drinking arsenic-contaminated water (Nordstrom 2002). Long term exposure of arsenic in drinking water and its impacts on human health are documented in Ng et al. (2003). Geochemical processes in the presence of oxygen dissolve arsenopyrite [FeAsS], leading to increased concentrations of dissolved arsenic in groundwater. Oxidation can be a major driver to mobilize arsenic already present in aquifer rocks and can be promoted as a result of recharge by oxygenated waters or through lowering of the groundwater surface by excessive pumping (Nordstrom 2002). Chemical reactions among nitrate, iron, and oxygen can also increase mobilization of arsenic in aquifers (Höhn et al. 2006). The incidence of high concentrations of arsenic in drinking water is significant in Asian countries. The problem was initially detected in Bangladesh, India, and China. Most recently, the problem has been reported in Myanmar, Cambodia, parts of Europe, the United States, and Australia. A global summary of arsenic contamination of groundwater is available in Ravenscroft et al. (2011) and Mukherjee et al. (2006).

Nitrate contamination of groundwater is a widespread and global problem both in developed and developing nations. Excessive application of commercial fertilizers or animal waste and inadequate waste disposal of municipal and animal waste are associated with this problem. High concentration of nitrate in municipal groundwater (10–50 mg/l) is considered a public health hazard. Nitrate contamination of groundwater due to agrochemicals has become a serious problem in China and India (Foster and Chilton 2003b). A detailed review of nitrate contamination of groundwater and its health impact is available in Spalding and Exner (1993) and Canter (1996).

2.4.3 Groundwater Contamination Due to Microorganisms

Microbial contamination of groundwater can be caused by inadequate protection of aquifers against release of sewage effluent into groundwater. Contamination of groundwater can occur via many pathways, such as from urban landfills in proximity to natural groundwater recharge sites, rural on-site sanitation facilities, leaking septic tanks and sewers, and waste from farm animals. The concentration of many harmful microorganisms attenuates (naturally reduces) when water passes through the unsaturated zone; however, the degree of pathogen removal depends on the type of soil, level of contamination, and type of contaminant. Natural attenuation generally is most effective in the unsaturated zone, especially in the top soil layers where biological activity is greatest (Morris et al. 2003).

Several viral and bacterial pathogens present in human and animal waste contaminate groundwater and cause human health problems. In 2012, more than 500,000 diarrhea deaths were estimated to be caused by microbially contaminated drinking water (Prüss-Ustün et al. 2014). Baldursson and Karanis (2011) give a comprehensive review of worldwide waterborne disease outbreaks that occurred and were documented between 2004 and 2010. Similarly, a recent study based on a systematic review by Ngure et al. (2014) provides a global assessment of drinking-water microbial contamination. All incidence of waterborne diseases cannot be attributed to groundwater, because microbial contamination of water can occur in surface water bodies and in distribution pipes. However, a significant fraction of waterborne disease outbreaks may be associated with groundwater, given that more than 50 % of population worldwide meet their primary drinking needs from groundwater that may be contaminated at some stage (Macler and Merkle 2000).

2.5 The Water-Energy Nexus

Water and energy are inextricably linked in many important ways and this issue is covered in more detail in Chap. 4. Water is used in the generation of energy, and energy is required for the movement and treatment of water. This linkage results in multiple management challenges.

The movement of water requires a significant portion of all energy generated worldwide. In California (United States), 19 % of all electrical energy produced is used for water-related conveyance and treatment (Navigant Consulting Inc. 2006)—nearly 2 % of all electrical energy in California is used for groundwater extraction through pumping (GEI Consultants/Navigant Consulting Inc 2010). Such energy requirements account also for significant contributions to greenhouse gas emissions, estimated as 0.6 % of China's emissions (Wang et al. 2012) and 4–6 % of India's emissions (Shah et al. 2012). These energy requirements increase with the distance the water must be lifted (depth to water) and decrease with pump efficiency. Hence, declining water levels will increase energy requirements for groundwater pumping unless offset by increased pump efficiency. This increased energy demand for pumping is exacerbated in India by government subsidies for electrical power for the purpose of groundwater extraction (Badiani et al. 2012)

In addition to energy use for water movement and treatment, groundwater plays an important role in the generation of energy—particularly the production of alternative energy such as biofuels (Gerbens-Leenes and Hoekstra 2012; Dominguez-Faus et al. 2009). Significant water is used both in the growing of feedstock to create ethanol and in the distillation of the feedstock into fuel. In the United States, governmental mandates require that ethanol from corn (maize) will continue into the future (Dominguez-Faus et al. 2009), although a wide range of water footprint calculations suggest that efficiencies may be found that could reduce groundwater extraction needs for irrigation and distillation (Gerbens-Leenes and Hoekstra 2012; Dominguez-Faus et al. 2009). Other alternative energy technologies can have surprising energy implications. Concentrated solar power generation on a large scale in desert environments can require large amounts of water for cooling and washing (Woody 2009; McKinnon 2010). In the United States, the National Research Council (2012) has also studied production of biofuel from algae, raising questions about sustainability.

In recent years, unconventional drilling for shale gas and coal bed methane—particularly in the United States, China, and Australia—has increased dramatically (Vidic et al. 2013; Moore 2012). Improvements in the accuracy of horizontal well drilling, coupled with hydraulic fracturing, have made it practical to extract methane from thin, deep and tight strata. These advantages, coupled with increasing energy demand, have resulted in massive expansion of exploitation of these unconventional gas reserves. Hydraulic fracturing uses a focused large amount of water for short periods of time, resulting in competition with other water users—particularly in arid regions like the Eagle Ford Formation in Texas (United States). Hydraulic fracturing also uses a variety of chemical additives in the process. Some water contaminated with these additives returns as flowback water and must be disposed of, leading to a potential groundwater contamination source (Vidic et al. 2013). One concern is that methane liberated by the hydraulic fracturing process and additive chemicals could migrate to shallow aquifers or the surface. A recent study (Myers 2012) attempted to address this issue and prompted discussion and criticism (Saiers and Barth 2012; Myers 2013; Cohen et al. 2013), highlighting the level of uncertainty about the degree and nature of potential contamination from this activity. Further research in the field and through modeling is necessary for understanding of the depth and breadth of potential groundwater impacts to catch up with the rapid increase in development of unconventional gas resources (Jackson et al. 2013).

2.6 Transboundary Water Conflict

Most of the literature discussing transboundary water conflict has focused on surface water. Groundwater conflict has received less attention. However, owing to both “uncertainty in defining ground water flow. . .[and]. . .uncertainty of the hydraulic connection between groundwater and surface water" (Jarvis et al. 2005) and combined with increasing water usage needs—particularly for agricultural irrigation (Llamas and Martinez-Santos 2005)—it seems that serious conflict over transboundary groundwater resources may be inevitable. This condition is exacerbated by a lack of regulation and management of groundwater, which is often blamed on the same uncertainties surrounding the quantity and dynamics of groundwater at the regional scale (Llamas and Martinez-Santos 2005; Jarvis et al. 2005; Puri 2003). Several conceptual models can apply to transboundary aquifers, including cases where the source of water to the aquifer is in one country but the main demand is in another (for example, Eckstein and Eckstein 2005). Transboundary aquifers meeting these definitions number as many as 408 (UN-IGRAC 2012). Using analysis similar to the groundwater footprint (Gleeson et al. 2012), Wada and Heinrich (2013) performed a quantitative assessment of water stress (considering recharge, extraction, and environmental flows) for the 408 identified transboundary aquifers and determined that 8 % of them are stressed by human consumption. They point out, however, that many of these transboundary aquifers are found in geopolitically charged areas such as the Arabian Peninsula, the United States—Mexico border, and India and Pakistan.

In one example of this type, the Ceylanpinar aquifer spans the border between Turkey and Syria, with recharge in the Turkish headwaters and the majority of discharge in the Ras al-Ain Springs in Syria (Oeztan and Axelrod 2011). Data availability is asymmetric, with much more information available about conditions in the aquifer in Turkey than in Syria. Nonetheless, Oeztan and Axelrod (2011) modeled the aquifer to try to calculate sustainable extraction rates based on discharge from the springs. Mutually beneficial organic agriculture along the border that previously was unfarmable due to extensive placement of landmines is proposed but would first depend on cooperation with respect to hydrogeologic and water use information. Joint management to prevent overdepletion requires collaboration, which may be at odds with other priorities of neighboring countries, but this example shows it can have positive outcomes.

Beyond water quantity, water quality concerns can arise when contaminants enter an aquifer under a different governance than that of the users of the aquifer; for example, such as bordering northeastern Greece (Vryzas et al. 2012) and Russia (Zektser 2012). Similar challenges as facing depletion problems are encountered in managing water quality. The parallel challenges of establishing responsibility for contamination and finding the motivation to remediate it can present opportunities for constructive collaboration but also may heighten tension in some areas.

In modern times (1948–present), no full-scale declarations or acts of war have been attributed to the tension related to the use of transboundary water (De Stefano et al. 2010). This is contrary to predictions stemming from at least the 1980s onward that major wars—particularly in the Middle East—would be fought over water because of stress over increasing demand for water resources due to increasing population, climate change, and depletion of water sources (see Cooley (1984) and Starr (1991), for example). It is still possible for this to happen, and indeed tensions and local violence have been attributed to water conflict, but thus far full-scale war has not resulted with the exception of the war between Sumerian city-states Lagash and Umma in 2500 BCE (Wolf 1998). Although a somewhat controversial notion, it has been argued that interactions among states involving water more often, of necessity, lead to cooperation than conflict (De Stefano et al. 2010; Wolf 2007).

In summary, transboundary aquifers present many challenges in integrated management. The connection between surface water and groundwater are all the more important because the source of water and the water's users (human or ecological) may be in different countries. Data sharing and integration are more challenging across national borders but are extremely important to reduce the uncertainties surrounding integrated management. An additional challenge is that protection of water resources in one country may depend on the actions taken in another country. This binding together for a common purpose provides the opportunity for cooperation but may also devolve into conflict. For these reasons, active management and communication are key to managing water resources across boundaries.

2.7 Conclusion

The issues outlined in this chapter highlight both the challenges and promise of the groundwater issue internationally. The growing importance of groundwater supply combined with the challenges in its characterization and measurement make management difficult. Yet, advances in data analysis, remote sensing, and modeling at regional to continental scales provide some hope for more informed planning, which may ultimately lead to sustainable and responsible management.

Depletion of groundwater—a precious resource for agriculture, ecosystem services, and domestic supply—has the potential to cause significant interruption of societal and ecological functions. The uncertainties inherent in managing a resource that is generally unseen create challenges in management and can lead to conflict among interests vying for the resource—because proving who is responsible for stresses and impacts is a challenge.

Advances in remote sensing (such as the GRACE satellite), data management, and numerical modeling provide hope of reducing the uncertainty of evaluating the magnitude and locations of depletion and degradation of groundwater resources. None of the technical and managerial issues raised in this chapter can be properly considered on its own. The water budget myth implied a simple balance between recharge and availability, but over the past century we have learned that the interconnections among groundwater-dependent ecosystems, human needs, and the groundwater system are deep and elaborate. Only an integrated approach to water management—viewing the components of the system together with competing needs—can maintain sustainability for future generations and a robust environment. Integration is also critical to manage the connections between seemingly disparate sectors of society and economics. As mentioned previously, the connection between electrical prices and agricultural pumping is an important consideration in India. The desire to mitigate climate change (see Chap. 5 for this issue) through alternative energy production can have a ripple effect of consequences on water resources, particularly in the case of biofuels. Agricultural policy beyond water use restrictions has important implications on water quality as it relates to chemical use and to salinization of soil and water. Even the stability of relationships among nations can hinge on proper water management.

References

Abu Zeid M (2006) The middle east water report. The 4th World Water Forum March 17–22, 2006, Mexico City, Mexico

Alley WM, Leake SA (2004) The journey from safe yield to sustainability. Ground Water 42(1): 12–16

ASCE (2004) Regulated riparian model water code. ASCE Standard, vol ASCE/EWRI 40–03. ASCE

Badiani R, Jessoe KK, Plant S (2012) Development and the environment: the implications of agricultural electricity subsidies in India. J Environ Dev 21(2):244–262. doi:10.1177/1070496512442507

Baldursson S, Karanis P (2011) Waterborne transmission of protozoan parasites: review of worldwide outbreaks—an update 2004–2010. Water Res 45(20):6603–6614

Barlow PM, Reichard EG (2010) Saltwater intrusion in coastal regions of North America. Hydrogeol J 18(1):247–260

Beck RE (2000) The regulated riparian model water code: blueprint for twenty first century water management. William & Mary Environ Law Policy Rev 25(1):113–167

Birol E, Koundouri P, Kountouris Y (2010) Assessing the economic viability of alternative water resources in water-scarce regions: combining economic valuation, cost-benefit analysis and discounting. Ecol Econ 69(4):839–847

Boisvert É, Brodaric B (2011) GroundWater markup language specification v. 1.0. http://ngwd-bdnes.cits.nrcan.gc.ca/service/api_ngwds:def/en/gwml.html. Accessed 12 July 2013

Bredehoeft J (1997) Safe yield and the water budget myth. Ground Water 35(6):929

Bredehoeft JD, Papadopulos SS, Cooper HH Jr (1982) Groundwater–the water-budget myth. In: Scientific basis of water-resource management. National Academy Press, Washington, DC, pp 51–57

Canter LW (1996) Nitrates in groundwater. CRC press, Boca Raton, Florida, USA

Cao G, Zheng C, Scanlon BR, Liu J, Li W (2013) Use of flow modeling to assess sustainability of groundwater resources in the North China plain. Water Resour Res 49(1):159–175. doi:10.1029/2012WR011899

Changming L, Jingjie Y, Kendy E (2001) Groundwater exploitation and its impact on the environment in the North China plain. Water Int 26(2):265–272

Chatterjee R, Purohit RR (2009) Estimation of replenishable groundwater resources of India and their status of utilization. Curr Sci 96(12):1581–1591

Chowdary VM, Rao NH, Sarma PBS (2005) Decision support framework for assessment of non-point-source pollution of groundwater in large irrigation projects. Agric Water Manag 75(3): 194–225. doi:10.1016/j.agwat.2004.12.013

Cohen HA, Parratt T, Andrews CB (2013) Potential contaminant pathways from hydraulically fractured shale to aquifers. Ground Water 51(3):317–319. doi:10.1111/gwat.12015

Cooley JK (1984) The war over water. Foreign Policy 54:3–26. doi:10.2307/1148352

Custodio E (2002) Aquifer overexploitation: what does it mean? Hydrogeol J 10(2):254–277

De Stefano L, Edwards P, de Silva L, Wolf AT (2010) Tracking cooperation and conflict in international basins: historic and recent trends. Water Policy 12(6):871–884. doi:10.2166/wp.2010.137

Dennehy KF, Litke DW, McMahon PB (2002) The high plains aquifer, USA: groundwater development and sustainability. Geol Soc Lond Spec Publ 193(1):99–119. doi:10.1144/gsl.sp.2002.193.01.09

DHI Software (2012) MIKE SHE. http://www.dhisoftware.com/Products/WaterResources/MIKESHE.aspx. Accessed 12 July 2013

Dominguez-Faus R, Powers SE, Burken JG, Alvarez PJ (2009) The water footprint of biofuels: a drink or drive issue? Environ Sci Technol 43(9):3005–3010. doi:10.1021/es802162x

Droogers P, Immerzeel WW, Terink W, Hoogeveen J, Bierkens MFP, van Beek LPH, Debele B (2012) Water resources trends in Middle East and North Africa towards 2050. Hydrol Earth Syst Sci 16(9):3101–3114

Dwivedi AK, Vankar PS (2014) Source identification study of heavy metal contamination in the industrial hub of Unnao, India. Environ Monit Assess 186(6):3531–3539

Eckstein Y, Eckstein GE (2005) Transboundary aquifers: conceptual models for development of international law. Ground Water 43(5):679–690. doi:10.1111/j.1745-6584.2005.00098.x

Einarson MD, Mackay DM (2001) Peer reviewed: predicting impacts of groundwater contamination. Environ Sci Technol 35(3):66A–73A

Fan Y, Li H, Miguez-Macho G (2013) Global patterns of groundwater table depth. Science 339(6122):940–943. doi:10.1126/science.1229881

FAO (2013) Aquastat. FAO—UN. http://www.fao.org/nr/water/aquastat/main/index.stm. Accessed 15 July 2013

Ferguson G, Gleeson T (2012) Vulnerability of coastal aquifers to groundwater use and climate change. Nat Clim Chang 2(5):342–345. http://www.nature.com/nclimate/journal/v2/n5/abs/nclimate1413.html#supplementary-information

Fishman RM, Siegfried T, Raj P, Modi V, Lall U (2011) Over-extraction from shallow bedrock versus deep alluvial aquifers: reliability versus sustainability considerations for India's groundwater irrigation. Water Resour Res 47(12):W00L05. doi:10.1029/2011WR010617

Foster S, Chilton P (2003a) Groundwater: the processes and global significance of aquifer degradation. Philos Trans R Soc Lond B Biol Sci 358(1440):1957–1972

Foster SS, Chilton PJ (2003b) Groundwater: the processes and global significance of aquifer degradation. Philos Trans R Soc Lond B Biol Sci 358(1440):1957–1972. doi:10.1098/rstb.2003.1380

Foster S, Hirata R, Gomes D, D'Elia M, Paris M (2002) Groundwater quality protection: a guide for water utilities, municipal authorities, and environment agencies. World Bank, Washington, DC

GEI Consultants/Navigant Consulting Inc. (2010) Embedded energy in water studies study 1: statewide and regional water-energy relationship. California Public Utilities Commission Energy Division. San Francisco, California, USA

Gerbens-Leenes W, Hoekstra AY (2012) The water footprint of sweeteners and bio-ethanol. Environ Int 40:202–211. doi:10.1016/j.envint.2011.06.006

Ghassemi F, Jakeman AJ, Nix HA (1995) Salinisation of land and water resources: human causes, extent, management and case studies. CAB International, Wallington, Oxon 526 p.

Giordano M (2009) Global groundwater? Issues and solutions. Annu Rev Environ Resour 34: 153–178. doi:10.1146/annurev.environ.030308.100251

Gleeson T, Marklund L, Smith L, Manning AH (2011) Classifying the water table at regional to continental scales. Geophys Res Lett 38(5), L05401. doi:10.1029/2010GL046427

Gleeson T, Wada Y, Bierkens MF, van Beek LP (2012) Water balance of global aquifers revealed by groundwater footprint. Nature 488(7410):197–200. doi:10.1038/nature11295

Harou JJ, Lund JR (2008) Ending groundwater overdraft in hydrologic-economic systems. Hydrogeol J 16(6):1039–1055

Hillel D (2000) Salinity management for sustainable irrigation: integrating science, environment, and economics. World Bank Publications, Washington, DC, USA

Hockey TA, Trimble V, Bracher K (2007) The biographical encyclopedia of astronomers. Springer reference. Springer, New York

Höhn R, Isenbeck-Schröter M, Kent DB, Davis JA, Jakobsen R, Jann S, Niedan V, Scholz C, Stadler S, Tretner A (2006) Tracer test with As(V) under variable redox conditions controlling arsenic transport in the presence of elevated ferrous iron concentrations. J Contam Hydrol 88(1–2):36–54. doi:10.1016/j.jconhyd.2006.06.001

IRIS (2013) Incorporated Research Institutions for Seismology. http://www.iris.edu/hq/. Accessed 15 July 2013

IWMI (2007) International Water Management Institute (IWMI). Technical reports on Salinity Management in Pakistan: multiple issues, IWMI, Colombo

Jackson RE, Gorody AW, Mayer B, Roy JW, Ryan MC, Van Stempvoort DR (2013) Groundwater protection and unconventional gas extraction: the critical need for field-based hydrogeological research. Ground Water 51(4):488–510. doi:10.1111/gwat.12074

Jarvis T, Giordano M, Puri S, Matsumoto K, Wolf A (2005) International borders, ground water flow, and hydroschizophrenia. Ground Water 43(5):764–770. doi:10.1111/j.1745-6584.2005.00069.x

Kahlown MA, Azam M (2002) Individual and combined effect of waterlogging and salinity on crop yields in the Indus basin. Irrig Drain 51(4):329–338

Khan S (2008) Managing climate risks in Australia: options for water policy and irrigation management. Aust J Exp Agric 48(3):265–273

Konikow LF (2011) Contribution of global groundwater depletion since 1900 to sea-level rise. Geophys Res Lett 38(17), L17401. doi:10.1029/2011gl048604

Konikow LF, Kendy E (2005) Groundwater depletion: a global problem. Hydrogeol J 13(1): 317–320

Langwaldt JH, Puhakka JA (2000) On-site biological remediation of contaminated groundwater: a review. Environ Pollut 107(2):187–197. doi:10.1016/S0269-7491(99)00137-2

Lapworth D, Baran N, Stuart M, Ward R (2012) Emerging organic contaminants in groundwater: a review of sources, fate and occurrence. Environ Pollut 163:287–303

Lee CH (1915) The determination of safe yield of underground reservoirs of the closed-basin type. Trans Am Soc Civ Eng 78:148–151

Llamas R (2004) Use of groundwater. Series on water and ethics. UNESCO, Paris

Llamas MR, Martinez-Santos P (2005) Intensive groundwater use: silent revolution and potential source of social conflicts. J Water Resour Plann Manag-Asce 131(5):337–341. doi:10.1061/(asce)0733-9496(2005)131:5(337)

Longuevergne L, Scanlon BR, Wilson CR (2010) GRACE hydrological estimates for small basins: evaluating processing approaches on the High Plains Aquifer, USA. Water Resour Res 46(11), W11517. doi:10.1029/2009wr008564

Macler BA, Merkle JC (2000) Current knowledge on groundwater microbial pathogens and their control. Hydrogeol J 8(1):29–40

Markstrom SL, Niswonger RG, Regan RS, Prudic DE, Barlow PM (2008) GSFLOW-coupled ground-water and surface-water FLOW model based on the integration of the Precipitation-Runoff Modeling System (PRMS) and the Modular Ground-Water Flow Model (MODFLOW-2005). Techniques and Methods 6-D1. Reston. http://pubs.usgs.gov/tm/tm6d1/pdf/tm6d1.pdf

McKinnon S (2010) Amid state's push for solar power, water-supply worries arise. The Arizona Republic, 17 January

MDBA (2012) Murray-Darling Basin Authority 2012, The proposed Groundwater Baseline and Sustainable Diversion Limits: methods report. MDBA publication no: 16/12. Murray-Darling Basin Authority, Canberra

Meinzer OE (1923) Outline of ground-water hydrology, Water-supply paper 494. Geological survey water-supply papers, vol 949. United States Geological Survey, Washington, DC

Michel D, Pandya A, Hasnain SI, Sticklor R, Panuganti S (2012) Water challenges and cooperative response in the middle east and north Africa. Brookings Institution, New York, Available at http://www.brookings.edu/about/projects/islamic-world

Moore TA (2012) Coalbed methane: a review. Int J Coal Geol 101:36–81. doi:10.1016/j.coal.2012.05.011

Morris BL, Lawrence AR, Chilton P, Adams B, Calow RC, Klinck BA (2003) Groundwater and its susceptibility to degradation: a global assessment of the problem and options for management. UNEP early warning and assessment vol report series RS. 03–3, Nairobi

Mukherjee A, Sengupta MK, Hossain MA, Ahamed S, Das B, Nayak B, Lodh D, Rahman MM, Chakraborti D (2006) Arsenic contamination in groundwater: a global perspective with emphasis on the Asian scenario. J Health Popul Nutr 24(2):142–163

Myers T (2012) Potential contaminant pathways from hydraulically fractured shale to aquifers. Ground Water 50(6):872–882. doi:10.1111/j.1745-6584.2012.00933.x

Myers T (2013) Potential contaminant pathways from hydraulically fractured shale to aquifers author's reply. Ground Water 51(3):319–321. doi:10.1111/gwat.12016

Narayan KA, Schleeberger C, Bristow KL (2007) Modelling seawater intrusion in the Burdekin Delta Irrigation Area, North Queensland, Australia. Agric Water Manag 89(3):217–228. doi:10.1016/j.agwat.2007.01.008

National Land and Water Resources Audit (2001) Australian water resources assessment 2000. Surface water and groundwater availability and quality. A report of the National Land and Water Resources Audit, Goanna Print, Turner

National Research Council (2012) Sustainable development of Algal Biofuels in the United States. The National Academies Press, Washington, DC

Navigant Consulting Inc. (2006) Refining estimates of water—related energy use in California. California Energy Commission, PIER Industrial/Agricultural/Water End Use Energy Efficiency Program. CEC-500-2006-118

Nevill C (2009) Managing cumulative impacts: groundwater reform in the Murray-Darling Basin, Australia. Water Resour Manag 23(13):2605–2631. doi:10.1007/s11269-009-9399-0

Ng JC, Wang J, Shraim A (2003) A global health problem caused by arsenic from natural sources. Chemosphere 52(9):1353–1359

Ngure FM, Reid BM, Humphrey JH, Mbuya MN, Pelto G, Stoltzfus RJ (2014) Water, sanitation, and hygiene (WASH), environmental enteropathy, nutrition, and early child development: making the links. Ann N Y Acad Sci 1308(1):118–128

Nordstrom DK (2002) Worldwide occurrences of arsenic in ground water. Science (Washington) 296(5576):2143–2145

Oeztan M, Axelrod M (2011) Sustainable transboundary groundwater management under shifting political scenarios: the Ceylanpinar Aquifer and Turkey-Syria relations. Water Int 36(5): 671–685. doi:10.1080/02508060.2011.601546

Open Geospatial Consortium Inc. (2011) Groundwater interoperability experiment. http://external.opengis.org/twiki_public/HydrologyDWG/GroundwaterInteroperabilityExperiment. Accessed 12 July 2013

Open Geospatial Consortium Inc. (2013) Groundwater interoperability experiment 2. http://external.opengeospatial.org/twiki_public/HydrologyDWG/GroundwaterInteroperabilityExperiment2. Accessed 12 July 2013

Pierce SA, Sharp JM, Guillaume JHA, Mace RE, Eaton DJ (2012) Aquifer-yield continuum as a guide and typology for science-based groundwater management. Hydrogeol J 21(2):331–340. doi:10.1007/s10040-012-0910-y

Post V (2005) Fresh and saline groundwater interaction in coastal aquifers: is our technology ready for the problems ahead? Hydrogeol J 13(1):120–123

Prüss-Ustün A, Bartram J, Clasen T, Colford JM, Cumming O, Curtis V, Bonjour S, Dangour AD, De France J, Fewtrell L, Freeman MC, Gordon B, Hunter PR, Johnston RB, Mathers C, Mäusezahl D, Medlicott K, Neira M, Stocks M, Wolf J, Cairncross S (2014) Burden of disease from inadequate water, sanitation and hygiene in low- and middle-income settings: a retrospective analysis of data from 145 countries. Trop Med Int Health 19(8):894–905. doi:10.1111/tmi.12329

Puri S (2003) Transboundary aquifer resources—international water law and hydrogeological uncertainty. Water Int 28(2):276–279

Ramillien G, Famiglietti JS, Wahr J (2008) Detection of continental hydrology and glaciology signals from GRACE: a review. Surv Geophys 29(4–5):361–374. doi:10.1007/s10712-008-9048-9

Ravenscroft P, Brammer H, Richards K (2011) Arsenic pollution: a global synthesis, vol 94. John Wiley and Sons, Hoboken, NJ, USA

Reilly TE, Dennehy KF, Alley WM, Cunningham WL (2008) Ground-water availability in the United States, vol 1323, U.S. Geological Survey circular. USGS, Reston

Rodell M, Velicogna I, Famiglietti JS (2009) Satellite-based estimates of groundwater depletion in India. Nature 460(7258):999–1002. doi:10.1038/nature08238

Saiers JE, Barth E (2012) Potential contaminant pathways from hydraulically fractured shale aquifers. Ground Water 50(6):826–828. doi:10.1111/j.1745-6584.2012.00990.x

Scanlon BR, Keese KE, Flint AL, Flint LE, Gaye CB, Edmunds WM, Simmers I (2006) Global synthesis of groundwater recharge in semiarid and arid regions. Hydrol Process 20(15): 3335–3370. doi:10.1002/hyp.6335

Scanlon BR, Faunt CC, Longuevergne L, Reedy RC, Alley WM, McGuire VL, McMahon PB (2012a) Groundwater depletion and sustainability of irrigation in the US High Plains and Central Valley. Proc Natl Acad Sci U S A 109(24):9320–9325

Scanlon BR, Faunt CC, Longuevergne L, Reedy RC, Alley WM, McGuire VL, McMahon PB (2012b) Groundwater depletion and sustainability of irrigation in the US High Plains and Central Valley. Proc Natl Acad Sci U S A 109(24):9320–9325. doi:10.1073/pnas.1200311109

Scanlon BR, Longuevergne L, Long D (2012c) Ground referencing GRACE satellite estimates of groundwater storage changes in the California Central Valley, USA. Water Resour Res 48(4): n/a–n/a. doi:10.1029/2011wr011312

Scanlon BR, Longuevergne L, Long D (2012d) Ground referencing GRACE satellite estimates of groundwater storage changes in the California Central Valley, USA. Water Resour Res 48(4), W04520. doi:10.1029/2011WR011312

Shah T (2009) Climate change and groundwater: India's opportunities for mitigation and adaptation. Environ Res Lett 4(3):035005. doi:10.1088/1748-9326/4/3/035005

Shah T, Molden D, Sakthivadivel R, Seckler D (2000) The global groundwater situation: overview of opportunities and challenges. International Water Management Institute, Colombo

Shah T, Giordano M, Mukherji A (2012) Political economy of the energy-groundwater nexus in India: exploring issues and assessing policy options. Hydrogeol J 20(5):995–1006

Shahid S (2013) Developments in soil salinity assessment, modeling, mapping, and monitoring from regional to submicroscopic scales. In: Abdelfattah MA, Taha FK, Shahid SA (eds) Developments in soil salinity assessment and reclamation. Springer, Dordrecht, The Netherlands, pp 3–43. doi:10.1007/978-94-007-5684-7_1

Smith AH, Lingas EO, Rahman M (2000) Contamination of drinking-water by arsenic in Bangladesh: a public health emergency. Bull World Health Organ 78(9):1093–1103

Sophocleous M (1997) Managing water resources systems: why "safe yield" is not sustainable. Ground Water 35(4):561

Spalding RF, Exner ME (1993) Occurrence of nitrate in groundwater—a review. J Environ Qual 22(3):392–402

Starr JR (1991) Water wars. Foreign Policy 82:17–36. doi:10.2307/1148639

Tapley BD, Bettadpur S, Watkins M, Reigber C (2004) The gravity recovery and climate experiment: mission overview and early results. Geophys Res Lett 31(9), L09607. doi:10.1029/2004GL019920

Theis CV (1940) The source of water derived from wells. Civ Eng 10(5):277–280

Therrien R, McLaren RG, Sudicky EA, Park Y-J (2012) HydroGeoSphere: a three-dimensional numerical model describing fully-integrated subsurface and surface flow and solute transport. Groundwater Simulations Group. Waterloo, Ontario, Canada

Todd DK (1959) Ground water hydrology. Wiley, New York

UN-IGRAC (2012) Transboundary Aquifers of the World 2012. United Nations International Groundwater Resources Assessment Centre. http://www.un-igrac.org/sites/default/files/resources/files/FINAL_MAP_CMYK_withLogos.jpg. Accessed 24 Jun 2014

Vidic RD, Brantley SL, Vandenbossche JM, Yoxtheimer D, Abad JD (2013) Impact of shale gas development on regional water quality. Science 340(6134):1235009. doi:10.1126/science.1235009

Voss KA, Famiglietti JS, Lo M, Linage C, Rodell M, Swenson SC (2013) Groundwater depletion in the Middle East from GRACE with implications for transboundary water management in the Tigris-Euphrates-Western Iran region. Water Resour Res 49(2):904–914. doi:10.1002/wrcr.20078

Vryzas Z, Papadakis EN, Vassiliou G, Papadopoulou-Mourkidou E (2012) Occurrence of pesticides in transboundary aquifers of North-eastern Greece. Sci Total Environ 441:41–48. doi:10.1016/j.scitotenv.2012.09.074

Wada Y, Heinrich L (2013) Assessment of transboundary aquifers of the world-vulnerability arising from human water use. Environ Res Lett 8(2). doi:10.1088/1748-9326/8/2/024003

Wada Y, van Beek LPH, van Kempen CM, Reckman JWTM, Vasak S, Bierkens MFP (2010) Global depletion of groundwater resources. Geophys Res Lett 37(20), L20402. doi:10.1029/2010gl044571

Wada Y, van Beek LPH, Sperna Weiland FC, Chao BF, Wu Y-H, Bierkens MFP (2012) Past and future contribution of global groundwater depletion to sea-level rise. Geophys Res Lett 39(9), L09402. doi:10.1029/2012gl051230

Wang J, Huang J, Rozelle S, Huang Q, Blanke A (2007) Agriculture and groundwater development in northern China: trends, institutional responses, and policy options. Water Policy 9(S1): 61–74

Wang J, Rothausen SGSA, Conway D, Zhang L, Xiong W, Holman IP, Li Y (2012) China's water–energy nexus: greenhouse-gas emissions from groundwater use for agriculture. Environ Res Lett 7(1):014035. doi:10.1088/1748-9326/7/1/014035

Werner AD, Bakker M, Post VEA, Vandenbohede A, Lu C, Ataie-Ashtiani B, Simmons CT, Barry DA (2013a) Seawater intrusion processes, investigation and management: recent advances and future challenges. Adv Water Resour 51(0):3–26. doi:10.1016/j.advwatres.2012.03.004

Werner AD, Zhang Q, Xue L, Smerdon BD, Li X, Zhu X, Yu L, Li L (2013b) An initial inventory and indexation of groundwater mega-depletion cases. Water Resour Manag 27(2):507–533

Wolf AT (1998) Conflict and cooperation along international waterways. Water Policy 1(2): 251–265

Wolf AT (2007) Shared waters: conflict and cooperation. Annu Rev Environ Resour 32(1): 241–269. doi:10.1146/annurev.energy.32.041006.101434

Wood EF, Roundy JK, Troy TJ, van Beek LPH, Bierkens MFP, Blyth E, de Roo A, Döll P, Ek M, Famiglietti J, Gochis D, van de Giesen N, Houser P, Jaffé PR, Kollet S, Lehner B, Lettenmaier DP, Peters-Lidard C, Sivapalan M, Sheffield J, Wade A, Whitehead P (2011) Hyperresolution global land surface modeling: meeting a grand challenge for monitoring Earth's terrestrial water. Water Resour Res 47(5), W05301. doi:10.1029/2010wr010090

Woody T (2009) Alternative energy projects stumble on a need for water. The New York Times September 29, 2009

World Bank (2010) Deep wells and prudence: towards pragmatic action for addressing groundwater overexploitation in India. Tech. rep, Washington, DC

World Commission on Environment and Development (1987) Our common future. Oxford University Press, Oxford/New York

Zektser IS (2012) Investigation of transboundary aquifers in Russia: modern state and main tasks. In: NATO advanced research workshop on sustainable use and protection of groundwater resources—transboundary water management proceedings, pp 79–85. doi:10.1007/978-94-007-3949-9_7

Zektser S, Loaiciga H, Wolf J (2005) Environmental impacts of groundwater overdraft: selected case studies in the southwestern United States. Environ Geol 47(3):396–404

Disentangling the Complexity of Groundwater Dependent Social-ecological Systems

3

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Abstract

Groundwater resources are part of larger social-ecological systems. In this chapter, we review the various dimensions of these complex systems in order to uncover the diversity of elements at stake in the evolution of an aquifer and the loci for possible actions to control its dynamics. Two case studies illustrate how the state of an aquifer is embedded in a web of biophysical and socio-political processes. We propose here a holistic view through an *IGM-scape* that describes the various possible pathways of evolution for a groundwater related social-ecological system. Then we describe the elements of this *IGM-scape* starting with physical entities and processes, including relations with surface water and quality issues. Interactions with society bring an additional layer of considerations, including decisions on groundwater abstraction, land use changes and even energy related choices. Finally we point out the policy levers for groundwater management and their possible consequences for an aquifer, taking into account the complexity of pathways opened by these levers.

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3.1 Introduction

As discussed in Chap. 1, aquifers are generally part of larger complex systems, increasingly referred to as social-ecological systems (Folke et al. 2005; Janssen et al. 2006; Olsson et al. 2006). Social-ecological systems are composed of interacting socio-economic, bio-physical and human-made components. All too often these groundwater-dependent social-ecological systems are studied from a single or narrow subset of perspectives. Groundwater quality and quantity, for example, are determined by physical flows (water, microbial population, chemical pollutants, etc.) which result from natural processes and human activities (Chap. 14). However, drivers of groundwater dynamics can only be fully understood by enlarging the scope of the analysis. Indeed, the evolution of pressures exerted on groundwater by socio-economic factors depends on non-water related policies such as urban development, agricultural or energy policy (Fig. 1.1). It is also influenced by global market, technological and societal changes. The intensity of the groundwater management challenge also depends on the existence and quality of alternative resources, such as surface water, other aquifers, imported resources, and non-conventional water sources. Furthermore, the evolution of human activities shapes land use patterns, and as a consequence the pressure on the resource. Human activities in turn are shaped by values, beliefs and norms (Chap. 19) and associated policies and governance (Part II, Chap. 6).

In this chapter, we propose a conceptual framework to describe the complexity of groundwater-dependent social-ecological systems, to understand their long term dynamics, and to present the main research and management challenges. It presents a new perspective on groundwater management through: (i) the explicit re-integration of aquifers within much larger social-ecological systems; (ii) the identification of factors important for the system; and (iii) management to promote sustainable use of groundwater resources.

We expect that with such a foundation, Integrated Groundwater Management (IGM), will be more efficient and effective in practice.

3.2 Groundwater: An Interaction Space of Several Interdependent Dynamics

Various processes connect important entities associated with an aquifer. These connections make up possible pathways with positive or negative outcomes on the aquifer. In this section Integrated Groundwater Management analysis and policies are evaluated, and entities and interactions are mapped. The framework presented focuses on the integration of components across and within the natural system, human system and governance setting dimensions, and the associated issues of concern (Chap. 1).

3.2.1 Crau Aquifer: A Water Circular Economy

The introductory example presented here is the Crau alluvial aquifer, located in southern France, east of the Rhone Delta. This agricultural area is known for its production of labeled high quality hay, irrigated with a traditional system of earthen gravity canals, developed in the sixteenth century. Water losses occurring in the canals and at field level contribute significantly to the recharge of the underlying alluvial aquifer (Mailhol and Merot 2008). Irrigation water is imported from the Durance River, a snow-fed regulated river with several competing uses along its course. The artificial recharge associated with flood irrigation has allowed groundwater use to develop. The aquifer is considered to have better water quality than the surface water, and is now used by a number of small cities nearby for water supply, by individual households, and by industries. The high water table in the aquifer also prevents sea water intrusion on its south-eastern fringe, where several industries are located. High water tables have also produced a specific agricultural and natural landscape considered as a regional heritage, to which people are culturally attached (Mérot et al. 2008).

This social-ecological system has evolved over several centuries, and is now threatened by a number of factors: (i) the traditional irrigation system needs maintenance work or redevelopment; (ii) the water abstraction fees charged by the Water Agency are increasing, pushing farmers to reduce water losses at canal and field level; (iii) there is increasing competition for the use of the Durance river basin, where the irrigation water originates. Several adaptation pathways are apparent. One pathway relies on modernization of the irrigation system, such as with drip irrigation to decrease water use and pumping in the aquifer, which will in turn drastically decrease aquifer recharge, with projected degradation of the amenities listed above. Another pathway consists in changes in surface irrigation techniques that generate little change in aquifer recharge, but is more expensive (Mérot et al. 2008). Non-farming beneficiaries of the externalities generated by high water tables (e.g., neighboring cities using the aquifer for domestic water supply) could also become part of the governance of the area and contribute funding to maintain the system.

This example illustrates the complexity of processes that determine the dynamics of a groundwater-dependent social-ecological system, highlighting the need to look beyond the confines of the aquifer.

3.2.2 The Gnangara Mound

Another example is the Gnangara Groundwater Mound, which currently supplies about half of the water needs of the city of Perth (population 1.8 million), Western Australia. In this social-ecological system, groundwater is an important source for the metropolitan water supply, irrigation of parks and gardens, horticulture and industry, and also supports a number of wetlands and groundwater dependent woodlands. However, groundwater levels have declined significantly over the past few decades, as a result of climate change, abstraction and land use changes.

Reduced rainfall has been identified as the major cause of groundwater level decline in the Mound (Yeserterner 2008). There has been strong evidence of a climate shift in the area since the mid-1970s, which has resulted in a 10–15 % reduction in annual rainfall and fewer storms with high rainfall intensities. Historically, such intense rainfall events were a key source of runoff for streams and recharge for aquifers (McFarlane et al. 2010). The climate shift has led to streamflow reductions of more than 50 % as a result of the reduced runoff in addition to the reduced contribution from groundwater due to the lowered watertable levels and loss of groundwater-surface water connectivity (Bates et al. 2008; Petrone et al. 2010). A significant increase in groundwater abstraction from the Gnangara Mound over the decades has coincided with the decline in surface water supplies, with some changes in water use indirectly driven through policy. For example the noticeable drop in reservoir levels in the late 1970s led to a ban on the use of drinking water for irrigating private gardens. These restrictions led to a surge in the number of private bores, from approximately 25,000 in 1975 to 65,000 in 1980 (McFarlane et al. 2010).

Land use factors affecting the Gnangara Mound include pine plantations, land clearance, bush fires, and urban development. Pine plantations have had the strongest influence on groundwater levels, particularly in areas of dense plantation due to increased evapotranspiration and reduced recharge (Yesertener 2008). Land use practices that reduce leaf areas, such as land clearance, plantation thinning and bushfires, can lead to groundwater level rises, however the raised levels generally occur only for a few years until the vegetation is re-established.

The lowered groundwater levels in the Mound have caused declines in total abundance of groundwater dependent plants, and shifts in species composition towards more drought-tolerant species (Froend and Sommer 2010). The declining groundwater levels have led to incidences of reduced groundwater quality, including salt water intrusion in some coastal and estuarine parts of the Gnangara Mound. The lowered groundwater levels have also contributed to the acidification of several wetlands in the area through the exposure of acid sulphate soils. Artificial maintenance of water levels has been shown to restore some of the impact of drought-induced acidification on macroinvertebrate communities but change the seasonal hydrological regime of the wetlands (Sommer and Horwitz 2009).

The Gnangara Mound case study highlights the intertwined connections between climate, surface water and groundwater resources, and the human and ecological communities that depend on groundwater, with relationships occurring through both direct and indirect pathways.

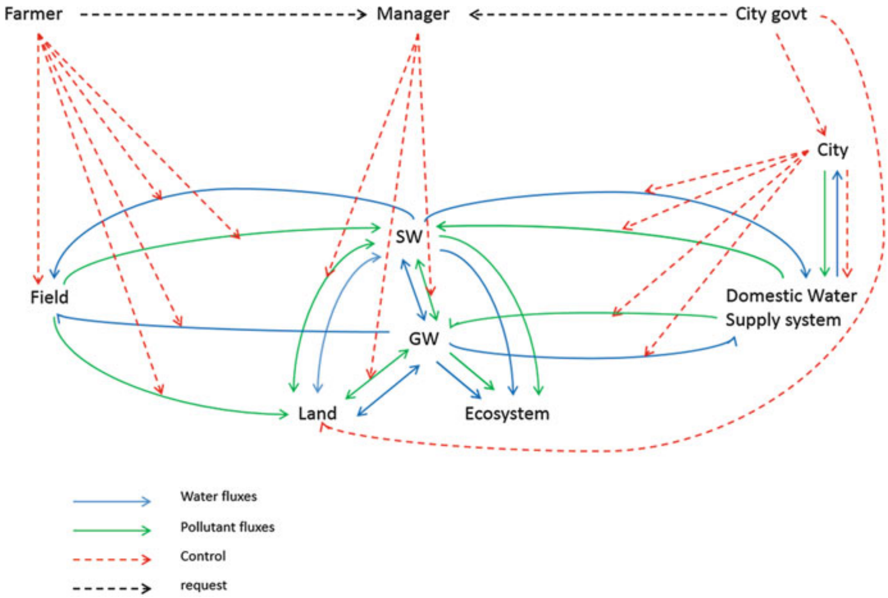

Fig. 3.1 Components at stake in groundwater management – The "IGM-scape"

3.2.3 An Enlarged and Integrated Perspective on Groundwater Management

The key external drivers, interactions and feedbacks, as well as a clear description of the metrics of desirable operation, are required so that performance and progress of any management can be evaluated. In this way, water becomes a means for optimization of a larger system rather than an end itself, and balances the needs for various water users, including the health of the ecosystems themselves. Main components and interactions to be taken into account for a social-ecological system features agricultural and domestic resource use (Fig. 3.1), and extends to include drivers like climate change (Chap. 5) and energy (Chap. 4). Altogether they constitute the "IGM-scape" – a holistic landscape of drivers important for effective IGM. A first circle lies around biophysical components: surface and ground water, but also related ecosystems and land. A second circle includes other material components through which water flows. A third circle represents main users, such as farming and urban development. The IGM-scape requires infrastructure; for example, conveyance mechanisms have to be available (Blomquist et al. 2001). Moreover, all these components are themselves dependent on external drivers: climate change, demographic development, markets, national or supranational policies, and knowledge development.

IGM benefits from such a starting framework because it identifies possible pathways that describe conduits for change to be expressed (i.e., external change or internal policy evolution). This enables identification of policy options, with subsequent conceptual and quantitative analysis of potential consequences.

IGM was defined in Chap. 1 as a structured process that promotes the coordinated management of groundwater and related resources in order to achieve shared economic, social welfare and ecosystem outcomes over space and time. To achieve this goal, IGM must: (i) identify most important pathways; (ii) consider policy options to control these pathways in line with holistic management objectives; and (iii) assess consequences and uncertainties. A key aspect of this stage is involving stakeholders and experts through use of hard and soft systems approaches (Chap. 24).

3.3 Understanding Hydrogeological Complexity

In this section we focus on hydrogeological processes that connect the "IGM-scape". Since these flows are not easily seen, we include some methodological suggestions on possible techniques to assess these fluxes.

3.3.1 Determinants of Groundwater Resource Quantity

An initial priority for groundwater managers to ensure a more sustainable exploitation of their aquifer (see also Chap. 2) is to determine how much water can be abstracted without depleting its quantity and degrading its quality, and minimizing negative impacts for other components of the groundwater-dependent social-ecological system. It means assessing flows between surface and ground water and the interactions with their environment. These components are covered individually below.

3.3.1.1 Aquifer Hydraulic Properties Characterization

The hydraulic properties of an aquifer can be characterized using specific hydraulic tests (Domenico and Schwartz 1990), in conjunction with upscaling using groundwater flow modeling (e.g., Anderson et al. 2015). Longer tests (at least 3-days) are typically needed to describe aquifer geometry, hydrodynamic characteristics, and the type of boundary conditions (impervious, constant-head, constant-flow) that are locally important (e.g., Kruseman and de Ridder 1994). The aquifer properties and insights can then be up-scaled using modeling to account for other hydraulic interactions such as with a river (Kollet and Zlotnik 2003) or sea-water intrusion (Terzić et al. 2007).

3.3.1.2 Aquifer Recharge Estimation

Generally, recharge is a process where infiltration from the terrestrial land surface or from surface water crosses the water table (see De Vries and Simmers (2002) and Scanlon et al. (2002) for a conceptual description of recharge). Recharge can be diffuse over large areas when caused by precipitation or irrigation. It can also be concentrated at specific locations such as tectonic fractures or preferential infiltration landforms (sinkholes in karstic systems, e.g., Andreo et al. 2008) or surface water bodies such as rivers or lakes. Recharge generally follows downward fluxes, but it can also lead to lateral fluxes in the case of interactions with neighboring aquifers or with surface water bodies such as rivers or lakes (De Vries and Simmers 2002). In the latter case, the flow direction can change depending on the season or the location, as it is mainly controlled by the water head gradient, i.e. the difference in altitude between the water table in the aquifer and in the river or the lake (Sophocleous 2002).

There are several methods to characterize the recharge of an aquifer (see Scanlon et al. 2002; Healy 2010). For discussion purposes, we group recharge characterization into direct measurement (physical or chemical), empirical and analytical or modeling methods.

Local measurements can be obtained using lysimeters. In order to obtain accurate measures of the recharge rate the base of the lysimeter should not be deeper than the root zone. By using 28 lysimeters (0.61 m diameter, 1 m long) in the Masser Recharge Site (central Pennsylvania, USA), Heppner et al. (2007) report that recharge averaged 32 % of the annual rainfall (ranging from 21 % to 52 %) between 1995 and 1999. Along with Seneviratne et al. (2012), they also discuss the main sources of uncertainty linked to the recharge estimation using lysimeter data.

Tracers (mainly chloride and environmental isotopes) can also be used to estimate local recharge. These methods are based on the analysis of the tracer concentration evolution between the input (in the rain water) and the output (springs, rivers or water table). For example, Marei et al. (2010) estimated the spatial distribution of recharge over the western side of the Jordan Rift Valley using chloride mass-balances. Tracer methods typically require several assumptions to account for anthropogenic perturbations, variability of climate, groundwater-rock chemical reactions, and difficulties inherent to output flux monitoring.

Empirical methods are based on linear correlations fitting between climate and recharge, and are typically calculated on an annual time scale. Although these correlations are generally specific to the climatic conditions of the locations calculated, they are relatively quick to perform. As an example, Kessler (1967) developed a way to optimize the calculation of recharge in carbonate aquifers, assuming that "*the amount of precipitation falling in the first four months of the year (that is, preceding the development of the vegetation and prior to the large losses due to evaporation) is determinative*". In order to consider the influence of the initial climatic context, a correction factor, derived from the amount of precipitation of the last four months of the previous year, is then applied. Finally, infiltration rates are proposed in order to estimate the recharge at the monthly time scale. This method has been applied to the Hungarian mountains, and later in a southern Spanish karstic aquifer where obtained results were realistic compared to other approaches (Andreo et al. 2008).

Modeling of aquifer recharge is typically most widely applied for large systems (see De Vries and Simmers (2002) and Scanlon et al. (2002) for extended reviews). There is a great variability in the approaches depending on the kind of data available to describe the aquifer dynamics. Simple hydrological balance methods such as those proposed by Thornthwaite (1948) or Dingman (2002) can be used to estimate infiltration out of the root zone and its availability for recharge. Therefore, the focus is on water that is not intercepted by the vegetation or consumed by evapotranspiration, nor lost as overland runoff. Typically, non-vegetated regions will have higher values than vegetated ones (Gee et al. 1994). The calculation of the distribution of infiltration (recharge) and other sinks is made using a combination of geomorphological, soil and lithology variables. There are methods that use spatially distributed information through GIS analysis (Mardhel et al. 2004) or though external computer codes designed to calculate recharge in space and time (e.g., Westenbroek et al. 2012). Typically models are applied at the daily time step, which can then be aggregated to longer time periods for groundwater analysis and modeling. Models simulating flow processes in the unsaturated zone can also be used to estimate recharge.

A variety of approaches for estimating recharge exist, ranging from soil-water storage-routing to numerical solutions to the Richards equation (see Scanlon et al. (2002) for an extended review); however their results can vary substantially. Fourteen different methods applied to the same arid setting in Nevada, USA led to recharge estimates ranging from 1 to 100 mm/year (Flint et al. 2002). A comparison of different methods to estimate recharge in another arid setting in the northern Sandveld area, Western Cape, South Africa also showed variability (Conrad et al. 2004), with estimates ranging from 0.2 % to 8 % of annual rainfall as recharge. Therefore, adequate description of how recharge was calculated for the IGM-scape is critical for acceptance by others.

3.3.1.3 Aquifer Interactions with Surface Water

Groundwater and surface water interaction is driven by hydraulic gradients (Gilfedder et al. 2012). The discharge of a river is often separated into two components, a fast and short response signal to rainfall corresponding to superficial and interflow sources and a slower response corresponding to aquifer drainage. Several techniques ranging from applying analytical methods for base flow separation to hydrographs (Gustard and Demuth 2009) to detailed hydrodynamic modeling or geochemical hydrograph separation (mainly using chloride concentration or stable isotopes of water) can be used to estimate the contribution of aquifer drainage to river discharge.

Commonly, aquifer water levels are highly sensitive to surface-water state, and can vary depending on the season (Allen et al. 2003). During high flows, river water typically recharges the aquifer and moves laterally away from the channel, causing groundwater levels to rise (Scibek et al. 2007); within a relatively short period after peak discharge, the groundwater flow direction is reversed. This is generally the case for river-aquifer interactions in natural conditions in humid climates. However, this relation can change in response to external stressors such as pumping. In some cases, water extracted from pumping wells can be almost exclusively derived from the surface water sources (e.g., Scibek et al. 2007). In some settings, extreme drought conditions and/or excessive pumping can lead to a complete river drying up. The potential for such adverse effects led regulators in several countries (e.g. France, Spain) to consider aquifer withdrawals close to rivers to be water withdrawals from the river itself. Even without complete drying, groundwater abstraction can affect ecological communities (Bradley et al. 2014; Chaps. 12 and 13).

In addition to well recognized surface water resources such as streams, rivers, and lakes, groundwater can also play a critical role for some wetlands (Chaps. 12 and 13). Groundwater contributes to the good ecological status of these water bodies through its effects on their physical and chemical characteristics. In terms of the IGM-scape, this importance has been recognized at the European level, where the EU Water Framework Directive (WFD) stipulates that groundwater abstraction must not unacceptably degrade ecological status of dependent wetlands. This relation to the groundwater system can be critically important even if the inflow from the aquifer represents a marginal part of the water supplying a wetland, and water levels in aquifers can represent the main environmental driver for wetland services (Gasca and Ross 2009).

Wetlands can also contribute significantly to the quality of the groundwater flowing through it, through soil characteristics that facilitate low oxidation-reduction conditions, filtration properties, and interaction with hydrophytic vegetation. This can be important for the retention and the recycling of some pollutants for groundwater, such as nitrates and pesticides, which can be important to consider in IGM approaches.

In the case of coastal aquifers, groundwater level decline due to pumping is one of the main causes of seawater intrusion, defined here as the landward subsurface incursion of seawater. Other factors such as land-use changes, climate variations or sea-level fluctuations also control the timing and magnitude of intrusion. Werner et al. (2013) provides a comprehensive review on the diversity of the challenges associated with seawater intrusion issues. Many diverse processes can influence IGM efforts. Dynamic hydrological conditions must be assessed taking into account density-salinity relationships. Together with the slow dynamics of the processes involved, it raises significant challenges for groundwater managers charged with determining optimal groundwater use. Effective groundwater management of coastal aquifers requires characterization of the position and thickness of the mixing zone between freshwater and intruding seawater (the seawater wedge toe) and monitoring that combines head measurements, geophysical methods, and environmental tracers. Simple measures such as head measurements in an observation well can be confounded by groundwater density effects caused by salinity and fluctuations at the toe in an observation well (Shalev et al. 2009). Geophysical methods typically can detect the large electrical resistivity contrast between seawater and freshwater, allowing 1D vertical or lateral to 3D characterizations (e.g., Poulsen et al. 2010). Even simple ion analysis of coastal groundwater can document seawater intrusion occurrence. High total dissolved solids in groundwater can also be caused by rock dissolution, connate saline water and irrigation return flow (e.g., Bouchaou et al. (2008).

3.3.2 Determinants of Groundwater Quality

Understanding infiltration processes, identifying flow direction, and information on aquifer lithology can provide first approximations of expected groundwater quality (see also Chaps. 14 and 15).

In addition to terrestrial recharge, surface water can supply appreciable recharge to an aquifer. The evolution of water quality in the surface-groundwater interaction context is typically influenced by several processes linked to geology (lithology of the aquifer, granulometry of the river banks), hydrogeology (aquifer permeability, confined/unconfined, clogging thickness and hydraulic conductivity of the river banks), hydrology (rain water chemistry, evaporation intensity, flow seasonality) and biology (temperature, micro-organisms, light, river bed vegetation, oxygenation and nitrate presence for the microbial activity). Interactions between surface water and aquifers can influence the water quality in both systems. The transition interface between surface water and aquifers (also called the hyporheic zone) can also play a significant role in the transformation and transport of pollution – for example by filtering suspended particles and interacting with bacteria, viruses, and organic matter. Longer residence times of the water in the hyporheic zone commonly enhance biogeochemical reactions that are favorable to a natural attenuation of pollution (Gandy et al. 2007). For example, when filtrating through river banks, several processes affecting water quality between surface and groundwater are involved (see Hiscock and Grischek 2002). Regional monitoring networks for surface and groundwater show that poor chemical conditions of shallow groundwater lead to lower quality in receiving surface waters, and monitoring of the water quality of surface water during non-storm conditions can provide an integrated measure of groundwater quality. Alternatively, when surface water recharges an aquifer, monitoring of surface water quality can provide warnings of potential aquifer contamination.

Groundwater-surface water interaction, and the water quality ramifications, are often influenced by hydrologic stress applied to either system. Stresses such as pumping and dam construction, for example, can influence the flow direction between aquifers and rivers and change the residence time within the hyporheic zone. Large hydrologic stress can also appreciably affect aquifer hydraulic properties through development of unsaturated conditions beneath the river, due to abstraction rates higher than can be supported by capture from the surface water resource.

3.4 Understanding the Complexity of Groundwater-Society Interactions

Over centuries, changes to water infrastructures and land use have significantly altered hydrogeological processes, frequently affecting groundwater and dependent ecosystems. Effective IGM requires understanding of these two drivers, and appropriate integration of the relevant components within and across the natural and human systems.

3.4.1 Infrastructures and Increased Human Interference in the Water Cycle

3.4.1.1 Groundwater Abstraction

The development of groundwater abstraction infrastructures, for urban, industrial and agricultural uses, is perhaps the most obvious driver in the IGM-scape. Although traditional exploitation technologies (e.g., Persian wells, galleries in the Middle East) were relatively small stresses to the groundwater system, the development of modern pumping technologies has increased groundwater use by several orders of magnitude. New problems of groundwater depletion have resulted, including sea water intrusion, land subsidence, and reduced river, spring, and wetland flows (see Chap. 2 for an overview of these problems and their international scale). Increased exploitation has also resulted in greater seasonal and annual fluctuation of groundwater levels, frequently impacting dependent ecosystems and groundwater quality. As an example, groundwater is a source of clean water for more than 13 million people in Kolkata, India, but its quality is appreciably degrading due to intensive pumping that has induced recharge from areas of known contamination with heavy metals and arsenic (Sahu et al. 2013). Pumping in groundwater increases vertical gradients and related velocities from surface water sources (Gilfedder et al. 2012). Some studies report intensive withdrawal impacting not only on the capacity of other people to pump in the same resource but also on return flows from groundwater to surface water in low water period that can be reversed (Howe 2002; Webb and Leake 2006).

Understanding the effects of groundwater development is essential to IGM. Tradeoffs must be recognized; in agriculture, the construction of private borewells has improved the living conditions of millions of farmers, in developed as well as in developing countries (Llamas and Martinez-Santos 2005). Accessing groundwater increases autonomy, thus flexibility with regards to production, and ultimately income. Pumping from the groundwater system also improves water supply reliability, in particular during drought (Tsur 1990; Tsur and Graham-Tomasi 1991). Municipal water utilities increasingly use groundwater to complement surface water supplies, again for increasing reliability of supply during drought or drier climate (e.g. the Gnangara Mound in Western Australia), or in the case of catastrophic events like floods, landslides, earthquakes or large scale nuclear contamination (Vrba and Verhagen 2011). Commonly industries develop groundwater self-supplies rather than purchase water from municipal utilities. Similarly, households may be tempted to drill bore wells for private use as in Perth (the Gnangara Mound case study above; Rinaudo et al. 2015); this phenomena has also been reported in other cities like Cape Town in south Africa (Saayman and Adams 2002), and southern France (Montginoul and Rinaudo 2011).

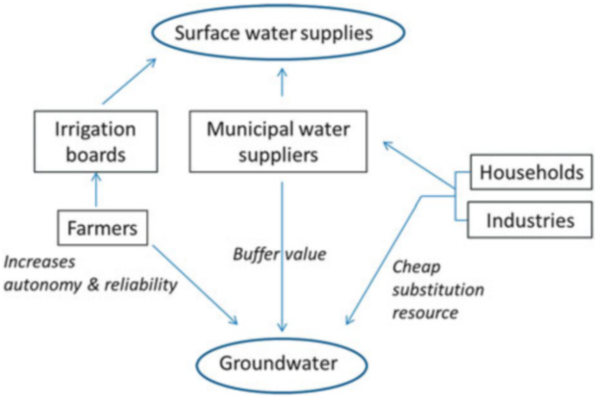

Fig. 3.2 Motivations for substituting surface with groundwater

Overall, the development of groundwater use reflects the decision of various categories of economic agents to substitute their traditional collective surface water supply with independent groundwater supply (see Fig. 3.2). Understanding the motivations underlying individuals' decision to undertake this shift in water supply source is essential to design an effective groundwater protection policy. Groundwater management policy needs to use policy levers that interface with other policies, such as pricing policies of agricultural or urban water services.

3.4.1.2 Irrigation and Drainage

In many parts of the world, the development of irrigation and drainage (Chap. 15) has been a key factor affecting groundwater dynamics. The construction of large scale gravity irrigation structures, which divert water from surface sources over long distances, has appreciably increased groundwater recharge, through water losses that occur in canals and at farm level. In this way, the groundwater cycle is made more artificial, generating significant unintended effects – both good and bad – for non-agricultural users (e.g., development of new surface ecosystems, waterlogging and enhanced salinization).

Scarcity of surface water resources led national and international agencies to promote more efficient surface irrigation schemes. Ancient gravity irrigation systems are progressively being turned into piped infrastructures, delivering pressurized water at farm level, where sprinkler and drip irrigation replace inefficient flood irrigation. While the technical and economic efficiency of irrigation has been rising, irrigation losses and artificial recharge of shallow aquifers is being reduced. Many unintended benefits generated for decades by gravity irrigation schemes are suddenly offset, as illustrated by the Crau case study presented earlier. This again illustrates the need for greater integration of various policy domains to ensure sustainable groundwater management.

3.4.1.3 Artificial Groundwater Recharge

Infrastructures have also been designed to increase aquifer recharge by using water diverted from rivers during high flow periods or with treated wastewater. Several

Managed Artificial Recharge (MAR) techniques are now available to increase infiltration as well as to treat water through soil processes (see Chaps. 16 and 17). In this way, groundwater can be considered as a natural infrastructure for water storage. Consistent with an aquifer and surface water being a single resource, MAR slows down surface water flows and/or facilitates soil infiltration in dedicated places via infrastructures such as infiltration pounds or ditches, or injection wells. Despite potential design uncertainties, it has been now successfully implemented in various arid or semi-arid places of the world, such as the Llobregat basin near Barcelona (Pedretti et al. 2012) or in the southwestern United States (Blomquist et al. 2001). “In lieu recharge” is a similar management technique, which calls for the use of surface water first, hence keeping groundwater stored in aquifers for future use only when required. Diversion is performed first in the input flow before tapping into the groundwater storage, rather than tapping groundwater storage filled by a MAR process somewhere else. This approach needs accessible surface water, but it has been used even in water scarce areas such as the southwestern United States (Blomquist et al. 2001). One impediment to wider implementation of MAR lies in the legal definition of ownership of recharged water. Economic investment in MAR infrastructure is often contingent on the ability to recover the volume stored in the aquifer at a later point in time, as it happens in Kern County Groundwater Bank (Hanak and Stryjewski 2012).

Artificial recharge may also take place at smaller scales, such as in households to re-infiltrate rain water collected from their roofs. The promotion of such decentralized artificial recharge schemes is often a feature of urban development planning and policy. The concept of water sensitive urban design is gaining momentum (Hussey and Kay 2015) but issues regarding property rights can affect ownership of re-infiltrating roof water into the aquifer. Artificial recharge also can target improving poor quality, such as in Teheran, Iran, where 60 % of domestic wastewater is re-injected into aquifers through some three million wells spread across the area (Bazargan-Lari et al. 2009). Once again, IGM for improving the groundwater resource is clearly affected by the integration of groundwater and urban development policies.

3.4.2 The Impacts of Land Use Change on Groundwater

The groundwater cycle can be significantly altered by land use changes (LUC). Land use influences local aquifer recharge and the quantity of pollutants produced at a point or diffuse source. IGM policy thus has to account for LUC, which calls for better understanding of LUC drivers and their impacts on the subsurface portion of the hydrological cycle. The four main LUCs impacting groundwater recharge and quality are shown in Fig. 3.3. Increased local demand for food or international market incentives (cash crops) generate significant conversion of natural landscapes (forest, rangeland, shrubland, wetlands) into agricultural land (❶). The opposite evolution is also reported in poor agricultural areas, where cultivated land is progressively abandoned due to economic pressures and migration of the

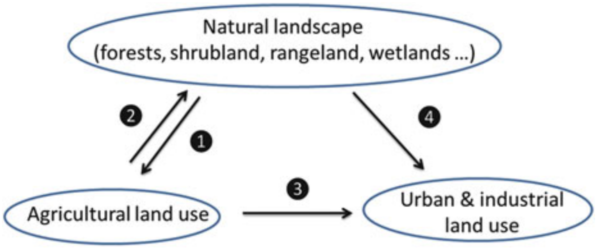

Fig. 3.3 Land use changes affecting groundwater recharge and quality

rural population towards cities (❷). Concentration of population in urban area results in massive conversion of agricultural land and/or natural land into housing, transport, commercial or industrial land use – often involving a reduction in groundwater recharge over large areas (❸ and ❹).

3.4.2.1 Agricultural Development and Groundwater

The conversion of natural lands into agricultural land impacts the water cycle in four different ways. First, change in vegetation cover significantly alters evapotranspiration patterns. In the early growing season, agricultural crops have a lower evapotranspiration than natural vegetation. Infiltration is increased due to the high proportion of bare soil in early crop stages. Infiltration is also higher during fallow periods due to reduced plant interception and the presence of bare soil. Additionally, plowing and other farming practices such as terracing increase permeability of upper soils, thus facilitating infiltration beyond the capture of the root zone. Alternatively, compaction of soil by heavy farm machinery may reduce infiltration and enhance surface runoff (Steuer and Hunt 2001). Lastly, the conversion of natural land into agriculture is often accompanied by the development of irrigation based on imported water supply, which further increases recharge. A number of studies have demonstrated that the conversion of natural land into agricultural fields increases recharge, under various climates. In the western states of the USA, in semi-arid parts of Australia, and in the Indian subcontinent, the process has resulted in significant rise of the water table, waterlogging and soil salinization (see Chaps. 2 and 15). In Sri Lanka deforestation associated with agricultural development has caused an increase in groundwater recharge (Priyantha Ranjan et al. 2006). In addition, the water quality of infiltrating water changes, which can affect use of the groundwater resources (see Chap. 15).

3.4.2.2 Urban and Industrial Land Use

Urbanization also influences the subsurface flow regime and groundwater quality in three main ways. The increase in impervious surfaces results in: (i) reduced infiltration and recharge; (ii) reduced evapotranspiration; and (iii) possible increases in groundwater abstraction by industrial and commercial activities which do not necessarily require high quality water, and sometimes by households tapping shallow aquifers for irrigation (Rinaudo et al. 2015). Urban development policies and planning can influence the degree of impact of these factors, for example, by careful selection of locations for large impervious surfaces (industrial and commercial sites, transportation infrastructure), associated mitigation, and promoting low impact designs (Dams et al. 2008; Cho et al. 2009). Water sensitive urban design can result in increasing recharge and available groundwater resources, by redirecting runoff from roofs and roads into the soil and thereby the shallow aquifer (Wong 2006; Barron et al. 2013; Hussey and Kay 2015). In extreme cases, urbanization accompanied with infiltration of storm water can lead to a long term rise of water tables (Barron et al. 2013). In this way LUC can have similar impacts to managed artificial recharge infrastructure – yet LUC has two main advantages, of larger cost distribution and spatial distribution over a large area.

A second main impact of urbanization is on groundwater quality (Lawrence et al. 1998) as economic, industrial and commercial development introduces new potential contamination sources. Point source pollution, due to accidental spillages or long term leakages of chemical products, can generate large pollution plumes (petroleum, chlorinated hydrocarbons, and synthetic organic compounds) that are often mixed with other contamination sources. Contaminated soils form a more diffuse contamination source. Small size industries such as tanneries, printing, laundries, and metal processing, can be widely dispersed and generate liquid effluents such as spent disinfectants, solvents, lubricants that often reside in adjacent soil. Leakage from wastewater lagoons and sanitary sewer systems can also be appreciable. Storm water can carry significant loads from impervious surfaces as well as pathogenic bacteria and viruses. Pathogen water quality issues can result in areas where sanitary treatment is deficient (cesspit, latrines, and septic tanks) or even through aging infrastructure where treatment methods are well developed (e.g., Hunt et al. 2010).

3.4.3 Energy: Groundwater Policy Interactions

Groundwater can also be significantly affected by changes in energy policy (see Chap. 4 which covers the water-energy-global change nexus). In countries where electricity is widely available in rural areas, some authors suggest that an important lever to ensure sustainable groundwater management policies is electricity pricing policy (Scott and Shah 2004; Shah et al. 2008). Energy pricing can lead to unintended effects: Moroccan and Indian government subsidies of respectively domestic gas cylinders and electricity were intended for social welfare; however, farmers changed or adapted their pump engines to benefit from subsidies, resulting in an unintended increased of groundwater use for irrigated agriculture and overexploitation (Shah et al. 2008; Shah 2014).

Through the energy-water nexus, groundwater policy can also conflict with renewable energy development policies. In solar energy for instance, a range of technological innovations are being adopted by industry, and their development might impact groundwater in the future (Mills 2004). The principle of thermo-solar power plants consists of harnessing solar energy to generate electrical production with steam turbines, which require the use of large quantities of cooling water.

Geothermal power plants use more water than conventional steam plants because of low heat-electricity conversion efficiency (Fthenakis and Kim 2010). Energy policy thus results in increased water demand, conflicting with a water conservation objective. The problem can be particularly acute in arid areas, which are characterized by high solar radiation and scarce water resources often stored in aquifers. In southern Spain, the development of thermo-solar power plants has already resulted in a transfer (and a concentration) of groundwater rights from agriculture to the energy sector, generating new groundwater management problems (Berbel, personal communication 2013).

Other issues may also occur with the development of low enthalpy geothermal energy, which uses large quantities of groundwater without recycling (open system). Where such open systems dominate, a competition for the groundwater resource could arise in the near future, between the low geothermal energy and drinking and agricultural water supply.

3.5 Policies for the IGM-Scape

The first order interactions between groundwater and society listed above (infrastructures, land use changes, or water energy nexus) have second order interactions when we include the impacts of one of them on another one. As such, their impacts on groundwater could be alleviated or magnified whenever they occur simultaneously, providing a strong impetus for an efficient governance setting for IGM and pathways across the IGM-scape of Fig. 3.1.

3.5.1 Policy Levers to Promote Sustainable Groundwater Management

Policy levers (as discussed in Parts II and IV) can be intentionally focused on the components (Sect. 5.1.1) or on fluxes (Sect. 5.1.2) of the IGM-scape as described on Fig. 3.1. The component versus flux distinction holds only at the level of intention of policy levers. Consequences of their activation disseminate all along pathways of the IGM-scape.

3.5.1.1 Policies Tackling Components of the IGM-Scape

Due to the connections across the IGM-scape, policies to promote sustainable groundwater management can either try to tackle head-on the isolated groundwater component of an aquifer system, or focus on a combination of components present in its IGM-scape. The hidden nature of the groundwater resource (Chap. 1) makes it difficult to effectively address directly; a focus on multiple components will likely be more effective.

Land use is a component that is highly suitable as a policy target. Therefore, controlling land use change is a key lever for ensuring sustainable management of groundwater as a matter of quantity as well as quality. In current practice, these levers can include:

– Innovative practices that favor recharge,
– Rules on urbanization that reduce impermeability of surfaces,
– Incentives to maintain agriculture instead of other urban land uses.

In the Perth region, for example, recognition of the impact of pine plantations on the groundwater levels led to a decision to progressively phase out the plantations on the Gnangara Mound by around 2030 (MacFarlane et al. 2010).

Fields and farming practices constitute a specific land use that can be more specifically controlled, first for improving groundwater quality and second for reducing the quantity of water withdrawn:

– Rules on agriculture practices can limit the use of potential pollutants, especially in domestic water supply catchment areas,
– Rules and economic incentives for crops with lower water demand can reduce abstraction.

These actions are targeted to farmers leading them to practices on their land suitable for larger aquifer system sustainability. Similar actions exist for urban uses, such as rules regarding digging private wells or economic incentives to implement low impact development techniques such as garden roofs. Inter-basin transfer of surface water is a similar lever, often with a direct impact on recharge due to leakage and infiltration occurring in canals, but also alleviation of needs in the area receiving water transfer.

3.5.1.2 Policies Tackling Fluxes in the IGM-scape

More direct policies can tackle fluxes in the IGM-scape, with emphasis on fluxes that end up in the aquifer. Artificial recharge is a policy lever that increases the flow capacity from surface to ground water. Still on the quantity side, one of the most common policies in water management deals with maximum abstraction flow controls. Typically the primary focus is on water scarcity and irrigation, where policy is designed to control abstraction with acceptable impacts on groundwater levels. With such a focus, levers can include simple actions such as equipping farmers with flow measuring devices.

On the quality aspects of fluxes in the IGM-scape, several means exist to mitigate poor water quality such as from pesticide pollution in a drained basin. In such settings, efforts focus on capturing pesticide before introduction into the groundwater system. These efforts might include focusing on enhancing ecosystem services provided by soil and vegetation. In practice several types of these levers exist, such as ditch networks and artificial wetlands (Stehle et al. 2011; Tournebize et al. 2012). The principle is either to treat the flux directly, or to divert it into parts of the ecosystem that can mitigate aspects of poor water quality.

3.5.2 Pathways Opened Up by These Policy Levers ... and Others

The existence of externalities is a rule more than an exception, as far as water is concerned (Howe 2002). We generalize the concept of externality to any type of unintended side effect, beyond the targeted economic domain. However, water availability and quality are also affected by externalities generated by actions with no direct intervention on water flows as well. Decisions regarding land use change, for example, have feedback loops that augment and mitigate the source of externalities coming from groundwater management choices, while others are rooted elsewhere.

Whether driven by groundwater concerns or not, the groundwater-dependent social-ecological system changes are constrained along the pathways partly explained in the IGM-scape, due to such feedback and cascade effects, where each step includes uncertainty. Therefore, uncertainty issues are important to consider along with the feedback and cascade effects (see Chap. 28 for coverage of uncertainty).

3.5.2.1 Policies with Indirect Effect on Groundwater

Most components of an "IGM-scape" are typically responding to actions of other non-groundwater focused policies. Policies affecting land uses are one easily seen example because they modify water needs, water direct abstraction, infiltration rates and the capture of solutes. Urban development policies are also typically driven by concerns outside of the realm of water management policies. Even when urban development is supposed to be consistent with water management regulations, local policy makers find ways to get around the rules (Barone 2012).

Affected parties may mitigate sources of adverse externalities. Mitigation may not only be directed at water flows, or even affected parties downstream. Yet, many of these mitigation actions modify flows indirectly. For example, in France groundwater used by a private company to produce highly valued mineral water was being negatively impacted by nearby nonpoint source pollution associated with farming. As a consequence, the private company offered funds to farmers if they followed specific cropping patterns with less impact on the water quality (Deffontaines et al. 2000). Dealing with externalities is often in conjunction with payments for ecosystem services, such as flood protection of cropping areas through compensation to cover losses (Erdlenbruch et al. 2009).

Yet, changes to the system driven by externalities, like many groundwater changes, are often masked by long time lags between the change and the expression of their consequences. Moreover, in some cases changes resulting from externalities may occur with little consideration regarding water. For example, switching from one crop type to another at a farm level is typically an economic decision. Yet, competing societal use of water can drive IGM decision making. For example, surface water may be progressively reserved for uses other than irrigation as human populations increase (Gemma and Tsur 2007).

3.5.2.2 Uncertainties in Groundwater-Related Social-Ecological Systems Dynamics

The previous discussion implicitly includes uncertainties (see Chap. 28), one of the salient dimensions of integrated assessment and modelling (Chap. 1). Beyond long term uncertainties, such as on climate change, IGM must handle uncertainties such as knowledge gaps, stochastic processes and external choices.

Henriksen et al. (2011) and Chap. 28 provide a good overview of sources of uncertainties associated with groundwater management. Implementation of managed aquifer recharge involves groundwater managers to make use of assumptions or imperfect representations of important processes, such as transfer of fluxes between surface water and groundwater. Socio-economic processes are also uncertain, since behavioral patterns of water users are never fully determined by their conditions of action as set by their social, economic and ecological environment. Managers have to monitor these uses and to constantly adapt and learn.

Several stochastic processes are also important. Rain and evaporation, as sources and sinks, constitute two easy to appreciate examples, but other forces like market prices also commonly possess a stochastic nature over various timescales. In general, stochastic processes can be associated with probabilities, which in turn can be used to assess the IGM-scape. Finally, external drivers to IGM like climate and international trade prices, present additional uncertainty as they involve choices beyond that of the domain of groundwater management. These influences have their own determinants and sources of uncertainties that may not be readily apparent to groundwater managers. In summary, the presence of such wide ranging sources of uncertainty underscores the need for adaptive understanding and flexibility for moving within the IGM-scape.

3.5.3 The Governance Challenge Extended

Throughout our discussion, several institutional factors can be seen as pushing the groundwater related social-ecological system along one pathway or another. Selection of policy levers as well as the complexity of the social-ecological system challenge governance frameworks. We consider that these challenges are of two types:

- a legitimacy challenge in order to involve the suitable people within the arena of IGM, i.e. those who are entitled to act on the components and fluxes all along the pathways of IGM-scape;
- a policy challenge that results in getting politically powerful groups to prioritize IGM issues.

3.5.3.1 The Legitimacy Challenge

Typically, government agencies remain the main regulator over land use, and often have the authority to limit possibilities of actions on water flows that would generate consequences unsuitable with the rights of others. However, the possibility of implementing effective controls depends on institutional authority and standing (see the chapters in the governance section). And, in practice, financial costs, land and water rights, transaction costs among the multiple stakeholders, can facilitate or impede actions implemented by policy makers (Blomquist et al. 2001). Availability of an appropriate knowledge base and suitable technologies is also a factor in implementing change.

Water rights are typically not straightforward, especially for hard to characterize aspects such as how ownership of land translates into ownership of terrestrial recharge and how competing uses of recharged water are prioritized (see Chap. 9). Institutions also commonly seek to establish benchmarks to assess use and its effects on recognized rights. Unfortunately, there is no widely accepted way to uniquely determine such benchmarks; rather, they typically result from site-specific historical precedent, economic drivers, perceptions of suitability for local land use policy, etc.

Setting water and non-water priorities can become a primary governance challenge. In many cases, the drivers come from outside formal governance entities, such as when a company sets its price for surface water delivery: it frames the choice of the farmer in using one or the other source, as an economic choice. Doing so, the company produces a major driver on groundwater use, but may not be part of the arena where groundwater management is discussed (Lenouvel and Montginoul 2010). In some cases the drivers are appreciably different. For example, land and water resources can be separated by law; hence, forestry companies are entitled to develop their land, but the impact on groundwater recharge and level can create conflicts with a farming sector also entitled to develop their land (Gillet et al. 2014). At the extreme, stakeholders involved in arenas with major impacts may neither be interested nor have legitimacy to regulate or act on groundwater, such as the case of interaction among various policy sectors (e.g., energy and agriculture). Tradeoffs are required, however appropriate criteria and frameworks for evaluating the tradeoff may be difficult to construct and legitimize.

3.5.3.2 Promoting Water at Policy Level

Even if technical and legal challenges are met, there is a need for policy support by the regulated public so that groundwater is prioritized appropriately with respect to other policy issues. However, interest in the policy may not be automatic, and other entities that are already prioritized highly may not be keen to enter a competing realm involving IGM policies. In practice, hidden benefits of appropriate groundwater management commonly become subordinate to other more visible benefits from land development, even when the law puts water first. Yet, when evaluated, even though it is hidden, groundwater conservation often appears as a first priority among respondents (Razès et al. 2013).

3.6 Conclusions

Aquifers are embedded in larger social-ecological systems whose components generate various multiple feedbacks impacting the state of the aquifer. All these components and their relations constitute an "IGM-scape", featuring potential pathways of evolution for groundwater as well as the social ecological systems in which it is embedded. An IGM-scape is based partly on physical components and fluxes. It increases the accuracy of the assessment of water flows and hence of water availability in the aquifer in pointing out the suitable levers to regulate it. In its most encompassing form, the IGM-scape extends this approach beyond physical processes, opening it up to institutional issues and interdisciplinary drivers. As a consequence, IGM must take into account non-water components in the system, including land, ecosystems, and economic drivers. Such holistic views of the IGM-scape facilitate the application of suitable levers for groundwater management.

Effective management of the IGM-scape requires, at a minimum, joint management of surface and groundwater at suitable scales. Management concerns and scale are temporal as well as spatial. If groundwater storage is a stated benefit of the IGM-scape, intervention to preserve surface water from being "lost" to groundwater reduces possible future uses and can affect larger areas when the aquifer at stake is transgressing boundaries, whether jurisdictional or attached to a river basin. Transfers across these boundaries need an IGM-scape approach to governance and explicit negotiation. Timing and lags between changes in land uses, water uses, and regulations may not be consistent. As such, effective management of the IGM-scape must recognize potentially irreversible consequences or important hysteresis effects, such as changes in soil structure, economies of scale with regard to costs of infrastructures, and important tipping points and thresholds that exist such as in the case of pollution of an aquifer. Although disconnection of water policies from other public policies has long been pointed out as a major issue for water governance, explicit recognition of the ties and pathways that characterize the IGM-scape is a first step towards effective integrated governance, so that inclusion of all important stakeholders in IGM arenas is possible.

References

Allen DM, Mackie DC, Wei M (2003) Groundwater and climate change: a sensitivity analysis for the Grand Forks aquifer, southern British Columbia. Can Hydrogeol J 12(3):270–290

Anderson MP, Woessner WW, Hunt RJ (2015) Applied groundwater modeling: simulation of flow and advective transports, 2nd edn. Elsevier, Amsterdam

Andreo B, Vías J, Durán JJ, Jiménez P, López-Geta JA, Carrasco F (2008) Methodology for groundwater recharge assessment in carbonate aquifers: application to pilot sites in southern Spain. Hydrogeol J 16:911–925

Barone S (2012) SCoT est-il plus SAGE ? Vertigo 12(2). http://vertigo.revues.org/12460

Barron OV, Barr AD, Donn MJ (2013) Effect of urbanisation on the water balance of a catchment with shallow groundwater. J Hydrol 485:162–176

Bates B, Hope P, Ryan B, Smith I, Charles S (2008) Key findings from the Indian ocean climate initiative and their impact on policy development in Australia. Clim Change 89:339–354

Bazargan-Lari MR, Kerachian R, Mansoori A (2009) A conflict-resolution model for the conjunctive use of surface and groundwater resources that considers water-quality issues: a case study. Environ Manage 43:70–482

Blomquist W, Heikkila T, Schlager E (2001) Institutions and conjunctive water management among three western states. Nat Resour J 41:653–684

Bouchaou L, Michelot JL, Vengosh A, Hsissou Y, Qurtobi M, Gaye CB, Bullen TD, Zuppi GM (2008) Application of multiple isotopic and geochemical tracers for investigation of recharge, salinisation, and residence time of water in the Souss-Massa aquifer, southwest Morocco. J Hydrol 352:267–287

Bradley DC, Streetly M, Farren E, Cadman D, Banham A (2014) Establishing hydroecological relationships to manage the impacts of groundwater abstraction. Water Environ J 28:114–123

Cho J, Barone VA, Mostaghimi S (2009) Simulation of land use impacts on groundwater levels and streamflow in a Virginia watershed. Agric Water Manag 96:1–11

Conrad J, Nel J, Wentzel J (2004) The challenges and implications of assessing groundwater recharge: a case study – northern Sandveld, Western Cape, South Africa. Water SA 30(5): 75–81

Dams J, Woldeamlak ST, Batelaan O (2008) Predicting land-use change and its impact on the groundwater system of the Kleine Nete catchment Belgium. Hydrol Earth Syst Sci 12: 1369–1385

De Vries JJ, Simmers I (2002) Groundwater recharge: an overview of processes and challenges. Hydrogeol J 10:5–17

Deffontaines J-P, Brossier J, Barbier M, Benoît M, Chia E, Fiorelli JL, Gafsi M, Gras F, Lemery H, Roux M (2000) Water quality, agricultural practices and changes in farming and agrarian systems. In: Collinson MP (ed) A history of farming systems research. CABI Publishing, New York, pp 382–390

Dingman L (2002) Physical hydrology, 2nd edn. Waveland Press, Long Grove. ISBN:978-1-57766-561-8

Domenico PA, Schwartz FW (1990) Physical and chemical hydrogeology. Wiley, New York, p 824

Erdlenbruch K, Thoyer S, Grelot F, Kast R, Enjolras G (2009) Risk-sharing policies in the context of the French food prevention action programmes. J Environ Manage 91:363–369

Flint AL, Flint LE, Kwicklis EM, Fabryka-Martin JT, Bodvarsson GS (2002) Estimating recharge at Yucca Mountain, Nevada, USA: comparison of methods. Hydrogeol J 10:180–204

Folke C, Hahn T, Olsson P, Norberg J (2005) Adaptive governance of social-ecological systems. Annu Rev Environ Resour 30:441–473

Froend R, Sommer B (2010) Phreatophytic vegetation response to climatic and abstraction-induced groundwater drawdown: examples of long-term spatial and temporal variability in community response. Ecol Eng 36:1191–1200

Fthenakis V, Kim HC (2010) Life-cycle uses of water in U.S. electricity generation. Renew Sustain Energy Rev 14:2039–2048

Gandy CJ, Smith JWN, Jarvis AP (2007) Attenuation of mining-derived pollutants in the hyporheic zone: a review. Sci Total Environ 373(2–3):435–446. doi:10.1016/j.scitotenv.2006.11.004

Gasca D, Ross D (2009) The use of wetland water balances to link hydrogeological processes to ecological effects. Hydrogeol J 17:115–133. doi:10.1007/s10040-008-0407-x

Gee GW, Wierenga PJ, Andraski BJ, Young MH, Fayer MJ, Rockhold ML (1994) Variations in water balance and recharge potential at three western desert sites. Soil Sci Soc Am J 58(1): 63–72

Gemma M, Tsur Y (2007) The stabilization value of groundwater and conjunctive water management under uncertainty. Rev Agric Econ 29:540–548

Gilfedder M, Rassam DW, Stenson MP, Jolly ID, Walker GR, Littleboy M (2012) Incorporating land-use changes and surface groundwater interactions in a simple catchment water yield model. Environ Model Software 38:62–73

Gillet V, McKay J, Keremane G (2014) Moving from local to state water governance to resolve a local conflict between irrigated agriculture and commercial forestry in South Australia. J Hydrol 519(C):2456–2467

Gustard A, Demuth S (2009) Manual on low flow estimation and prediction, Operational hydrology report no 50, WMO No 1029. World Meteorological Organization, Geneva

Hanak E, Stryjewski E (2012) California's water market, by the numbers: update 2012. PPIC Press/Public Policy Institute of California, San Francisco

Healy RW (2010) Estimating groundwater recharge. Cambridge University Press, Cambridge, 245

Henriksen HJ, Zorilla-Miras P, de la Hera A, Brugnach M (2011) Use of Bayesian belief networks for dealing with ambiguity in integrated groundwater management. Integr Environ Assess Manag 8:430–444

Heppner CS, Nimmo JR, Folmar GJ, Gburek WJ, Risser DW (2007) Multiple methods investigation of recharge at a humid-region fractured rock site, Pennsylvania, USA. Hydrogeol J 15(5): 915–927

Hiscock KM, Grischek T (2002) Attenuation of groundwater pollution by bank filtration. J Hydrol 266:139–144

Howe CW (2002) Policy issues and institutional impediments in the management of groundwater. Environ Dev Econ 7:625–641

Hunt RJ, Borchardt MA, Richards KD, Spencer SK (2010) Assessment of sewer source contamination of drinking water wells using tracers and human enteric viruses. Environ Sci Technol 44(20):7956–7963. doi:10.1021/es100698m

Hussey K, Kay E (2015) The opportunities and challenges of implementing water sensitive urban design: lessons from stormwater management in Victoria, Australia. In: Grafton Q, Chan N, Daniell K, Nauge C, Rinaudo J.-D (eds) Understanding and managing urban water in transition. Springer, Dordrecht (in press)

Janssen M, Bodin O, Anderies JM, Elmqvist T, Ernstson H, McAllister RRJ, Olsson P, Ryan P (2006) Toward a network perspective of the study of resilience in social-ecological systems. Ecol Soc 11:15

Kessler H (1967) Water balance investigations in the karstic regions of Hungary. Hungarian Research Institute, Paris, pp 91–105

Kollet SJ, Zlotnik VA (2003) Stream depletion predictions using pumping test data from a heterogeneous stream–aquifer system (a case study from the Great Plains, USA). J Hydrol 281(1–2):96–114

Kruseman GP, de Ridder NA (1994) Analysis and evaluation of pumping test data. ILRI Publication 47, Wageningen

Lawrence AR, Morris BL, Foster SSD (1998) Hazards induced by groundwater recharge under rapid urbanization. Geol Soc Lond Eng Geol Spec Publ 15:319–328

Lenouvel V, Montginoul M (2010) Groundwater management instruments in a conjunctive use system: assessing the impact on farmers' income using a Mixed Integer Linear Programming (MILP). Ger J Agric Econ 59:158–172

Llamas MR, Martinez-Santos P (2005) Intensive groundwater use: silent revolution and potential source of social conflicts. J Water Resour Plan Manag 131(5):337–341

Mailhol J-C, Merot A (2008) SPFC: a tool to improve water management and hay production in the Crau region. Irrig Sci 26:289–302

Mardhel V, Frantar P, Uhan J, Andjelov M (2004) Index of development and persistence of the river networks (IDPR) as a component of regional groundwater vulnerability assessment in Slovenia. In: Proceedings on the international conference on groundwater vulnerability assessment and mapping, Ustron, 15–18 June 2004

Marei A, Khayat S, Weise S, Ghannam S, Sbaih M, Geyer S (2010) Estimating groundwater recharge using the chloride mass-balance method in the West Bank, Palestine. Hydrol Sci J 55(5):780–791. http://dx.doi.org/10.1080/02626667.2010.491987

McFarlane D, Strawbridge M, Stone R, Paton A (2010) Managing groundwater levels in the face of uncertainty and change: a case study from Gnangara. Water Sci Technol 12(3):321–328

Mérot A, Bergez J-E, Capillon A, Wery J (2008) Analysing farming practices to develop a numerical, operational model of farmers' decision-making processes: an irrigated hay cropping system in France. Agr Syst 98:108–118

Mills D (2004) Advances in solar thermal electricity technology. Sol Energy 76:19–31

Montginoul M, Rinaudo J-D (2011) Controlling households' drilling fever in France: an economic modeling approach. Ecol Econ 71:140–150

Olsson P, Gunderson LH, Carpenter SR, Ryan P, Lebel L, Folke C, Holling CS (2006) Shooting the rapids: navigating transitions to adaptive governance of social-ecological systems. Ecol Soc 11:18

Pedretti D, Barahona-Palomo M, Bolster D, Fernàndez-Garcia D, Sanchez-Vila X, Tartakovsky DM (2012) Probabilistic analysis of maintenance and operation of artificial recharge ponds. Adv Water Resour 36:23–35

Petrone KC, Hughes JD, van Niel TG, Silberstein RP (2010) Streamflow decline in southwestern Australia, 1950–2008. Geophys Res Lett 37:L11401

Poulsen SE, Rasmussen KR, Christensen NB, Christensen S (2010) Evaluating the salinity distribution of a shallow coastal aquifer by vertical multielectrode profiling (Denmark). Hydrogeol J 18:161–171

Priyantha Ranjan S, Kazama S, Sawamoto M (2006) Effects of climate and land use changes on groundwater resources in coastal aquifers. J Environ Manage 80:25–35

Razès M, Barone S, Richard-Ferroudji A, Guerin-Schneider L (2013) Enquête sur les « élus de l'eau » des bassins Rhône-Méditerranée et Corse. Agence de l'eau Rhône Méditerranée et Corse

Rinaudo J-D, Montginoul M, Desprats JF (2015) The development of private bore-wells as independent water supplies: challenges for water utilities in France and Australia. In: Grafton Q, Chan N, Daniell K, Nauge C, Rinaudo J-D (eds) Understanding and managing urban water in transition. Springer, Dordrecht (in press)

Saayman IC, Adams S (2002) The use of garden boreholes in Cape Town, South Africa: lessons learnt from Perth, Western Australia. Phys Chem Earth A/B/C 27:961–967

Sahu P, Michael HA, Voss CI, Sikdar PK (2013) Impacts on groundwater recharge in areas of megacity pumping: analysis of potential contamination of Kolkata, India, water supply. Hydrol Sci J 58:1340–1360

Scanlon RB, Healy RW, Cook PG (2002) Choosing appropriate techniques for quantifying groundwater recharge. Hydrogeol J 10:18–39

Scibek J, Allen DM, Cannon AJ, Whitfield PH (2007) Groundwater–surface water interaction under scenarios of climate change using a high-resolution transient groundwater model. J Hydrol 333(2–4):165–181

Scott CA, Shah T (2004) Groundwater overdraft reduction through agricultural energy policy: insights from India and Mexico. Int J Water Resour Dev 20:149–164

Seneviratne SI, Lehner I, Gurtz J, Teuling AJ, Lang H, Moser U, Grebner D, Menzel L, Schroff K, Vitvar T, Zappa M (2012) Swiss prealpine Rietholzbach research catchment and lysimeter: 32 year time series and 2003 drought event. Water Resour Res 48(6): W06526

Shah T (2014) Towards a managed aquifer recharge strategy for Gujarat India: an economist's dialogue with hydro-geologists. J Hydrol 518:94–107. doi:10.1016/j.jhydrol.2013.12.022

Shah T, Bhatt S, Shah RK, Talati J (2008) Groundwater governance through electricity supply management: Assessing an innovative intervention in Gujarat, western India. Agric Water Manag 95:1233–1242

Shalev E, Lazar A, Wollman S, Kington S, Yechieli Y, Gvirtzman H (2009) Biased monitoring of fresh water-salt water mixing zone in coastal aquifers. Ground Water 47:49–56

Sommer B, Horwitz P (2009) Macroinvertebrate cycles of decline and recovery in Swan Coastal Plain (Western Australia) wetlands affected by drought-induced acidification. Hydrobiologia 624:191–203

Sophocleous M (2002) Interactions between groundwater and surface water: the state of the science. Hydrogeol J 10:52–67

Stehle S, Elsaesser D, Gregoire C, Imfeld G, Niehaus E, Passeport E, Payraudeau S, Schafer RB, Tournebize J, Schulz R (2011) Pesticide risk mitigation by vegetated treatment systems: a meta-analysis. J Environ Qual 40:1068–1080

Steuer JJ, Hunt RJ (2001) Use of a watershed-modeling approach to assess hydrologic effects of urbanization, North fork pheasant branch basin near Middleton, Wisconsin. U.S. Geological Survey Water-Resources Investigations Report 01–4113, p 49. http://pubs.er.usgs.gov/publication/wri014113

Terzić J, Šumanovac F, Buljana R (2007) An assessment of hydrogeological parameters on the karstic island of Dugi Otok, Croatia. J Hydrol 343(1–2):29–42

Thornthwaite CW (1948) An approach toward a rational classification of climate. Geogr Rev 38: 55–94

Tournebize J, Gramaglia C, Birmant F, Bouarfa S, Chaumont C, Vincent B (2012) Co-design of constructed wetlands to mitigate pesticide pollution in a drained catch-basin: a solution to improve groundwater quality. Irrig Drain 61:75–86

Tsur Y (1990) The stabilization role of groundwater when surface water supplies are uncertain: the implications for groundwater development. Water Resour Res 26:811–818

Tsur Y, Graham-Tomasi T (1991) The buffer value of groundwater with stochastic surface water supplies. J Environ Econ Manag 21:201–224

Vrba J, Verhagen B (2011) Groundwater for emergency situations. IHP – VII series on groundwater. UNESCO – International Hydrological Programme, Paris

Webb RH, Leake SA (2006) Ground-water surface-water interactions and long-term change in riverine riparian vegetation in the southwestern United States. J Hydrol 320:302–323

Werner AD, Bakker M, Post VEA, Vandenbohede A, Lu C, Ataie-Ashtiani B, Simmons CT, Barry DA (2013) Seawater intrusion processes, investigation and management: recent advances and future challenges. Adv Water Resour 51:3–26

Westenbroek SM, Doherty JE, Walker JF, Kelson VA, Hunt RJ, Cera TB (2012) Approaches in highly parameterized inversion: TSPROC, a general time-series processor to assist in model calibration and result summarization. U.S. Geological Survey Techniques and Methods Report, Book 7, Section C, Chapter 7, 73 p. http://pubs.usgs.gov/tm/tm7c7/

Wong T (2006) Water sensitive urban design – the journey thus far Australian. J Water Res 10: 213–222

Yesertener C (2008) Assessment of the declining groundwater levels in the Gnangara groundwater mound. Report for the Department of Water, Western Australia. Hydrogeological Record Series HG14

Groundwater Management Under Global Change: Sustaining Biodiversity, Energy and Food Supplies

4

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Abstract

This chapter grapples with the challenge of simultaneously sustaining biodiversity, energy and food supplies in conjunction with efforts to mitigate and adapt to climate change. Managing groundwater supplies sustainably is critical to that challenge, and the chapter assesses the positive synergies and perverse impacts for sustaining groundwater resources from both climate change mitigation and adaptation policies. The chapter finds that the pressures on groundwater resources will likely increase in the future, with the location, scale and magnitude of groundwater use shifting in response to other pressures. For example, changing energy policies are resulting in rapid deployment of thirsty technologies. Similarly, climate change adaption will increasingly rely on the water storage capacity of aquifers, yet many adaptation measures may also increase groundwater use. For better groundwater management under global change pressures we recommend a focus on complementary measures to: integrate information, deploy appropriate new technologies, apply market-based incentives and improve cross-sectoral governance. The key challenge for proponents of sustaining groundwater resources is to engage stakeholders and decision-makers outside the water sector in governance institutions.

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4.1 Introduction

Increased demand for freshwater wrought by an increasing population, wealth and consumption of thirstier products will be exacerbated by climate change. While the direct impacts of climate change on groundwater recharge is uncertain, it is certain that climate change mitigation and adaptation policies will change. In some cases, shifts in policy will exacerbate the challenges associated with groundwater use and management. This chapter extends the detailed technical and governance information on groundwater in the following chapters (see especially Part II) to consider the implications of these significant and urgent global changes for the management of groundwater, and to suggest approaches to sustaining biodiversity while maintaining energy and food supplies under a changing climate.

In the next section, the little-appreciated synergies between climate mitigation policies and groundwater resources are explored. Energy demand management measures have positive synergies in reducing consumption of water, but the impacts of new energy technologies on groundwater are mixed: some increase and others decrease water consumption, the location of water use will change, and governments are being challenged to adequately regulate the rapid uptake of these new industries. Carbon sequestration in the landscape will have neutral impacts at best, but is more likely to have negative impacts on groundwater resources. In particular, the beguiling political appeal of tree planting and soil carbon heightens the risk that perverse impacts on groundwater will be poorly managed. Similarly, groundwater plays a significant role in climate change adaptation for water supply, food production and biodiversity conservation, due in part to the longer-term processes of recharge and storage that buffers aquifers from the short-term climatic and surface hydrology variability. These roles require more active and sustainable management of aquifers than has been achieved to date around the world.

The final section of this chapter considers options for meeting the challenge of more effectively managing groundwater to offset negative impacts of these global changes. The magnitude and location of tensions between groundwater, food and energy vary considerably from country to country and aquifer to aquifer. The drivers of groundwater depletion and demand for use vary at the local, regional and global scales. Thus, analysis of future impacts and associated solutions is complex and a range of disciplines is needed to understand how to manage the inter-linkages between the numerous drivers of groundwater use, from technology assessment through to the international political economy. It is with this multi-disciplinary framing that we begin to step through issues and options for managing groundwater more sustainably in a growing world and under a changing climate.

4.2 Implications of Climate Change for Groundwater

4.2.1 Direct Impacts from Climate Change

Modified weather patterns resulting from global climate change will affect rates of groundwater recharge differently in different parts of the world as outlined in Chap. 5. Precipitation will likely change in intensity, duration and frequency. In many areas, groundwater recharge may increase, as a result of increased precipitation totals, from more frequent large floods, or as a result of melting of permafrost (IPCC 2007a). In other regions, reduced precipitation and higher evapotranspiration are likely to decrease aquifer recharge. A number of these counter-veiling factors may occur in the same region making the outcome uncertain. For example, in the Murray-Darling Basin in south eastern Australia, while surface water availability may decline, under a changing climate, the infrequent but large floods may significantly contribute to aquifer recharge (CSIRO 2008; Hirabayashi et al. 2013).

Changes in vegetation land cover affecting runoff and recharge will occur due to climatic change and will exacerbate human impacts such as deforestation. Shifting of traditional climate and vegetation zones will result in alterations in the species composition of forests, rising snow lines, and more frequent wildfires. The latter may impact flood frequency and intensity, erosion, and dam siltation. The resultant effects on groundwater recharge will in turn affect rates and volumes of groundwater discharge to springs, stream base-flow and the availability of groundwater for pumping (Bates et al. 2008). The challenge for groundwater managers is to develop strategies that account for uncertainty, in a manner that can provide satisfactory outcomes for water use under a range of climate conditions (WWDR 2012). Example strategies range from conservative allocation limits to the use of threshold or contingency policies that trigger alternative management arrangements according to water availability conditions, and augmentation of storage through managed aquifer recharge (Chaps. 17 and 18).

In addition to the need for robust management that accounts for uncertainty, questions arise as to how climate change mitigation policies may avoid unsustainable impacts on groundwater, or how they may even benefit the resource.

4.2.2 Climate Change Mitigation Policies

Climate change mitigation policies typically fall into three categories: demand side, supply side and sequestration or storage focused strategies (IPCC 2007b). Demand side policies aim to reduce energy consumption and thus emissions of greenhouse gasses. Supply side policies shift the generation of energy away from fossil fuels to low-carbon sources. Sequestration approaches encourage the use of natural storage of greenhouse gasses in the landscape. Reducing greenhouse gas concentrations in the atmosphere to achieve an oft-expressed desire to limit global warming below 2 °C will require all of these approaches (Rogelj et al. 2013), and they all have implications for groundwater storage inventories. However, the groundwater consumption and storage implications of different mitigation measures vary considerably. Wallis et al. (2014) reviewed the water use implications of 74 mitigation measures for Australia and found that positive synergies existed between conserving energy and conserving water in a variety of demand management interventions. However, they also found that neutral and negative outcomes for water consumption are evident for a range of emerging low-emission energy technologies, and similarly, that very negative consequences could be expected from carbon sequestration measures. These findings are elaborated on below, specifically in relation to groundwater.

4.2.2.1 New and Emerging Energy Technologies

The quest for low-emission energy sources is driving rapid policy change as regulations, carbon pricing and technological innovation combine to favour rapid deployment of more modern energy technologies. The focus on reducing greenhouse gas emissions has meant that the impacts on water resources have received very little attention. Booming industries, such as biofuels in the United States (US) and unconventional gas production globally, have developed in advance of efforts by government regulators to require application of better practices, including sustaining groundwater resources (Hussey and Pittock 2012). In Australia, new financial incentives for low-emission energy sources have been adopted without fully considering how well carbon, energy and water markets are harmonised to avoid externalities (Pittock et al. 2013). To inform this analysis a number of cases with risks to groundwater from expansion of emerging energy technologies are considered, including biofuels, (hot-rock) geothermal, unconventional gas, solar thermal and ground-source heating and cooling systems.

Biofuels

First generation biofuels use crops that are frequently irrigated from groundwater like corn, sugar cane and beet to produce ethanol and oil palm and soy to generate biodiesel. Water consumption to grow these feed stocks means that these alternative fuels have water footprints several orders of magnitude higher than most conventional and renewable energy systems (Gerbens-Leenes et al. 2008). Yet, there has been a rapid expansion of these industries driven by subsidies and renewable fuel quotas in jurisdictions including Australia, Brazil, the European Union and the US (Pittock 2011).

There are reports that up to 28 l of irrigation water are needed to produce enough soybeans to propel an average vehicle 1 km. In comparison, water needs for gasoline (petrol) are merely 0.33 l of water for each vehicle 1 km (King and Webber 2008). As is true for the agricultural sector generally, limiting the impacts on groundwater resource use by biofuels requires good governance, including allocation systems that cap extraction at sustainable levels and maximise social and economic benefits from the water consumed. However, the political power of biofuel industries in some countries may compel policies that encourage non-sustainable use and allocation (Notaras 2011). For example, the 2007 Energy Independence and Security Act in the US mandates an increase in annual biofuels production, requiring an additional 56.8 billion litres of ethanol by 2015 and an additional 60.6 billion litres of biofuels from cellulosic crops by 2022 (Dominguez-Faus et al. 2009). These mandated increases will likely increase the demand for groundwater resources, potentially pitting biofuel production against other irrigated agriculture, including food production. In the absence of appropriate governance arrangements to allocate water resources efficiently between uses, this increased competition could have deleterious effects on both the water supply base and commodity prices.

Simultaneously a number of transitions in less developed countries are beginning to revolve around biofuel related opportunities. Many producers are securing land and water resources in developing countries for production of crops, including for export of biofuels (Vermeulen and Cotula 2010; Zoomers 2010). In Africa, for example, agricultural proponents are pointing to little exploited groundwater resources as a major opportunity to expand production (MacDonald et al. 2012). To avoid the depletion of aquifers that has taken place in developed economies, groundwater governance will need to be strengthened in developing countries so as to manage these resources sustainably for both consumptive and non-consumptive purposes.

At the same time, there is a considerable global research effort into second generation biofuels from processing grass or timber cellulose (Sims et al. 2010) and third generation feedstock crops and techniques, which also raises interception questions for aquifer recharge. These ‘wonder’ crops, like jatropha, are untested. While these species may be able to grow on degraded lands and generate benefits for people in developing countries (Openshaw 2000), it is likely that widespread plantings would more effectively intercept precipitation and reduce aquifer recharge and surface runoff as land is cleared to establish the new crop (van Dijk and Keenan 2007). Proposals for third generation biofuels from farming microbes suggest that saline or wastewater may be used in these processes in the future (Yang et al. 2011), though commercial scale application has yet to be demonstrated. Each technological advance offers improvements in fuel production and may also meet other goals such as a reduction in GHG emissions, but biofuels are intrinsically linked with groundwater resources and can compete directly with agricultural food crops for water and land.

In essence, current commercial biofuel production consumes significant water, for crop production, processing and transport, and if production is increased then pressures to exploit aquifers globally will also increase. Biomass for fuel production where irrigation and crop chemicals are also used results in greater risks of aquifer contamination and hence a potential reduction of economically-usable groundwater. Given the complex and often uncertain knock-on consequences of biofuels, policy interventions which aim to increase biofuel production must account for these risks.

Geothermal

The generation of electricity from steam from underground aquifers where circulating groundwater is “boiled” by geological heat sources is a commercial energy technology and is sustainable in regions with substantial aquifer recharge, such as in Iceland and New Zealand. Geothermal energy proponents are now exploring ways of generating electricity from 'hot rock' sources, where aquifers are small or absent, by injecting water in one borehole to be heated through fractured strata, then extracted as steam up a parallel borehole to generate electricity. Geothermal generation may be sustainable in regions where there is plentiful water but in dry areas the source of water is uncertain. For example, much of the geothermal 'hot rock' resource in Australia is located in arid areas or in the wet-dry tropics where surface water resources are seasonal or absent (Goldstein et al. 2009).

Linking strata through boreholes and by fracking also raises the same questions (as for unconventional gas production) of managing potential risks of natural contaminants becoming incorporated in the production water and moving into previously constrained aquifers through fractures or borehole failures.

Unconventional Gas

Rising costs of petroleum on international markets, the political drive to achieve greater energy independence, and the development of directional drilling and hydraulic fracturing techniques have significantly improved the economics of natural gas as an energy source. Compared to conventional, free-flowing natural gas extraction, unconventional gas development involves production of methane from multiple types of geological strata where the deposits are dewatered and/or fractured (fracked) to enable withdrawal. This discussion will focus on the two most widespread resources, those in coal seams and those in shale (Cook et al. 2013).

Natural gas is a fossil fuel and governments around the world facilitate its exploitation for reasons of domestic energy security and to reduce greenhouse gas emissions. Scientists disagree on the extent to which unconventional gas production reduces greenhouse gas emissions owing to the risk of fugitive methane leaking from poorly maintained valves and connections in the surface storage and pipe-line infrastructure (Burnham et al. 2011). Nevertheless, in the best case scenario natural gas may reduce greenhouse gas emissions by around half compared to coal-fired generators (Burnham et al. 2011), thus receiving favourable treatment under carbon pricing schemes.

Coal seam, or coal bed, methane deposits are usually closer to the surface and production requires dewatering strata, resulting in the production of lower quality water. Shales with gas potential generally lie deeper in the earth, and gas development and most production methods currently used require the injection of large volumes of water. The directional drilling process and the subsequent hydraulic fracture of the shale target area involve the addition of various chemicals, compounds and proppants which are pumped under pressure to liberate natural gas from the rock formations. Contaminated flow-back water from hydraulic fracturing and 'produce water' (from the geological formations) over the lifetime of the gas well requires careful attention with respect to storage, treatment and disposal so as to avoid contamination risks to both surface and groundwater resources.

Common concerns for aquifer management for coal seam, coal bed, and shale gas production identified by representatives from industry, researchers and regulators (Williams and Pittock 2012; Mauter et al. 2014), include potential for the creation of pathways for contaminant migration both at depth and from surface infrastructure, toxicity information for fracking chemicals, and to a lesser extent risks from induced seismicity. Fracking chemicals are used to develop and maintain boreholes and prop open the cracks in the strata to allow the gas to flow out. The toxicity of these chemicals is disputed, however many companies involved in the industry are supporting public disclosure laws and practices to demonstrate their confidence that the fluids will cause no harm. There are concerns that fracking may connect different rock strata and enable contaminated water and methane to migrate up into overlying freshwater aquifers, or even to the surface. The industry disputes this concern, saying that fracking is able to be limited to the target, gas producing coal seam or shale strata. However, industry and other stakeholder groups agree that inadequate borehole construction may enable methane and contaminated water to migrate into higher freshwater aquifer and to the surface.

There is a wealth of anecdotal accounts in the news media about the negative environmental impacts of shale-gas development. However, a common concern expressed by many groundwater specialists about gas production, is the lack of hard data and information in relation to migratory pathways. Knowledge and characterization about potential flow paths in the zone between the deep shale targets (usually 2–3 km beneath the surface) and the freshwater aquifer zones that may occur at depths up to 1 km is limited (Council of Canadian Academies 2014). At the same time, risks from gas related contamination appear to be low, to date very few instances of possible methane migration are documented in the US. Well blowouts (casing failure) are rare because industry standard operating practices require a test of vertical well casing integrity before proceeding with any hydraulic fracturing. Added to this is increased risk of earthquakes induced by the injection of fluids, which in turn compounds the risk of that injected fluid leaking into other aquifers, either during the production of gas or at some later date. However, while research undertaken in the US indicates that injection-via-disposal wells may cause tremors (National Research Council 2013), there is very little evidence hitherto of fault or fracture propagation resulting from hydraulic fracturing.

Industry and many researchers consider that the greatest risk to water resources from gas production is leaks from production water containment ponds and other spills on the surface, including accidents with fluid transport trucks on rural roads (Mauter et al. 2014; Williams and Pittock 2012). Once production water is at the surface it requires treatment, re-use or disposal. In the US, the reinjection of production waters into saline zones in deep geological formations is common practice but not all gas producing areas have the geologic conditions for disposal by injection, and there is increased environmental risk involved in transport to suitable areas. This raises questions as to the risk of polluting potentially beneficial aquifers in other locations. The practice of using closed or evaporative basins to treat production water, especially saline water, was abandoned in Texas as erosion often resulted in the breakdown of containment structures.

This analysis exposes a number of risks to aquifers from unconventional gas production that each has a technical solution, but only if the industry is consistently well governed and adheres to the highest standards of practice. As a result of public and political concerns, and because of the economic costs related to water use and disposal, the US oil and gas industry is currently researching and field-testing many different on-site water treatment technologies. In addition, technologies that reuse water or actually use zero water for the hydraulic fracturing process are in development. However, until there is a rise in the market value of gas, many of the promising technologies are unlikely to achieve widespread implementation.

One concern that has not yet been well addressed in the development of the unconventional gas industry is the future of groundwater in depleted and abandoned gas fields. Aquifer depletion can be expected over long periods of time if associated with gas deposits, or fractured strata newly capable of holding water will recharge. What is unclear is how this will affect other water resources on basin scales, for example whether other surface and groundwater deposits may be depleted if they begin to fill the new, often deeper voids that are left behind.

Solar Thermal

Solar thermal power is an emerging technology that uses mirrors in large scale facilities to boil water and generate steam for electricity production. Currently deployed in California and Spain, these power stations work best when located in sunny, arid and semi-arid regions where water is naturally scarce. While the volumes of water required are modest compared with many other forms of energy technologies, sustainable groundwater availability may be a limiting factor for the location of these stations in deserts.

The world's largest solar thermal plant in the Mojave Desert near the border of California and Nevada is the 392-MW Ivanpah project. At the official opening in 2014, the US Energy secretary stated that the station's water needs for steam production "...will use roughly the same amount of water as two holes at the nearby golf course" (Phillips 2014). An additional water demand from the desert aquifers will be to regularly wash dust from the project's 347,000 mirrors.

As with all thermal power stations, there is the option of deploying dry rather than wet cooling technology. Dry cooling systems use less than 10 % of the water of a wet cooling system but have several drawbacks, including a higher, upfront capital cost; reduction in energy generation of around 8 %; and less effective operation with higher air temperatures, such as the arid areas where these power stations are located (DoE 2008).

Ivanpah uses a directly heated steam cycle that can only generate power when the sun shines. In the future, large-scale solar plants will likely use an energy storage technology (such as the process that heats molten salt) so that energy can be stored and then 'released' whenever there is a load demand (Phillips 2014). Globally, large schemes have been proposed to power countries like Australia (BZE 2010) or whole regions such as northern Africa and Europe based on solar thermal power stations, though the economies of such ventures has yet to prove favourable.

Production of hydrogen for use as a renewable fuel in fuel cells, from the electrolysis of water using solar generated electricity, is another possibility. If this hydrogen is combined with atmospheric nitrogen at high temperatures (which is possible in a solar thermal power station) to produce ammonia (NH_3) as a renewable energy fuel, it could regenerate the water, but some loss of water might be expected (Andrews and Shabani 2012; Balat 2008).

Aquifer Thermal Energy Systems

Aquifer thermal energy storage systems (ATES) are common in Europe and typically operate by running groundwater through a cooling tower in winter and returning it to the aquifer for storage. In summer, the chilled water is withdrawn, used for air conditioning and put back into the aquifer as warm water for use in winter to reduce heating costs. If closed loops are used to transfer heat the loop pipes are typically filled with food-grade glycol so that in the unlikely event of a leak, there is minimal risk to groundwater quality. Now, there is a growing trend in the US for using ground source heating and cooling technology for individual homes, schools, churches and office buildings. There are already over one million such installations in operation in the US. Ball State University in Muncie, Illinois has installed a ground source system involving 3,600 boreholes to service 622,450 m^2 of building space which will save the burning of 36,000 t of coal that was previously used each year (Roulo 2011).

When applied on a large scale for college campuses, military installations etc. this technology is providing a developing field for hydrogeologists to characterize subsurface heat transfer capabilities and to assess potential impacts on aquifers, particularly if the heat dissipation is dependent on groundwater flow. A concern is the potential build-up of groundwater temperatures which could progressively decrease heat transfer efficiency.

ATES technology and ground source heating and cooling raise a number of issues for future groundwater management. As with other technologies, their rapid increase in popularity since the 1990s has seen deployment in advance of adequate regulatory oversight (Bonte et al. 2011). Both systems can interfere with other underground infrastructure for electricity, water distribution and telecommunications technologies. The technology also raises questions of who owns the underground lands and waters and under what circumstances they can be exploited. The open systems risk diminishing biological and chemical water quality of aquifers through moving water about, and heating and cooling. The closed systems raise questions as to standards for containing the chemicals used and responsibilities for leaks and decommissioning.

Fossil Substitution

As the above examples illustrate, new energy technologies offer opportunities to reduce greenhouse gas emissions but with some risks for groundwater resources. A number of the proponents of these newer technologies argue that they can be substitutes for water-intensive fossil fuel-fired power stations and thus may free up water for other uses. For example, Beyond Zero Emissions argues that its proposal for a solar thermal power station in Port Augusta, Australia can be watered by decommissioning the local coal-fired power station (BZE 2010). Certainly in regions with high concentration of coal-fired power stations this may free up water, for example, in the Latrobe and Hunter valleys in Australia. However, this may also shift water consumption from places where water use is well-regulated to places where governance is poorer, for instance, from the two Australian coastal valleys to arid locations in the interior, where each litre of water may have more environmental and socio-economic value to other users. If governments and societies want this sort of water substitution to occur, then it will require active facilitation and regulation.

4.2.2.2 Risks to Groundwater from Carbon Sequestration in the Landscape

Carbon sequestration in the landscape, a subset of geoengineering proposals, is another component of mitigation policies that may impact on groundwater management and use. Two approaches to store greenhouse gases in the landscape are discussed here: geological carbon capture and storage, and carbon farming, including plantations.

Carbon dioxide (CO_2) capture and sequestration (CCS) is a process that involves underground injection and geologic storage (sequestration) of CO_2 in deep underground rock formations that are overlain by impermeable rock that trap the CO_2 and prevent it from migrating upward. CCS can significantly reduce emissions from industrial sources such as fossil fuel-fired power plants (EPA 2013). The US Department of Energy estimates that between 1,800 and 20,000 billion metric tons of CO_2 could be stored underground in the US (c, 2012), a volume that is equivalent to 600–6,700 years of current level emissions from large stationary sources in the US (GHGRP 2012). Moreover, while sequestration removes CO_2, that might otherwise impact the atmosphere, according to the US EPA Greenhouse Gas Reporting Program, CO_2 capture for industrial reuse is currently occurring at over 120 facilities in the US. End users of CO_2 include enhanced oil recovery, food and beverage manufacturing, pulp and paper manufacturing, and metal fabrication.

The success of CCS requires very low rates of leakage. The widespread drilling of gas wells has been cited as a risk to the security of potential CCS sites (Elliot and Celia 2012) and widespread bore-holes used previously in searches for oil and other minerals may also cause leakages. Thousands of such bore-holes were drilled in the early twentieth century, and their precise locations and seals are often unknown. In terms of groundwater, the primary concern is whether placement of waste gases underground will result in reductions of groundwater quality.

In contrast with CCS, sequestration of carbon in land and vegetation is practised internationally. In some nations, it is used either to earn or sell carbon credits in a formal market or in schemes to offset emissions in other sectors. As an example, many airlines now offer passengers the option of paying extra to offset the emissions from their flights through tree planting.

Planting trees to sequester carbon is the most common method advanced because of its many co-benefits, in terms of such services as biodiversity and soil conservation, production of non-timber forest products, and aesthetic improvements to the landscape. However, forests will normally intercept more precipitation than non-forested land uses, diminishing surface runoff into streams and aquifer recharge (van Dijk and Keenan 2007; Jackson et al. 2005). This inflow interception may not have significant impacts in wet environments such as in the wet tropics, but in the temperate zone significant reductions in flows are likely. In past decades in Australia, tree planting has been actively encouraged to reduce groundwater recharge in areas subject to salinity. Several means of reducing these impacts on water resources are possible, including: incorporating the plantation sector into cap and trade water markets, as occurs in South Australia and South Africa; limiting afforestation to landscapes where the impacts may be acceptable, such as the wet tropics and salinity prone lands; or scheduling planting over decades so that the impacts are spread over a longer period of time (Pittock et al. 2013).

A number of other methods are being actively promoted to sequester more carbon in soils, although there is little evidence of widespread application thus far. Incorporating more biomass into soils is promoted as a way of enhancing agricultural productivity by improving soil structure, fertility and water infiltration, as well as sequestering carbon (Henriksen et al. 2011). Biochar – adding charcoal to soils – has a very active group of promoters (Kleiner 2009; Sohi et al. 2009). A lot of research investment has focussed at the field scale on the longevity of the carbon sequestration with often disappointing results (Lam et al. 2013). A common claim is that by developing more friable soils that these methods will enable more precipitation to be stored in the soil and advantage crop growth. If this proves to be the case one potential outcome is diminished surface runoff and aquifer recharge.

Internationally, carbon sequestration in the landscape has a mandate under the umbrella of 'land use change and forestry' and it is being deployed through two programs of the UN Framework Convention on Climate Change. The Clean Development Mechanism and proposed REDD+ scheme (Reduced Emissions from Degradation and Deforestation plus) enable projects applying approved methodologies for reducing emissions or sequestering carbon in land and vegetation in developing countries to generate carbon credits (CDM Executive Board 2010; Pritchard 2009). However, the Clean Development Mechanism's current procedures for assessing and considering any negative impacts of proposed projects on water resources are token (Pittock 2010).

Australia is one nation that has legislated in the Carbon Credits (Carbon Farming Initiative) Act 2012 for market-based carbon sequestration in the landscape, based on the Clean Development Mechanism's approach of approved methodologies (Australian Government 2011). The Act's regulations attempt to limit the impact of carbon plantations on water by prohibiting commercial timber production and planting in areas within the 600 mm/year and above rainfall isohyet, subject to a number of exemptions (DCCEE 2011). The 600 mm/year rainfall isohyet was chosen as a threshold above which surface water runoff may be expected, however this may unreasonably restrict planting in environments where impacts may be insignificant, as in the tropics. The exemptions include planting for biodiversity conservation, and those agreed by poorly-resourced, state government mandated natural resource management organisations. National policy agreements to include significant inflow interception activities (including groundwater recharge) within cap and trade water markets have only been implemented by one of the eight states and territories (NWC 2011). Consequently this odd collection of half implemented policies and the exemptions mean that there is a strong prospect of perverse impacts on groundwater recharge.

Many other nations have prioritised reforestation in their climate mitigation policies, including China, India and Mexico, indicating that managing the trade-offs between planting for carbon sequestration and water use is a growing global challenge (Pittock 2011). The links between the projected impacts of climate change and the sustainable management of surface and groundwater resources makes the challenge all the more complex. For example, with so many countries pursuing carbon sequestration through tree plantings, and the Intergovernmental Panel on Climate Change's projections for increased wildfire frequency and intensity, it is not inconceivable that governments may be increasing the risks of even bigger and more devastating wildfires by pursuing policies that are, ironically, attempting to mitigate the impacts of climate change. And, of course, the knock-on consequences of more frequent and intense wildfires are insidious: denuded catchments which in turn lead to more floods, erosion and siltation of water storages, which has important implications for the sustainable use of groundwater resources.

4.2.3 Climate Change Adaptation Policies

Having discussed the implications of climate change mitigation on groundwater resources, we now turn to consider how groundwater may be used and sustained through climate change adaptation measures. Climate change is likely to impact surface water supplies in particular places in a number of ways, including: increasing or decreasing precipitation; changing seasonality of snowmelt and river flows; increasing evapotranspiration, the intensity of storms and frequency of floods and droughts. Groundwater resources have the potential to complement or buffer surface water shortages to deliver key services (Bates et al. 2008). Three examples are now elaborated, namely urban water supply, food production and freshwater biodiversity conservation.

4.2.3.1 Water Supply

Sustaining a reliable supply of drinking water to urban areas is essential for the well-being of the majority of the planet's people. Not only does good health depend on clean drinking water, but so too does the economic health of these communities. Climate change impacts, increasingly, jeopardise cities that depend on surface water catchments. Australia provides a salutary example. In the mid-1970s inflows into the city of Perth's water storages began a series of 'step changes' such that a decline in the order of 70 % of the previous long-term average was experienced (Petrone et al. 2010). During the 2002–2010 Millennium Drought another five cities in southern Australia also saw their water storages reduced to perilously low levels. A common response of the impacted states was to diversify the supplies of water for these cities by adding reuse, groundwater, and desalination sources. In particular, Adelaide, Perth and Sydney each drew on new groundwater resources, applied managed aquifer recharge, or set aside aquifers as drought reserves.

This Australian example highlights the potential of aquifers to grow in importance as existing urban water storage and sources become more sensitive to increasingly variable climatic and surface hydrological conditions. This capacity can be enhanced through managed aquifer recharge, as detailed in Chaps. 17 and 18. These same storage characteristics will also make aquifers more attractive as a source of water for food production.

Additionally, an important buffering role of groundwater can be provided by individual on-site water wells. Private wells can reduce demand pressures on larger aquifers. In the US over 40 million people are supplied with their water needs from 15 million private wells (US Census Bureau 2007). In most instances homeowner wells (often in bedrock fractures) are accessing small discrete aquifer systems that are economically unusable for any major supply. Provided there is limited outside lawn watering, virtually all the pumped water is treated and returned to the sub-surface via septic systems and leach-fields. The key to continuing this harmonious use of groundwater is to ensure through zoning regulations that well density does not exceed renewability and that the rights of private well owners sharing access to aquifers with major pumpers are protected. “Deepest well wins” is not a good basis for groundwater management.

4.2.3.2 Irrigated Food Production

In 2007, the International Water Management Institute (IWMI)’s “Comprehensive assessment of water management in agriculture” (CAoWMiA) reviewed the world’s future food needs and explored scenarios for how the required water may be sourced (CAoWMiA 2007). Around half of the globally accessible freshwater is already diverted for human uses and 70 % of the world’s water consumption is in agricultural production. CAoWMiA (2007) reported that food demand will double over the next 50–80 years, and that without improvements in productivity, water use in food production will need to increase by 70–90 % under a changing climate (CAoWMiA 2007). From a business perspective, a McKinsey & Company global report estimates “that the annual pace at which supply is added over the next 20 years in water and land would have to increase by 140 % and up to 250 %, respectively, compared with the rate at which supply expanded over the past two decades. This expansion of supply could have a wide range of potentially negative effects on the environment. In this case, there would be an additional 1,850 km^3 of water consumption by 2030, 30 % higher than today’s levels ...” (Dobbs et al. 2011: 8).

A study by Wada et al. (2012) shows that on a global basis non-renewable groundwater abstraction represents 18 % of global gross irrigation water demand. In other words, on a global basis we are draining aquifer systems (see also Chap. 2 for more detail on aquifer depletion). This loss of groundwater inventory has greatly reduced the capacity of aquifers to serve as a buffer against current or future drought.

In the US over the last 100 years over 1,000 km^3 of groundwater has been removed from major aquifers with the greatest losses from the High Plains Aquifer (350 km^3) and California's Central Valley (150 km^3) (Konikow 2013). These trends in groundwater depletions in the US have been observed and known for many years. However, effective and sustainable management strategies have eluded policy makers and only now, because of severe drought conditions, are end users and legislators in California, Texas and other impacted states beginning to talk about water metering and devising workable criteria for prioritizing allocations of the progressively scarce groundwater resources. These discussions are clouded by the issue of "water rights" and the spectre of litigation from end-users whose pumping might be curtailed.

The Asian Development Bank raises similar concerns. Noting "total annual sustainable freshwater supply remaining static at 4,200 billion cubic meters (m^3), the annual deficit for 2030 is forecasted to be 2,765 billion m^3, or 40 % of unconstrained demand, assuming that present trends continue. India and China are forecasted to have a combined shortfall of 1,000 billion m^3 – reflecting shortfalls of 50 % and 25 %, respectively. There is little evidence of changing trends. Signals of scarcity and stress have had little impact on policies, demand, or the market. On the supply side, there is little room for finding and abstracting more water. In areas with physical water scarcity (including north [China], south and northwest India, and Pakistan), demand needs to lessen" (ADB 2013: vi).

The increasingly frequent droughts predicted with climate change means that the greater security of food production afforded by irrigation will become increasingly popular. In Africa, for example, national governments have extensive plans to expand irrigated production (Sullivan and Pittock 2014). There has been extensive debate about why irrigated agriculture has performed very poorly in Africa, which points to a combination of problems with infrastructure, human capacity and economic viability (Lankford 2009). A number of researchers have pointed to extensive, but little used, groundwater resources in Africa as the basis for increased agricultural production (MacDonald et al. 2012). The arguments for greater use of groundwater are many, but the most compelling are the increased cost efficiencies and drought resilience gained over traditional small-scale rainwater harvesting, and the capacity for groundwater resources to be developed to support more people across the landscape compared to centralised, surface irrigation schemes (Stirzaker and Pittock 2014).

The obvious question about greater reliance in Africa on groundwater for agriculture is how to avoid the over-exploitation that has afflicted many parts of the world. The management of consumption using cap and trade groundwater markets as practised in Australia is unlikely to work in most of Africa where the reach of the state is not as strong. Work by the International Water Management Institute in regions of over-exploited groundwater in India indicates two examples of unconventional approaches that may be addressing the problem of over-exploitation of groundwater due to subsidized electricity for pumping. Reducing these power subsidies has not been politically feasible but other solutions have emerged. Over the past decade in Gujarat, India a USD $260 million scheme called Jyotigram Yojana ("Lighted Village") has sought to overcome electricity theft and blackouts while rationing groundwater and ensuring the financial viability of utilities (IWMI 2011). Installation of a dual electricity distribution system has enabled one distribution system to be dedicated to providing reliable supplies to villages while the other system provides power for 8 h/day to groundwater pumps. This approach has curtailed energy consumption, encouraged more efficient groundwater pumping, and facilitated a tripling of agricultural production.

More recently the state government of West Bengal scrapped a permit system, instead connecting small pumps to the power grid at a fixed cost that only enables farmers to access annual monsoon recharge from shallow aquifers, conserving deeper groundwater resources. IWMI estimate that the area irrigated will expand in 3–5 years from 2.98 to 4.83 million hectares, increasing annual paddy rice production by 4.62 million tonnes (IWMI 2012).

4.2.3.3 Freshwater Biodiversity Conservation

Freshwater biodiversity has been significantly impacted by overexploitation of surface and groundwaters (MEA 2005; see also Chaps. 14 and 15). Current approaches to conserving freshwater biodiversity, including for climate change adaptation, have focussed on providing surface environmental flows and in some countries, environmental water demand management (also called environmental works and measures in Australia) (Poff and Matthews 2013; Pittock and Lankford 2010; Richter 2010). In countries like Australia, environmental flow programs have focussed on conserving large wetland systems, often in the lower reaches of river systems (Pittock and Finlayson 2011). An assumption is that surface water environment flows under conditions of short-term variability, and long-term climate change, will be sufficient to sustain the ecological character of these wetlands. Yet evidence is that desiccation and water quality impacts of drought events, exacerbated by climate change, are not adequately ameliorated by the current environmental watering programs (Pittock 2013; Pittock et al. 2010). In particular, these strategies assume that large wetlands in downstream reaches of river basins and ecosystems can be maintained in a similar state to the present.

Contrary to this approach, there is an emerging focus on the importance of conserving groundwater flows as a key strategy for retaining freshwater biota in refugia during severe drought and climate change (Pittock and Finlayson 2011). The potential exists for groundwater inflows into river channels to maintain reaches with sufficient volumes of water of acceptable quality to sustain biota that may otherwise perish. There are numerous management challenges if this adaptation option is to succeed, not least gaining community support to conserve connected aquifers for this purpose (Lukasiewicz et al. 2013). Importantly, these refugia are often different to the freshwater habitats currently prioritised for conservation. For instance, in Australia's Murray-Darling Basin, gaining reaches are often located in the mid and upper river systems rather than the downstream wetlands currently favoured (CSIRO 2008; Pittock and Finlayson 2011).

This example of changing groundwater management priorities highlights the governance challenges brought on by global change.

4.3 Discussion and Conclusion

The need for Integrated Groundwater Management (IGM) is set out in the first chapter of this volume and defined as: "a structured process which promotes the coordinated management of groundwater and related resources (including conjunctive management with surface water), taking into account non-groundwater policy interactions, in order to achieve shared economic, social welfare and ecosystem outcomes."

Groundwater governance arrangements available to policy-makers vary from the local to global scales (see Part II which is devoted to governance issues). International scale processes, such as climate change, may have major impacts on groundwater at the national scale. Similarly policy decisions at the national scale on natural resources management, such as on the extent of forests, will impact on aquifers. Groundwater systems are usually sub-national in scale such that sound national policy will only be effective if it supports sustainable management at the regional or local levels. Implementation of effective policies will require fostering of human capacity and institutions at appropriate levels, international to local scale. The earlier discussion also highlights the importance of integrating interventions across sectors. For example, managing groundwater sustainably may require intervention in the food sector more than the water sector. What then are some of the key mechanisms that may facilitate sustainable groundwater management? Is there a case for IGM, to complement Integrated Water Resources Management (IWRM; and its various iterations)?

As this chapter has elucidated, sustainable management of aquifers across competing water-use sectors requires positive synergies to be seized and perverse impacts to be identified and minimised. IGM under global change requires four key interventions (Pittock et al. 2013; Hussey and Pittock 2012; Pittock et al. 2015):

1. Information. The often unseen nature of groundwater and the lack of a common currency with competing natural resource uses can lead to decisions with deleterious impacts on aquifers. We contend that making publicly available, and generating where necessary, compatible information on groundwater resources and major uses like the environment, energy, food and domestic water can facilitate integrated decision making. Examples of such information transparency include: publicly available water accounts, such as those of the Australian Bureau of Meteorology (BoM and ABS 2011); the Australian Government's online atlas of matters of national environmental significance that includes listed groundwater dependent biota (DOE n.d.); simple, online decision making models, such as one in Texas that enables businesses and regulators to match water resources to proposed power generators (Webber

Energy Group n.d.); and 'traffic light' status reports on the state of aquifers and other resources (Pittock et al. 2013).

2. Technology. There are many technologies that may use less groundwater while facilitating climate change mitigation and adaptation, such as dry cooling thermal power stations (NETL 2008) and more efficient irrigation equipment (Mushtaq et al. 2009).
3. Market incentives. Establishing cap and trade water markets can create powerful incentives for using groundwater more efficiently and sustainably, as is now practised in many parts of Australia (Grafton et al. 2011). However, given the lower price of water per volume compared to many other natural resources and the potential for externalities, it is essential that markets for natural resources such as water, timber and carbon are harmonised to prevent negative impacts on groundwater (Pittock et al. 2013).
4. Reforming governance. Systematically integrating decisions across sectors like water and climate policy will expose many of the perverse outcomes identified in this chapter, though such integration is difficult to achieve. Pittock (2011) argues that there are five attributes of integrated governance, namely: (i) leadership; (ii) legal mandates for agencies to work across sectors in the interests of sustainability, for example, for electricity utilities to use fees to conserve water; (iii) mechanisms for vertical integration for local to national and international institutions, such as Australia's National Water Initiative (Commonwealth of Australia et al. 2004); (iv) horizontal integration between sectoral agencies, such as inter-departmental committees; and (v) accountability mechanisms such as periodic reviews, auditors, and capacity for third parties to challenge unsustainable decisions in the courts. As the examples discussed above with underground thermal energy systems and unconventional gas highlight, such integration is particularly required when new technologies emerge, to establish frameworks to govern their deployment.

Combined, actions in these four areas will go a long way to managing groundwater resources sustainably. However, the complexity of sustainable groundwater management raises the obvious question of whether an overarching conceptual framework is needed, as was deemed the case nearly 30 years ago when IWRM emerged. Indeed, espousing as it does "the coordinated 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" (GWP 2000), IWRM does in principle at least incorporate groundwater resources. In practice, though, the emphasis of IWRM has been on surface water resources, with scant attention afforded to groundwater – a fact which is borne out by the excellent chapters in this book. However, advocates of an IGM framework should be aware of IWRM's limitations. While there is evidence of broad acceptance of IWRM principles, success has been limited. Three particular deficiencies will likely be relevant in any attempt at IGM. First, the acceptance of IWRM has not changed the underlying power differences between stakeholders that make integrated management, and more sustainable outcomes, so difficult to achieve. Second, as an all-encompassing framework IWRM is intellectually robust but practically very difficult to implement. Finally, conceptual frameworks do not address the underlying governance and institutional capacity challenges that beset many developing countries, and which are, arguably, the major barrier to more sustainable practices. It is salient that many proponents of IWRM have been calling for a new approach for the last decade (Biswas 2004).

There is value in an overarching framework to manage groundwater resources, but perhaps more importantly there is a need for the advocates of IGM to engage stakeholders 'out of the *water* box', with a view to advocating the four interventions listed above. Global changes are increasing the pressures on groundwater resources, but with these difficult problems and crises come policy reform windows. The challenge for decision-makers and water managers is to be prepared to seize the opportunities to implement more sustainable groundwater management.

References

ADB (2013) Thinking about water differently: managing the water–food–energy nexus. Asian Development Bank, Manilla

Andrews J, Shabani B (2012) Re-envisioning the role of hydrogen in a sustainable energy economy. Int J Hydrogen Energy 37(2):1184–1203. doi:10.1016/j.ijhydene.2011.09.137

Australian Government (2011) Carbon Credits (Carbon Farming Initiative) Act 2011, No. 101, An Act about projects to remove carbon dioxide from the atmosphere and projects to avoid emissions of greenhouse gases, and for other purposes. Commonwealth of Australia, Canberra

Balat M (2008) Potential importance of hydrogen as a future solution to environmental and transportation problems. Int J Hydrogen Energy 33(15):4013–4029. doi:10.1016/j.ijhydene.2008.05.047

Bates BC, Kundzewicz ZW, Wu S, Palutikof JP (eds) (2008) Climate change and water, Technical paper of the Intergovernmental Panel on Climate Change. IPCC Secretariat, Geneva

Biswas AK (2004) Integrated water resources management: a reassessment. Water Int 29(2): 248–256. doi:10.1080/02508060408691775

BoM, ABS (2011) Australian Government water accounting, Activities of the Bureau of Meteorology and the Australian Bureau of Statistics. Bureau of Meteorology and Australian Bureau of Statistics, Melbourne/Canberra

Bonte M, Stuyfzand PJ, Hulsmann A, Van Beelen P (2011) Underground thermal energy storage: environmental risks and policy developments in the Netherlands and European Union. Ecol Soc 16(1):22. http://www.ecologyandsociety.org/vol16/iss1/art22/

Burnham A, Han J, Clark CE, Wang M, Dunn JB, Palou-Rivera I (2011) Life-cycle greenhouse gas emissions of shale gas, natural gas, coal, and petroleum. Environ Sci Technol 46(2):619–627. doi:10.1021/es201942m

BZE (2010) Zero carbon Australia stationary energy plan. Beyond Zero Emissions, Melbourne

CAoWMiA (2007) Water for food, water for life. A comprehensive assessment of water management in agriculture. Earthscan and International Water Management Institute, London/Colombo

CDM Executive Board (2010) Project design document form for afforestation and reforestation project activities (CDM-AR-PDD), version 05. UN Framework Convention on Climate Change, Bonn

Commonwealth of Australia, Government of New South Wales, Government of Victoria, Government of Queensland, Government of South Australia, Government of the Australian Capital Territory, Government of the Northern Territory (2004) Intergovernmental agreement on a national water initiative. Council of Australian Governments, Canberra

Cook P, Beck V, Brereton D, Clark R, Fisher B, Kentish S, Toomey J, Williams J (2013) Engineering energy: unconventional gas production. Australian Council of Learned Academies, Melbourne

Council of Canadian Academies (2014) Environmental impacts of shale gas extraction in Canada. Council of Canadian Academies, Ottawa

CSIRO (2008) Water availability in the Murray-Darling Basin, A report from CSIRO to the Australian Government. CSIRO, Canberra

DCCEE (2011) Carbon credits (carbon farming initiative) regulations 2011. Department of Climate Change and Energy Efficiency, Canberra

Dobbs R, Oppenheim J, Thompson F, Brinkman M, Zornes M (2011) Resource revolution: meeting the world's energy, materials, food, and water needs. McKinsey & Company, Chicago

DoE (2008) Concentrating solar power commercial application study: reducing water consumption of concentrating solar power electricity generation, Report to congress. US Department of Energy, Golden

DOE (n.d.) Protected matters search tool. Department of the Environment, Canberra

Dominguez-Faus R, Powers SE, Burken JG, Alvarez PJ (2009) The water footprint of biofuels: a drink or drive issue? Environ Sci Technol 43(9):3005–3010. doi:10.1021/es802162x

Elliot TR, Celia MA (2012) Potential restrictions for CO_2 sequestration sites due to shale and tight gas production. Environ Sci Technol 46(7):4223–4227. doi:10.1021/es2040015

EPA (2013) Carbon dioxide capture and sequestration. United States Environmental Protection Agency, Washington, DC

Gerbens-Leenes PW, Hoekstra AY, van der Meer TH (2008) Water footprint of bio-energy and other primary energy carriers, The value of water research report series no 29. UNESCO-IHE Institute for Water Education, Delft

Goldstein B, Hill A, Long A, Budd A, Holgate F, Malavazos M (2009) Hot rock geothermal energy plays in Australia. In: Proceedings of the 34th workshop on geothermal reservoir engineering, Stanford University, Stanford

Grafton RQ, Libecap G, McGlennon S, Landry C, O'Brien B (2011) An integrated assessment of water markets: a cross-country comparison. Rev Environ Econ Policy 5(2):219–239. doi:10.1093/reep/rer002

Greenhouse Gas Reporting Program (GHGRP) (2012) EPA Greenhouse Gas Reporting data–subpart PP–suppliers of carbon dioxide. US Environmental Protection Agency, Washington, DC, http://www2.epa.gov/ghgreporting

Henriksen CB, Hussey K, Holm PE (2011) Exploiting soil-management strategies for climate mitigation in the European Union: maximizing "win–win" solutions across policy regimes. Ecol Soc 16(4). doi:10.5751/ES-04176-160422

Hirabayashi Y, Mahendran R, Koirala S, Konoshima L, Yamazaki D, Watanabe S, Kim H, Kanae S (2013) Global flood risk under climate change. Nature Clim Change, advance online publication.

doi:10.1038/nclimate1911, http://www.nature.com/nclimate/journal/vaop/ncurrent/abs/nclimate1911.html#supplementary-information
Hussey K, Pittock J (2012) The energy-water nexus: Managing the links between energy and water for a sustainable future. Ecol Soc 17(1):31 [online]. doi:10.5751/ES-04641-170131
IPCC (2007a) Climate Change 2007: impacts, adaptation and vulnerability, Working Group II contribution to the fourth assessment report of the Intergovernmental Panel on Climate Change. International Panel on Climate Change, Geneva
IPCC (2007b) Mitigation of climate change: summary for policymakers, Contribution of Working Group III to the fourth assessment report of the Intergovernmental Panel on Climate Change. Intergovernmental Panel on Climate Change, Geneva
IWMI (2011) Innovative electricity scheme sparks rural development in India's Gujarat State. Success Stories (9). http://www.iwmi.cgiar.org/Publications/Success_Stories/PDF/2011/Issue_9-Innovative%20electricity%20scheme%20sparks.pdf
IWMI (2012) Boosting water benefits in West Bengal. Success Stories (14) http://www.iwmi.cgiar.org/Publications/Success_Stories/PDF/2011/Issue_14-Groundwater_in_West_Bengal.pdf
Jackson RB, Jobbágy EG, Avissar R, Roy SB, Barrett DJ, Cook CW, Farley KA, le Maitre DC, McCarl BA, Murray BC (2005) Trading water for carbon with biological carbon sequestration. Science 310(5756):1944–1947. doi:10.1126/science.1119282
King CW, Webber ME (2008) Water intensity of transportation. Environ Sci Technol 42(21): 7866–7872
Kleiner K (2009) The bright prospect of biochar. Nature (0906):72–74
Konikow LF (2013) Groundwater depletion in the United States (1900–2008). US Department of the Interior, US Geological Survey, Reston, pp 2013–5079. http://pubs.usgs.gov/sir/2013/5079/
Lam SK, Chen D, Mosier AR, Roush R (2013) The potential for carbon sequestration in Australian agricultural soils is technically and economically limited. Sci Rep 3. doi:10.1038/srep02179, http://www.nature.com/srep/2013/130710/srep02179/abs/srep02179.html#supplementary-information
Lankford B (2009) Viewpoint – the right irrigation? Policy directions for agricultural water management in Sub-Saharan Africa. Water Altern 2(3):476–480
Lukasiewicz A, Finlayson CM, Pittock J (2013) Identifying low risk climate change adaptation in catchment management while avoiding unintended consequences. National Climate Change Adaptation Research Facility, Gold Coast
MacDonald AM, Bonsor HC, Dochartaigh BÉÓ, Taylor RG (2012) Quantitative maps of groundwater resources in Africa. Environ Res Lett 7(2):024009
Mauter MS, Alvarez PJJ, Burton A, Cafaro DC, Chen W, Gregory KB, Jiang G, Li Q, Pittock J, Reible D, Schnoor JL (2014) Regional variation in water-related impacts of shale gas development and implications for emerging international plays. Environ Sci Technol. doi:10.1021/es405432k
Millennium Ecosystem Assessment (MEA) (2005) Ecosystems and human well-being: wetlands and water synthesis. World Resources Institute, Washington, DC
Mushtaq S, Maraseni TN, Maroulis J, Hafeez M (2009) Energy and water tradeoffs in enhancing food security: a selective international assessment. Energy Policy 37(9):3635–3644
National Research Council (2013) Induced seismicity potential in energy technologies. The National Academies Press, Washington, DC
NETL (2008) Estimating freshwater needs to meet future thermoelectric generation requirements, 2008 update. DOE/NETL-400/2008/1339, September 30. US Department of Energy, National Energy Technology Laboratory, Pittsburgh
Notaras M (2011) All biofuel policies are political. Our World, 18 Feb 2011) [online]
NWC (2011) The national water initiative – securing Australia's water future: 2011 assessment. National Water Commission, Canberra
Openshaw K (2000) A review of Jatropha curcas: an oil plant of unfulfilled promise. Biomass Bioenerg 19(1):1–15. doi:10.1016/S0961-9534(00)00019-2

Petrone KC, Hughes JD, Van Niel TG, Silberstein RP (2010) Streamflow decline in southwestern Australia, 1950–2008. Geophys Res Lett 37(11), L11401. doi:10.1029/2010GL043102
Phillips A (2014) World's largest solar thermal plant uses as much water as two holes on nearby golf course. Climate Progress, 13 Feb 2014 [online]
Pittock J (2010) A pale reflection of political reality: integration of global climate, wetland, and biodiversity agreements. Climate Law 1(3):343–373
Pittock J (2011) National climate change policies and sustainable water management: conflicts and synergies. Ecol Soc 16(2):25 [online]
Pittock J (2013) Lessons from adaptation to sustain freshwater environments in the Murray–Darling Basin, Australia. Wiley Interdiscip Rev Clim Change 4(6):429–438. doi:10.1002/wcc.230
Pittock J, Finlayson CM (2011) Australia's Murray-Darling Basin: freshwater ecosystem conservation options in an era of climate change. Mar Freshw Res 62:232–243
Pittock J, Lankford BA (2010) Environmental water requirements: demand management in an era of water scarcity. J Integr Environ Sci 7(1):75–93. doi:10.1080/19438151003603159
Pittock J, Finlayson CM, Gardner A, McKay C (2010) Changing character: the Ramsar convention on wetlands and climate change in the Murray-Darling Basin, Australia. Environ Plan Law J 27(6):401–425
Pittock J, Hussey K, McGlennon S (2013) Australian climate, energy and water policies: conflicts and synergies. Aust Geogr 44(1):3–22. doi:10.1080/00049182.2013.765345
Pittock J, Hussey K, Dovers, S (eds) (2015) Climate, energy and water. Cambridge University Press, Cambridge
Poff NL, Matthews JH (2013) Environmental flows in the Anthropocence: past progress and future prospects. Curr Opin Environ Sustain 5(6):667–675. doi:10.1016/j.cosust.2013.11.006
Pritchard D (2009) Reducing emissions from deforestation and forest degradation in developing countries (REDD) – the link with wetlands. Foundation for International Environmental Law and Development, London
Richter BD (2010) Re-thinking environmental flows: from allocations and reserves to sustainability boundaries. River Res Appl 26(8):1052–1063. doi:10.1002/rra.1320
Rogelj J, McCollum DL, Riahi K (2013) The UN's 'Sustainable Energy for All' initiative is compatible with a warming limit of 2°C. Nat Clim Change 3(6):545–551. doi:10.1038/nclimate1806
Roulo C (2011) Ball State University's geothermal system will be largest in US Contractor (5 December 2011) [online]
Sims RE, Mabee W, Saddler JN, Taylor M (2010) An overview of second generation biofuel technologies. Bioresour Technol 101(6):1570–1580
Sohi S, Loez-Capel E, Krull E, Bol R (2009) Biochar's roles in soil and climate change: a review of research needs, CSIRO land and water science report 05/09. CSIRO, Canberra
Stirzaker R, Pittock J (2014) The case for a new irrigation research agenda for Sub-Saharan Africa. In: Pittock J, Grafton RQ, White C (eds) Water, food and agricultural sustainability in Southern Africa. Tilde University Press, Prahran, pp 91–107
Sullivan A, Pittock J (2014) Agricultural policies and irrigation in Africa. In: Pittock J, Grafton RQ, White C (eds) Water, food and agricultural sustainability in Southern Africa. Tilde University Press, Prahran, pp 30–54
U.S. Census Bureau (2007) American Housing Survey for the United States. Current Housing reports, Series H150/07. U.S. Government Printing Office, Washington, DC
van Dijk AIJM, Keenan RJ (2007) Planted forests and water in perspective. For Ecol Manage 251(1–2):1–9
Vermeulen S, Cotula L (2010) Over the heads of local people: consultation, consent, and recompense in large-scale land deals for biofuels projects in Africa. J Peasant Stud 37(4): 899–916. doi:10.1080/03066150.2010.512463
Wada Y, Beek L, Bierkens MF (2012) Nonsustainable groundwater sustaining irrigation: a global assessment. Water Resour Res 48(6):W00L06

Wallis PJ, Ward MB, Pittock J, Hussey K, Bamsey H, Denis A, Kenway SJ, King CW, Mushtaq S, Retamal ML, Spies BR (2014) The water impacts of climate change mitigation measures. Clim Change 125(2):209–220. doi:10.1007/s10584-014-1156-6

Webber Energy Group (n.d.) Texas interactive power stimulator. University of Texas, Austin

Williams J, Pittock J (2012) Unconventional gas production and water resources, Lessons from the United States on better governance – a workshop for Australian Government officials. The Australian National University and United States Studies Centre, Canberra

WWDR (2012) Managing water under uncertainty and risk, Fourth world water development report. UNESCO, Paris

Yang J, Xu M, Zhang X, Hu Q, Sommerfeld M, Chen Y (2011) Life-cycle analysis on biodiesel production from microalgae: water footprint and nutrients balance. Bioresour Technol 102(1): 159–165. doi:10.1016/j.biortech.2010.07.017

Zoomers A (2010) Globalisation and the foreignisation of space: seven processes driving the current global land grab. J Peasant Stud 37(2):429–447. doi:10.1080/03066151003595325

Linking Climate Change and Groundwater

5

Timothy Richard Green

Abstract

Projected global change includes groundwater systems, which are linked with changes in climate over space and time. Consequently, global change affects key aspects of subsurface hydrology (including soil water, deeper vadose zone water, and unconfined and confined aquifer waters), surface-groundwater interactions, and water quality. Research and publications addressing projected climate effects on subsurface water are catching up with surface water studies. Even so, technological advances, new insights and understanding are needed regarding terrestrial-subsurface systems, biophysical process interactions, and feedbacks to atmospheric processes. Importantly, groundwater resources need to be assessed in the context of atmospheric CO_2 enrichment, warming trends and associated changes in intensities and frequencies of wet and dry periods, even though projections in space and time are uncertain. Potential feedbacks of groundwater on the global climate system are largely unknown, but may be stronger than previously assumed. Groundwater has been depleted in many regions, but management of subsurface storage remains an important option to meet the combined demands of agriculture, industry (particularly the energy sector), municipal and domestic water supply, and ecosystems. In many regions, groundwater is central to the water-food-energy-climate nexus. Strategic adaptation to global change must include flexible, integrated groundwater management over many decades. Adaptation itself must be adaptive over time. Further research is needed to improve our understanding of climate and groundwater interactions and to guide integrated groundwater management.

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5.1 Introduction and Motivation

Present understanding of how global change affects water resources around the world is limited. Potential impacts of global change on surface water, particularly projected regional climate patterns and trends have been studied in some detail. Studies of how subsurface waters will respond to climate change coupled with human activities have started to catch up only recently (Green et al. 2011; Taylor et al. 2013).

Challenges of understanding climate-change effects on groundwater are unique, because climate change may affect hydrogeological processes and groundwater resources directly and indirectly, in ways that have not been explored sufficiently (Dettinger and Earman 2007). Data limitations have made it impossible to determine the magnitude and direction of groundwater change due solely to climate change (Kundzewicz et al. 2007; Taylor et al. 2013). Even so, groundwater has been an historical buffer against climate variability, and *our dependence on groundwater resources is likely to increase* as water supplies are further stressed by population increase and projected increases in temperature and climatic variability over much of the globe.

Observational data and climate predictions provide abundant evidence that freshwater resources (both surface and subsurface water resources) are vulnerable and have the potential to be strongly affected by climate change, with wide-ranging consequences for society and ecosystems (Bates et al. 2008). According to Jorgensen and Yasin al-Tikiriti (2003) the effect of historical climate change on groundwater resources, which once supported irrigation and economic development in parts of the Middle East, is likely the primary cause of declining cultures there during the Stone Age. Climate change may account for approximately 20 % of projected increases in water scarcity globally (Sophocleous 2004). *Integrated groundwater management and planning into the future requires careful evaluation and understanding of climatic variability* over periods of decades to centuries, while considering the increasing stresses on those groundwater resources from population growth and industrial, agricultural, and ecological needs (Warner 2007).

5.1.1 Rising Interest in Impacts of Climate Change on Subsurface Water

In recent decades, a wide array of scientific research has been conducted to explore how water resources might respond to global change. However, research has been focused dominantly on surface-water systems, due to their visibility, accessibility and more obvious recognition of surface waters being affected by global change. Only recently are water resources managers and politicians recognising the important role played by groundwater resources in meeting the demands for drinking water, agricultural and industrial activities, and sustaining ecosystems, as well as in the adaptation to and mitigation of the impacts of climate change and coupled human activities.

Changes in global climate are expected to affect the hydrological cycle, altering surface-water levels and groundwater recharge to aquifers with various other associated impacts on natural ecosystems and human activities. Although the most noticeable impacts of climate change could be changes in surface-water levels and quality (Leith and Whitfield 1998; Winter 1983), there are potential effects on the quantity and quality of groundwater (Bear and Cheng 1999; Zektser and Loaiciga 1993).

5.1.2 What Is Global Change?

Global change may include natural and anthropogenic influences on terrestrial climate and the hydrologic cycle. Greenhouse gases are assumed to drive much of the contemporary climate change, and global atmospheric CO_2 concentration is the primary indicator of greenhouse gases, as well as a primary regulator of global climate (Petit et al. 1999). Atmospheric CO_2 concentration has been measured in the middle of the Pacific Ocean atop Mauna Loa, Hawaii at the National Centre for Environmental Prediction since 1958 (Keeling et al. 1976; Keeling et al. 2004; Thoning et al. 1989). Both CO_2 concentration and its rate of change have increased continuously over most of our lifetimes. Green et al. (2011) showed a power-law increase in CO_2 concentration with time, but projections of future greenhouse gas concentrations are based on complex "storylines" (IPCC 2007b) or Representative Concentration Pathways (RCPs) used in the Fifth Assessment Report (AR5) (IPCC 2013). *Projected climate change is based primarily on simulated responses to these projected emissions and resulting greenhouse gases.*

Atmospheric scientists are exploring complex interactions and causative factors using available data and climate models. Ice-core data have shown long-term correlation between entrapped atmospheric CO_2 and (surrogate) temperature (Petit et al. 1999); however, CO_2 changes lag behind temperature changes by approximately 1,300 years (Mudelsee 2001). The Earth's orbit and "Milankovitch cycles" seem to explain the apparent paradox, possibly working in tandem with global greenhouse warming and ocean circulation (Monnin et al. 2001). Loáiciga (2009) provided a helpful discussion of several factors in the debate over dominant drivers of climate as it relates to (ground)water resources. These types of issues in the theory and prediction of climate have not been fully resolved.

Although "global warming" is the topic of greatest public interest, changing patterns of surface level air humidity and precipitation are very important for predicting eco-hydrological impacts of multifaceted climate change. Projections from the Intergovernmental Panel on Climate Change (IPCC) show significant global warming and alterations in frequency and amount of precipitation in the twenty-first century (Le Treut et al. 2007; Mearns et al. 2007).

5.2 Climate Projections

Aquifers are recharged mainly by precipitation or through interaction with surface-water bodies. In order to quantify potential effects of climate change on groundwater systems, *future projections of climate are needed at the scales of application*.

5.2.1 Global Climate Models

Climate models come in different forms, ranging from simple energy-balance models to Earth-system models of intermediate complexity to comprehensive three-dimensional general circulation models of the atmosphere and oceans or global climate models (GCMs). GCMs are the most sophisticated tools available for simulation of the current global climate and future climate scenario projections. Over the last few decades, physical processes incorporated into these models have increased from simple rain and CO_2 emissions to complex biogeochemical (including water vapor) feedbacks (Le Treut et al. 2007: Fig. 1.2). The dominant terrestrial processes that affect large-scale climate over the next few decades are included in current climate models. Some processes important on longer time scales (e.g., global glaciation), however, are not yet included. The spatial resolution of GCMs has improved, but the simulation of extreme precipitation is dependent on model resolution, parameterisation and the thresholds chosen. In general, GCMs tend to produce too many days with weak precipitation (<10 mm d^{-1}) and too little precipitation during intense events (>10 mm d^{-1}) (Randall et al. 2007).

Considerable advances in model design have not reduced the variability of model forecasts of climate, partially because climate predictions are intrinsically affected by uncertainty and deterministic chaos (Lorenz 1963). Lorenz (1975) defined two distinct kinds of prediction problems: (1) prediction of actual properties of the climate system in response to a given initial state due to non-linearity and instability of the governing equations, and (2) determination of responses of the climate system to changes in the external forcings. Estimating future climate scenarios as a function of the concentration of atmospheric greenhouse gases is a typical example of predictions of the second kind (Le Treut et al. 2007).

Uncertainties in climate predictions arise mainly from model uncertainties and errors. A number of comprehensive model intercomparison projects were set up in the 1990s under the auspices of the World Climate Research Programme to undertake controlled conditions for model evaluation (e.g., Taylor 2001). Use of multiple simulations from a single model (ensemble or Monte Carlo approach) is a necessary and complementary approach to assess the stochastic and chaotic behaviors of the climate system. Such single-model ensemble simulations clearly indicated a large spread in the climate projections (Le Treut et al. 2007).

The ability of any particular GCM to reproduce present-day mean climate and its historical characteristics with respectable realism and good overall performance in comparison with the other models are presumed to indicate that it can be used to project credible future climates IPCC (2007b). The atmosphere-ocean coupled climate system shows different modes of variability that range widely from intra-seasonal to inter-decadal time scales. Successful simulation and prediction over a wide range of these phenomena increase confidence in the GCMs used for climate predictions of the future (Randall et al. 2007). In addition, the IPCC (2007a) showed that the global statistics of the extreme events in the current climate, especially temperature, are generally simulated well. However, GCMs have been more successful in simulating temperature extremes than precipitation extremes (Randall et al. 2007).

Uncertainty is expected with respect to what the future "picture" of global climate will be. GCMs are forced with concentrations of greenhouse gases and other constituents derived from various emissions scenarios ranging from non-mitigation scenarios to idealised long-term scenarios. The IPCC (2007b) considered six scenarios for projected climate change in the twenty-first century. These included a subset of three IPCC Special Report on Emission Scenarios (SRES; Nakićenović and Swart 2000) non-mitigation emission scenarios representing 'low' (B1), 'medium' (A1B) and 'high' (B1) scenarios. Green et al. (2011) discussed some potential spatial patterns of these scenarios across the globe. These include different projected changes in precipitation for the tropics (Neelin et al. 2006), subtropics (Wang 2005; Rowell and Jones 2006), and high latitudes (Emori and Brown 2005).

5.2.2 Downscaling

GCMs cannot provide information at scales finer than their computational grid (typically of the order of 200 × 200 km), yet processes at smaller unresolved scales are important. Thus, the usefulness of the raw output from a GCM for climate change assessment in specific regions is limited. To bridge the spatial resolution gaps for GCMs to produce realistic local climate projections, downscaling techniques are usually applied to the GCM output.

Downscaling addresses the disparity between the coarse spatial scales of GCMs and observations from local meteorological stations (Hewitson and Crane 2006; Wilby and Wigley 1997). GCMs do not accurately predict local climate, but the internal consistency of these physically-based climate models provides most-likely estimates of ratios and differences (scaling factors) from historical (base case) to predicted scenarios (Loaiciga et al. 1996) for climatic variables, such as precipitation and temperature.

Improvements to climate projections will likely come by developing regional climate models and GCMs that couple groundwater and atmospheric processes (Cohen et al. 2006; Gutowski et al. 2002). The primary challenge is the difference in scale between the large (continental) scale of GCMs and the local scale of groundwater or surface-water models, requiring daily data and spatial resolution of a few square kilometers (Bouraoui et al. 1999; Loaiciga et al. 1996).

A clearer picture of the robust aspects of regional climate change is emerging due to improvement in model resolution, the simulation of processes of importance for regional change, and the expanding set of available simulations (Christensen et al. 2007). Downscaling techniques are grouped into two main types: (1) dynamic climate modelling, and (2) empirical statistical downscaling.

A number of different approaches have been used to derive climate data series for hydrogeological studies. The complexity of approaches for obtaining the climate data series appears to have increased in recent years, ranging from the use of global averages (Loaiciga et al. 1996; Zektser and Loaiciga 1993) to the use of regional "bulk" projections (Allen et al. 2004; Brouyere et al. 2004; Vaccaro 1992; Yusoff et al. 2002) to the direct application of downscaled climate data (Jyrkama and Sykes 2007; Scibek and Allen 2006b; Scibek et al. 2007; Serrat-Capdevila et al. 2007; Toews and Allen 2009) to the use of regional climate models (Rivard et al. 2008; van Roosmalen et al. 2007, 2009). Some of the early efforts to assess potential hydrologic impacts were reviewed by Gleik (1986). Most of these hydrologic models used daily weather series generated stochastically, with climate change shifts applied for future climate scenarios. Many studies have considered a range of GCMs or the average projection from several GCMs, and a few studies have considered different downscaling methods.

Green et al. (2011) discussed dynamic and statistical downscaling as alternatives for applying GCM results at the local scales of interest. Downscaled daily temperature generally compares well with observed data, but daily precipitation amounts often do not, particularly seasonal amounts and durations of wet and dry periods. Such discrepancies are important because of the highly nonlinear responses and sensitivities of dynamic vegetation growth and water use (transpiration) to precipitation regimes (Green et al. 2007). Allen et al. (2010) used state-of-the-art downscaling methods to predict variations in recharge. They found that the variability in recharge predictions indicates that the seasonal performance of the downscaling tool is important, and that a range of GCMs should be considered for water management planning. Yang et al. (2005) noted that sufficient potential evaporation (PE) data are rarely available to identify long term trends. Thus, they made use of limited daily data to study sub-weekly structure, and used this information to downscale weekly sequences. In this way the dual objectives of downscaling weekly data and simulating daily PE sequences could both be achieved.

5.3 An Holistic View of Groundwater Hydrology: Selected Studies

This section summarizes the current state of research and understanding of climate-change effects on subsurface hydrology and surface-subsurface hydrologic interactions. Climate change, including anthropogenic-global warming and natural climate variability, can affect the quantity and quality of various components in the global hydrologic cycle in the space, time, and frequency domains (Holman 2006; IPCC 2007b; Loaiciga et al. 1996; Milly et al. 2005; Sharif and Singh 1999).

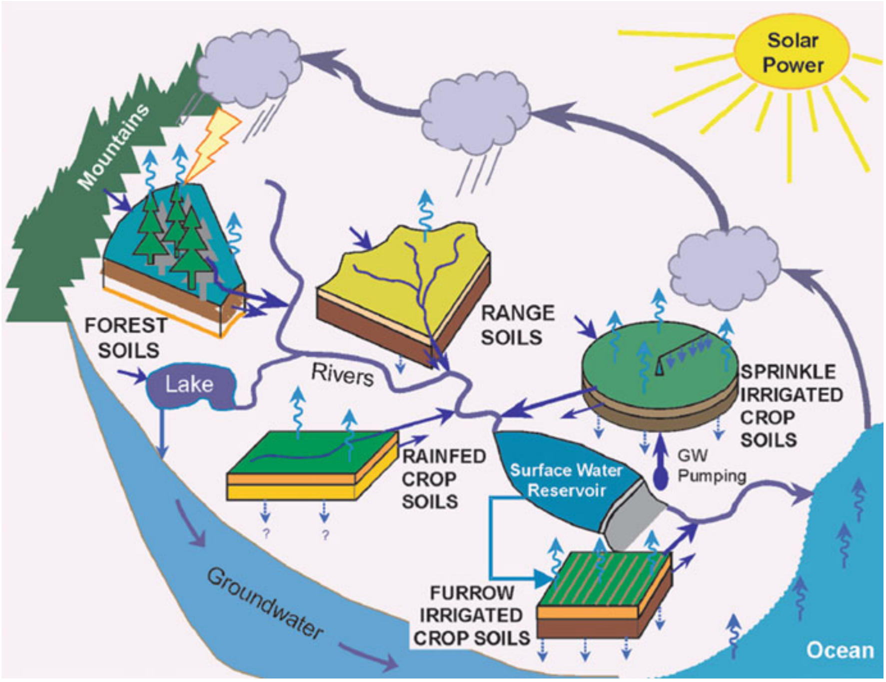

Fig. 5.1 Schematic illustration of the hydrologic cycle, including rainfed and irrigated agriculture with potential groundwater abstraction (Taken from Green and van Schilfgaarde 2006)

The components of the surface hydrologic cycle (Fig. 5.1) affected by climate change include atmospheric water vapor content, precipitation and evapotranspiration patterns, snow cover and melting of ice and glaciers, soil water content (SWC) and temperature, and surface runoff and stream flow (Bates et al. 2008). Such changes to the atmospheric and surface components of the global hydrologic cycle will likely result in changes to the subsurface hydrologic cycle within the soil, vadose zone, and aquifers of the world (Van Dijck et al. 2006). However, the potential effects of climate change on groundwater and groundwater sustainability are poorly understood. Gleeson et al. (2012) considered groundwater sustainability to include environmental, economic, or social consequences over multigenerational time scales (50–100 years). The relation between climate variables and groundwater is considered more complicated than with surface water (Holman 2006; IPCC 2007b). This understanding is confounded by the fact that groundwater-residence times can range from days to tens of thousands of years, which delays and disperses the effects of climate change, and challenges efforts to detect responses in the groundwater (Chen et al. 2004).

5.3.1 Precipitation, Evapotranspiration, and Surface Water Affect Groundwater

Precipitation and evapotranspiration are particularly important because they directly affect groundwater recharge and indirectly affect human groundwater withdrawals or discharge. Even small changes in precipitation may lead to large changes in recharge in some semiarid and arid regions (Green et al. 2007; Sandstrom 1995; Woldeamlak et al. 2007). The current section describes recent research findings regarding how atmospheric and surface-water changes will generally affect subsurface hydrologic processes in the soil and vadose zone that control infiltration and recharge to groundwater resources.

Global warming is expected to increase the spatial variability in projected precipitation producing both positive and negative changes in regional precipitation, as well as changes in seasonal patterns (Cook et al. 2014; IPCC 2007b). There is little agreement on the direction and magnitude of predicted evapotranspiration patterns (Barnett et al. 2008). However, higher air temperatures are likely to increase evapotranspiration, which may result in a reduction in runoff and SWC in some regions (Chiew and McMahon 2002). In temperate regions where plants senesce during the winter, groundwater recharge and stream baseflow could be less affected than evapotranspiration would infer due to the seasonal timing of recharge events (e.g., Hunt et al. 2013). In seasons of above average precipitation, recharge is likely to increase, and water demand, such as for irrigated agriculture, will decline because of lower temperature and solar radiation and higher humidity in such periods (Rosenberg et al. 1999). In contrast, the spatial extent and temporal duration of extreme drought are predicted to increase under future climate change (Bates et al. 2008; IPCC 2007b).

The increased variability in precipitation, temperature, and evapotranspiration that is predicted under many climate-change scenarios will likely have variable effects on different aquifers and different locations within an aquifer depending on spatial variability in hydraulic properties and distance from the recharge area(s). Chen et al. (2002) observed that groundwater levels responded to precipitation variability in a mid-continent carbonate-rock aquifer differently from well to well because of the spatial differences in permeability of overlying sediments and recharge characteristics. Additionally, groundwater levels at some locations of the aquifer responded to high-frequency precipitation events while groundwater levels in other areas did not respond. The groundwater-level response to high-frequency events may indicate the existence of highly permeable channels or preferential-flow paths from land surface to the water table (Chen et al. 2002), or differences in thickness of the unsaturated zone (e.g., Hunt et al. 2008).

Other studies indicate that even modest increases in near-surface air temperatures will alter the hydrologic cycle substantially in snowmelt-dominated regions. Seasonal streamflow is altered because the snowpack acts as a reservoir for water storage (Barnett et al. 2008; Cayan et al. 2001; Hunt et al. 2013; Mote et al. 2005; Stewart et al. 2004; Tague et al. 2008). For example, Eckhardt and Ulbrich (2003) predicted a smaller proportion of the winter precipitation will fall as snow due to warming trends in mountainous regions of central Europe and that the spring-snowmelt peak will likely be reduced while the flood risk in winter will probably increase. Unless additional reservoir storage is created to account for the earlier snowmelt runoff, the use of groundwater may increase, where available, to offset the lack of surface water later in the season when water demands are typically higher.

Spatial differences in groundwater dynamics in mountainous regions also can play a substantial role in determining streamflow responses to warming (Tague et al. 2008; Tague and Grant 2009). Tague et al. (2008) suggested that groundwater dynamics, such as subsurface drainage, are as important as topographic differences in snow regimes in determining the response of mountain landscapes to climate change. The changes in streamflow, shifting spring and summer streamflow to the winter, will likely increase competition for reservoir storage and in-stream flow for endangered species (Payne et al. 2004) and lead to summer water shortage throughout the western United States (Tague et al. 2008) and other similar semiarid and arid regions globally.

In mountainous regions, how will forecasted changes to the surface hydrologic regime affect infiltration, evapotranspiration, SWC distribution, and ultimately recharge? Singleton and Moran (2010) noted that recharge mechanisms, storage capacity, and residence times of high elevation aquifers are poorly understood. The net change in recharge in mountain aquifers due to changes in the timing of snowpack melting is generally not known in direction or magnitude, making it difficult to predict the response of mountain groundwater systems to climate change (Singleton and Moran 2010). How will mountain-front recharge and recharge in other types of mountainous systems be affected by predicted changes in the snowmelt-dominated regions? A negative feedback between early timing of snowmelt and evapotranspiration may exist in snowmelt-dominated watersheds, as earlier snowmelt increases SWC in the season when potential evapotranspiration is relatively low (Barnett et al. 2008), which may increase infiltration and recharge in mountainous regions. Later in the year, when potential evapotranspiration is greater, the shift in snowmelt timing may reduce SWC, which again reduces the effect of evapotranspiration change but has an unknown effect on net infiltration and recharge. These and other questions remain regarding subsurface hydrologic responses to climate-change effects on surface-water hydrology.

5.3.2 Soil Water and Vadose Zone Hydrology

Climate-related variables that have a substantial control on soil water include spatiotemporal patterns in precipitation, evapotranspiration, and surface-water conditions. Land use, soil texture, slope, and other biological, chemical, and physical characteristics also are known to affect SWC (Jasper et al. 2006) with associated effects on groundwater and baseflow to streams (Wang et al. 2009). Seneviratne et al. (2010) provided an extensive review of interactions and feedbacks between SWC and climate, specifically atmospheric temperature and precipitation.

Climate change and variability are expected to have profound effects on soil water and temperature (Jasper et al. 2006; Jungkunst et al. 2008). Soil water content and temperature are important factors in terrestrial biogeochemical reactions, land-atmosphere interactions, and a critical determinant of terrestrial climate. Variability in vadose-zone hydrology, shallow water tables that support SWC, and ultimately infiltration that feeds aquifers are also affected by SWC and temperature (Cohen et al. 2006; Fan et al. 2007). Spatial variations in SWC also influence atmospheric processes, such as the cumulus convective rainfall (Pielke 2001). Jungkunst et al. (2008) noted that some soil types, such as hydromorphic soils (i.e., soils which formed under prolonged periods of water saturation with seasonal aeration), will likely exhibit a higher climate-change feedback potential than other, well-aerated soils because soil organic matter losses in hydromorphic soils are predicted to be much greater than those from well-aerated soils.

Water evaporated from soils and transpired by plants is recirculated into the atmosphere, thus promoting a positive feedback mechanism for precipitation (Salas et al. 2014). The importance of this feedback depends upon the scale of interest. At the global scale, circulation of water between the land, atmosphere, and ocean is obviously important. Simulation of such circulation patterns is the basis for projecting future climates in GCMs. Moving down in scale, the coupling of land-atmosphere interactions may become looser. For this reason, hydrologic models are typically driven by measured precipitation without considering feedbacks. However, regional-scale feedback has been shown to account for a "weakly dependent" pattern of annual rainfall via "precipitation recycling" in central Sudan (Elthahir 1989), the Amazon Basin (Eltahir and Bras 1994), and other regions of the world (e.g. Eltahir and Bras 1996). At watershed areas $< 90{,}000\ km^2$, however, the recycling ratio (P/ET) of a watershed is expected to be less than 10 % based on simple scaling of annual precipitation in the Amazon basin (Eltahir 1993).

Koster et al. (2006) described the Global Land–Atmosphere Coupling Experiment (GLACE) as a model intercomparison study addressing how soil moisture anomalies affect precipitation at the GCM grid-cell resolution over the globe. The simulated strength of coupling between soil moisture and precipitation varied widely, but the ensemble multi-GCM results provided "hot spots" of relatively strong coupling based on a precipitation similarity metric. All studies indicate that the land's effect on rainfall is relatively small, though significant in places, relative to other atmospheric processes.

The vadose zone is the region between the land surface and saturated zone through which groundwater recharge occurs. It comprises complex interactions between thermal-hydrologic-geochemical processes that can affect groundwater quantity and quality. The timing and amount of groundwater recharge can be affected by the thickness of the vadose zone, as simulated for a temperate zone (Hunt et al. 2008). The vadose zone of some semiarid and arid regions responds slowly to terrestrial climate, and its long-term dynamics pose important challenges for understanding of the effects of climate change and variability on the vadose zone (Glassley et al. 2003; Phillips 1994). Glassley et al. (2003) showed that vadose-zone pore-water chemistries in the southwestern United States are still adjusting to relatively recent, post-glacial climate changes, and are not at a steady state (Phillips 1994).

5.3.3 Saturated Zone/Groundwater

Groundwater is an important component of the global water balance (Chap. 2). The use of groundwater can mitigate droughts, because many aquifers have a large storage capacity and are potentially less sensitive to short-term climate variability than surface-water bodies, which often rely on groundwater discharge to maintain baseflow conditions (Dragoni and Sukhija 2008). However, the ability to use groundwater storage to buffer rainfall deficits that affect surface-water resources will be constrained by the need to protect groundwater-dependent environmental systems (Skinner 2008).

Groundwater has and will continue to respond to changes in climate. Paleoclimate-change conditions and subsequent responses in recharge, discharge, and changes in storage are preserved in the records of groundwater major and trace-element chemistry, stable and radioactive isotope composition, and noble gas content (Bajjali and Abu-Jaber 2001; Castro et al. 2007; Hendry and Woodbury 2007). Other important components of hydrogeological systems include groundwater-fed lakes in arid and semiarid regions (Gasse 2000) and temperate climates (Hunt et al. 2013), pore-water chemistry of the vadose zone (Zuppi and Sacchi 2004), and subsurface-thermal regimes (Miyakoshi et al. 2005; Taniguchi 2002; Taniguchi et al. 2008).

Groundwater acts as a low-pass filter and provides long time-series of reconstructed temperatures and information on atmospheric-moisture transport patterns (Gasse 2000). Hiscock and Lloyd's (1992) paleohydrogeologic reconstruction of the North Lincolnshire Chalk aquifer in England revealed that recharge during the late Pleistocene (approximately the last 140,000 years) has been restricted to periods when the climate and sea-level position were similar to those of the present day. Forest clearance since about 5,000 years ago is likely to have resulted in increased recharge rates and enhanced the rate of Chalk permeability development (Hiscock and Lloyd 1992). Falling global sea levels during the last five glacial periods of the Pleistocene Ice Ages likely resulted in increased hydraulic heads in inland aquifers relative to those in the continental shelf, enhancing groundwater flow toward the coast (Faure et al. 2002). Faure et al. (2002) suggested that the "coastal oases" that formed from the groundwater discharge as springs along the exposed continental shelf had profound effects on biodiversity and carbon storage during periods of severe climatic stress. At present sea levels, submarine groundwater discharge is a well-established phenomenon that contributes substantial mass flux to oceans (Burnett et al. 2006). Gasse (2000) recommended that future paleohydrological research needs to develop solid chronologies, but also to analyze the mechanisms of water storage and losses in aquifers, obtain quantitative reconstructions of hydrological cycles, and identify atmospheric-moisture transport patterns at regional scales that affect groundwater resources.

Groundwater resources have been affected by a number of non-climatic forcings, such as contamination, reduction in streamflow (reduction in recharge), and lowering of the water table and decreased storage due to groundwater mining (primarily for irrigated agriculture). Kundzewicz et al. (2007) noted that climate-related changes to groundwater have been relatively small compared with non-climate drivers. Juckem et al. (2008) demonstrated that changes in landuse influence how climate change is translated to the groundwater system. Additionally, groundwater systems often respond more slowly and have a more substantial temporal lag to climate change than surface-water systems (Chen et al. 2004; Gurdak 2008; Gurdak et al. 2007; Hanson et al. 2004, 2006; Kundzewicz et al. 2007). Persistent and severe dry periods have even altered the hydraulic properties of aquifers, such as the transmissivity of a regional karst aquifer in France (Laroque et al. 1998).

Current vulnerabilities in water resources are strongly correlated with climate variability, due largely to precipitation variability, especially for semiarid and arid regions (Kundzewicz et al. 2007; Ouysse et al. 2010). Such regions are particularly vulnerable to climate change if groundwater reservoirs are small or not available. Even if groundwater resources are currently available, communities become more vulnerable to climate change if the ratio of stored groundwater volumes to recharge is smaller and if there are no other local water resources, such as in the isolated alluvial aquifers of Yemen (van der Gun 2010). Groundwater levels correlate more strongly with precipitation than with temperature, but temperature becomes more important for shallow aquifers (Kundzewicz et al. 2007). The complexity is exacerbated because predictions of global precipitation spatiotemporal patterns are less certain than are predicted temperature patterns. As a result, the IPCC (2007a) stated that there is no evidence for ubiquitous climate-related trends in groundwater.

Green et al. (2011) discussed climate-change effects on components of the groundwater system in some detail, including recharge, discharge, flow and storage, surface-subsurface hydrological interactions, and groundwater quality. These topics are summarized below.

5.3.4 Groundwater Recharge

Predicting the dynamics and processes interactions affecting groundwater recharge over time requires a reliable prediction of critical climate variables (Gurdak et al. 2008; Herrera-Pantoja and Hiscock 2008; Jyrkama and Sykes 2007). Recharge occurs via two general pathways in many environments: diffuse recharge to the water table and focused recharge that occurs at locations where surface-water flow is concentrated at the land surface, including stream channels, lakes, topographic depressions, irrigated-agricultural land, and other macropore, preferential-flow pathways (Small 2005). Thus, recharge is a spatially and temporally complex, sensitive function of the climate regimes, local geology and soil, topography, vegetation, surface-water hydrology, coastal flooding, and land-use activities (Candela et al. 2009; de Vries and Simmers 2002; Green et al. 2007; Holman 2006; McMahon et al. 2006). Understanding of the controls on recharge is improving (Healy 2010; Scanlon et al. 2002, 2006), but knowledge of recharge rates and mechanisms is often poor (Kundzewicz et al. 2007).

Recharge will be affected by forecasted changes in precipitation patterns. Sharif and Singh (1999) divided groundwater resources into four categories:

1. confined aquifers with upper impermeable layers where recharge primarily occurs from precipitation where the water-bearing formations outcrop at land surface.
2. unconfined (phreatic) aquifers in wet regions where rainfall is high and evapotranspiration is low. These aquifers are highly renewable because precipitation exceeds evapotranspiration throughout much of year.
3. unconfined aquifers in semiarid and arid regions that are likely to have variable annual balances between precipitation and evapotranspiration and a general drying trend under most climate-change forecasts.
4. coastal aquifers vulnerable to rising sea levels (Döll 2009) and salt-water intrusion.

Climate change and variability will likely have variable long-term effects on recharge rates and mechanisms (Aguilera and Murillo 2009; Green et al. 2007; Kundzewicz et al. 2007; Vaccaro 1992). Many climate-change studies have predicted reduced recharge (Herrera-Pantoja and Hiscock 2008); however, the effects of climate change on recharge may not necessarily be negative in all aquifers during all periods of time (Döll 2009; Gurdak and Roe 2010; Jyrkama and Sykes 2007). Case studies (listed chronologically) included various predictions for recharge in Germany (2001), eastern England (2002), western Canada (2004) and Scibek and Allen (2006a), Ontario, Canada (Döll 2009; Gurdak and Roe 2010; Jyrkama and Sykes 2007), western United States (Dettinger and Earman 2007), Russia(Kovalevskii 2007), Australia (Green et al. 2007), and upper Midwestern United States (Hunt et al. 2013). Overall, simulated trends in recharge were highly variable depending upon the base climate zone and combinations of soil and vegetation types.

Temporal climate variability, especially variability in precipitation, can have substantial effects on recharge and groundwater levels. For example, Thomsen (1989) noted that recharge in most of western Denmark at the end of the nineteenth century was only half of the recharge during the period 1964–1983 because of much greater winter rainfall. A similar study of recharge sensitivity in Western Australia by Sharma (1989) concluded that a ±20 % change in rainfall would result in a ±30 % change in recharge beneath natural grasslands and ±80 % change in recharge beneath a pine plantation, indicating that recharge is greatly influenced by land use and precipitation variability. Subsequently, Green et al. (2007) demonstrated the potential importance of changes in the timing of rainfall regimes on evapotranspiration and recharge. Eckhardt and Ulbrich (2003) predicted that mean monthly recharge and streamflow will be reduced by up to 50 % under changed precipitation regimes, that may lead to issues of local water quality, groundwater withdrawals, and hydropower generation.

Groundwater recharge and corresponding vulnerability indices have been mapped globally using a simple water balance model (Döll 2009). As noted above, estimates of recharge vary spatially with vegetation, soils and land use, and change in time depending upon the emissions scenario. For the mid-twenty-first century, Döll (2009) estimated that approximately 18 % of the global population would be affected by decreased recharge of at least 10 %, and up to a third of the population may experience increased recharge of at least 10 %. The latter increases may have pronounced effects in areas with already shallow water tables, which may be more significant than sea level rise in coastal aquifers (Kundzewicz and Döll 2009).

Temperature-depth profiles in deep boreholes are useful for estimating ground-surface temperature history and recharge, because climate change at the ground surface is stored in the subsurface thermal regime (Miyakoshi et al. 2005; Taniguchi 2002). Taniguchi (2002) showed that subsurface thermal profiles near Tokyo, Japan reveal that recharge rates increased from the 1890s to 1940s and decreased from the 1940s to 1990s, in large part related to climatic variations in the precipitation regime. Climatic conditions affect the direction of groundwater flow and the relation between surface-water bodies and subsurface-water resources. and Dragoni (Cambi and Dragoni 2000; Dragoni and Sukhija 2008; Winter 1999).

Permafrost-groundwater dynamics respond to climate change at many scales, particularly in sub-permafrost groundwater that is highly climate dependent (Haldorsen 2010). Recharge is likely to increase in areas of Alaska that experience permafrost thaw (Dragoni and Sukhija 2008; Kitabata et al. 2006). Additionally, Walvoord and Striegl (2007) proposed that long-term (>30 year) streamflow records of the Yukon River in Alaska indicate a general upward trend in ground-water contribution to streamflow. In the Qinghai-Tibet Plateau of China, ground-water flow may play an important role in permafrost degradation (Cheng and Wu 2007), where degrading permafrost caused regional lowering of the groundwater table, which has resulted in falling lake levels, shrinking wetlands, and degenerating grasslands. Climate change is expected to reduce snow cover and soil frost in boreal environments of Finland, which will increase winter floods and cause the maximum recharge and water levels to occur earlier in the year in shallow unconfined aquifers (Okkonen et al. 2009; Okkonen and Kløve 2010).

Groundwater is a crucial component of the hydrologic cycle and many water-resource projects. Thus, potential effects of climate change on recharge deserve more attention (Dettinger and Earman 2007). Scientists currently lack the necessary tools and data, such as long-term continuous monitoring of recharge processes to confidently predict recharge responses to future climate change in most environments. In many regions of the world, it is unknown whether recharge will increase or decrease under predicted climate change (Green et al. 2007). The location and timing of recharge and associated effects on groundwater supplies are insufficiently understood under future climate change and variability (Gurdak et al. 2007; Sophocleous 2004). However, water resources, especially in many semiarid and arid regions, are particularly vulnerable to the effects of climate change (Aguilera and Murillo 2009; Barthel et al. 2009; Novicky et al. 2010).

5.3.5 Groundwater Discharge

Groundwater discharge is the loss of water from an aquifer to a surface-water body, the atmosphere, or abstraction for human uses. Groundwater depletion (see Chap. 3) occurs when rates of groundwater recharge are less than rates of discharge. Over the last 50 years, groundwater depletion from direct or indirect effects of climate change and human activities, such as groundwater pumping for irrigated agriculture or urban centers (Bouraoui et al. 1999; Dams et al. 2007), has expanded from a local issue to one that affects large regions in many countries throughout the world (Alley et al. 2002; Brouyere et al. 2004; Hsu et al. 2007; Martin-Rosales et al. 2007; Moustadraf et al. 2008). Changing global groundwater discharge has even contributed to sea-level rise during the past century (Taylor et al. 2013). In particular, the rise in sea level would have been even greater if substantial quantities of water had not been stored in land-surface reservoirs or channeled into aquifers by irrigation return-flow (Sahagian et al. 1994).

Some groundwater resources could be affected substantially by climate change even if the present groundwater pumping rates are not increased, such as in the Edward aquifer in Texas, USA (Loaiciga et al. 2000) and the Chalk aquifer in eastern England (Yusoff et al. 2002). Direct or indirect effects of climate change on groundwater discharge include soil degradation, changes in water demand, and changes in irrigation or land-use practices (Brouyere et al. 2004).

The notable increase in groundwater depletion beginning in the mid-1900s is consistent with increased population in many regions and the development of high-capacity well pumps that are used to support agricultural industries and public and private drinking-water supplies. For example, parts of the High Plains (or *Ogallala*) aquifer in the United States have had substantial water-level declines since the 1950s that range from 3 to more than 50 m depending on the relative magnitudes of discharge and recharge in the aquifer (McMahon et al. 2007). Declining baseflow in the Sand Hills of Nebraska, USA has also been correlated with soil texture (Wang et al. 2009).

Under some climate scenarios, many regions may receive more precipitation. Woldeamlak et al. (2007) showed that under wet-climate scenarios, runoff was the most sensitive component, and when combined with the predicted increases in groundwater discharge, may result in rising groundwater levels and winter precipitation that increase the risk of flooding. Under dry-climate scenarios, recharge was the most sensitive component and decreases in all seasons, resulting in annual groundwater level declines by as much as 3 m. This could have adverse effects on local aquatic life in local wetlands and riverine ecosystems that rely on groundwater discharge to support baseflow (Woldeamlak et al. 2007).

Submarine groundwater discharge (SGD), or the net groundwater discharge that occurs beneath the ocean, is a large component of the global hydrologic cycle, accounting for as much as 12,000 km^3/year (Speidel and Agnew 1988) and may otherwise provide fresh water for human needs (Burnett et al. 2006; Taniguchi 2000). Quantifying submarine groundwater discharge and the biogeochemical effects on the ocean has important implications for understanding climate-change effects on oceanic processes (Windom et al. 2006). For example, high dissolved nitrogen–phosphorus ratios in SGD relative to surface waters may drive the coastal oceans toward phosphorus limitation within the coming decades, perhaps changing the present nitrogen-limited coastal primary production (Slomp and Van Cappellen 2004; Taniguchi et al. 2008).

5.3.6 Aquifer Flow and Storage

Alley (2001) noted the critical importance of groundwater storage in successfully dealing with climate change and variability. In particular, changes in groundwater storage and agricultural groundwater pumping in active semiarid basins are substantial, yet poorly understood, components of the water balance (Ruud et al. 2004). The use of groundwater storage to moderate the effects of drought increases in importance as surface-water storage becomes more limited, especially during drought periods (Alley 2001).

Prior to development, the water in storage of most s worldwide was based on local-climate conditions, ecological demands, and interactions with surface water. Water-table declines and loss of storage worldwide during the second half of the twentieth century were consistent with the development of high-capacity well pumps, aquifer development for human use, and a warming climate (Kertesz and Mika 1999). Although some regions of the world, including parts of Russia (Dzhamalov et al. 2008), may have sufficiently reliable groundwater storage under future climate change and variability, the rate of global groundwater depletion was approximately 1.6×10^{11} m^3/year during the second half of the twentieth century (Brown 2001). Postel (2001) estimated that if this rate of groundwater depletion continues, the number of people globally that will live in water-stressed countries will increase from 500 million to 3 billion by the year 2025. This problem will likely be compounded by future global-population growth, which correlates with higher groundwater pumping rates that further threaten the groundwater sustainability of many aquifers at the global scale (Loaiciga 2003). Taniguchi et al. (2008) showed that population growth and the associated increase in demand for water resources, groundwater pumping, and temporary loss of groundwater storage, have resulted in substantial land-subsidence problems for many Asian urban centers. Bultot et al. (1988) simulated changes in groundwater storage of three aquifers in Belgium in response to climate change (a doubling of CO_2 in their study) that were largely dependent on aquifer specific hydrogeologic properties, such as transmissivity, presence of perched lens, or confining units.

The water-table declines and loss of groundwater storage in the High Plains aquifer in the United States were consistently large from about the 1940s, when aquifer development became widespread across the aquifer, until about the early 1980s when rates of water-table drawdown diminished. Rosenberg et al. (1999) noted that this turn-around occurred despite a very large increase in the total acreage of irrigated agriculture between the early 1980s and mid-1990s. McGuire (2011) attributed the changes in water tables over this period to more efficient irrigation methods and economic factors, but also to the fact that precipitation in the High Plains was well above normal between 1980 and 1999 (Garbrecht and Rossel 2002).

The responsiveness of the High Plains aquifer, and other similar aquifers, is strongly suggestive that natural and human-induced changes in climate can profoundly affect the availability and future sustainability of groundwater resources. The above-normal precipitation across the High Plains aquifer region between 1980 and the late-1990s can be attributed to teleconnections from natural variations in sea-surface temperatures and atmospheric pressures across the Atlantic and Pacific Oceans (Garbrecht and Rossel 2002). During the 1980s and early 1990s, the Pacific Decadal Oscillation (PDO) (Mantua and Hare 2002) was in the positive phase of variability and the Atlantic Multidecadal Oscillation (AMO) (Kerr 2000) was in the negative phase of variability, which generally results in wetter conditions and lower frequency of drought for the High Plains region (McCabe et al. 2004).

Natural climate variability occurs on all time scales, from annual to decadal, centennial, and millennial time scales. Ghil (2002) noted that the complex nature of climate variability on multiple time scales is a major obstacle to the reliable characterisation of global climate change resulting from human activities. When anthropogenic effects on aquifers are on the same time scale as some natural climate variabilities, it is difficult to distinguish between the two (Gurdak et al. 2007; Hanson et al. 2004; Mayer and Congdon 2008). These natural variations in climate, when combined, can have profound effects on the surface-hydrologic cycle largely because of the magnitude and phase relation that can cause average or extreme climate forcings (Hanson and Dettinger 2005), such as drought, low flow in streams, changes to water quality, and adverse effects on stream ecosystems (Caruso 2002).

As a result, research efforts have characterised subsurface hydrologic and geochemical responses to climate variability on interannual to multidecadal time scales because variability on these time scales has the most tangible implications for water-resource management (Chen et al. 2002, 2004; Gurdak et al. 2007; Hanson and Dettinger 2005; Hanson et al. 2004, 2006). Climate forcings on these timescales, such as the PDO, AMO, and the El Niño/Southern Oscillation (ENSO), substantially control recharge and water-table fluctuations of the High Plains aquifer (Gurdak et al. 2007, 2008, 2009; McMahon et al. 2007), other aquifer systems of the southwestern United States (Barco et al. 2010; Hanson et al. 2006; Hanson et al. 2004), and a number of other aquifers worldwide (Ngongondo 2006), including those in many small, tropical islands in the Pacific, Indian, and Atlantic oceans (White et al. 2007). A few studies have relied on long-term historical hydrologic time series to identify climate-variability effects on groundwater levels (Chen et al. 2004; Gurdak et al. 2007; White et al. 2007).

Many questions remain regarding the control of natural climate forcings on subsurface hydrologic processes and how anthropogenic global warming may affect the frequency and magnitude of these forcings. Historical temporal patterns in the hydrologic cycle may not provide a reasonable guide to future climate conditions and hydrologic processes (Bates et al. 2008; IPCC 2007b). Future climate conditions may have substantial consequences for groundwater management and infrastructure (van der Gun 2010). Statistical stationarity of the temporal hydroclimatic dynamics is not a reasonable assumption under climate variability that has low-frequency and internal variability (such as ENSO, PDO, or AMO (McCabe et al. 2004)). Milly et al. (2008) suggested that stationarity assumptions must be replaced by nonstationary conceptual and statistical models for relevant variables in the hydroclimatic system to be properly analyzed. The concept of "shifts" instead of gradual changes in temporal statistics has been applied previously to hydrological systems (Salas and Boes 1980; Salas et al. 2014).

5.3.7 Surface-Subsurface Hydrological Interactions

Climate change has substantial implications for surface-water processes (Gosling et al. 2010), including groundwater/surface-water interactions. Some studies suggest that climate change will result in less surface-water availability, which will likely increase the need for groundwater development (Chen et al. 2004; Hsu et al. 2007). For example, climate change may extend the dry season of no or very low flows in some semiarid and arid regions, which can have a substantial effect on the overall water resources of the region if no deep or otherwise reliable groundwater resources are available (Giertz et al. 2006). Surface-water storage structures can play a vital role in augmenting groundwater recharge, especially in semiarid and arid regions (Sharda et al. 2006). Accurate low-flow stream measurements are important for groundwater-fed streams to assess the potential effects of climate change and variability, and to assess in-stream flow requirements and the nature of groundwater-surface interactions (Berg and Allen 2007). Cohen et al. (2006) showed that the responses in surface-water bodies to climate change were controlled in part by groundwater hydrodynamics and position within the watershed; water-table fluctuations were consistent and had larger-amplitude fluctuations with lake levels within the upland portions of a watershed in central Minnesota, USA. Groundwater-supported evapotranspiration varied with topography and aquifer-hydraulic conductivity, and small yet important feedbacks exist between groundwater and atmospheric processes on decadal and longer time scales. Moreover, hydrologic sensitivity of a watershed to climate change depends on feedbacks between groundwater, overland flow, and land-surface water and energy balance (Ferguson and Maxwell 2010) as well as the hydrologic regime such as lakes with and without stream outflows (e.g., Hunt et al. 2013).

The magnitude and seasonality of groundwater feedbacks to surface hydrologic processes is highly sensitive to climate change (Ferguson and Maxwell 2010).

A projected increase in the frequency of droughts has implications for surface-groundwater interactions. For example, the summer of 2003 was the hottest in Europe in more than 500 years, linked to an estimated 500 deaths in the Netherlands alone, but this could become a close-to-normal summer by about 2050 (Kabat et al. 2005). The extremely low freshwater discharge by the river Rhine in 2003 resulted in groundwater seepage of seawater to the low-lying delta, which threatened substantial areas of Dutch agriculture and horticulture. As a result, studies are underway to develop freshwater canals and additional summer water storage facilities for the region. Across regions of the High Plains aquifer in Kansas, USA, streamflow declines are historically caused by high rates of groundwater pumping, but also correlate with climate variability since the mid-1980s (Brikowski 2008). Projected climate change for the Kansas region will likely continue streamflow declines, resulting in severe consequences for surface-water supply and the strong possibility of unsustainable surface storage of water resources in the region. This will likely create even more pressure on the groundwater resources of the already-stressed High Plains aquifer. Similar findings have been identified in other climate regions, including humid, tropical and arctic catchments. Both observations and modelling suggest that climate-warming induced permafrost degradation will markedly increase baseflows of arctic and subarctic rivers and streams (Bense et al. 2009; St. Jacques and Sauchyn 2009; Walvoord and Striegl 2007).

Understanding future climate change effects will be crucial, especially for groundwater/surface-water resources already close to the limits of sustainability and under forecasted drought conditions. Groundwater withdrawals can affect streamflow strongly during dry periods (Lee and Chung 2007). Therefore, it is critically important to accurately understand the links between climate change and variations and the cycles of supply and demand that drive recharge and withdrawal of water resources. Accurate projections of climate change and variations and simulations of the responses in the water-resources system are required (Hanson and Dettinger 2005).

5.3.8 Groundwater Quality

Most studies of the effects of climate change and variability on groundwater have focused on processes that affect water quantity. Relatively few studies of climate change and variability effects on groundwater have focused on processes that will affect groundwater quality. Groundwater quality is a function of the chemical, physical, and biological characteristics of the resource. Thus, groundwater quality is expected to respond to changes in climate and human activities because of the influences of recharge, discharge, and land use on groundwater systems. The quality of water is related to specific water-use standards. The protection and enhancement of groundwater quality has been a high-priority environmental concern because of the direct implications for drinking-water health standards (Alley 1993). Also, water quality may be a limiting factor for other uses of groundwater, such as agriculture, industry, or ecosystem needs. Therefore, sustainability of water supplies under future climate change and variability is not only dependent on the quantity and quality of groundwater resources, but also on the physical hydrogeologic characteristics of the aquifer, laws, regulations, and socioeconomic factors that control the demand and use of groundwater (Reilly et al. 2008).

Global change may affect the quality of groundwater in many ways (Alley 2001; Dragoni and Sukhija 2008). Changes to recharge rates, mechanisms, and locations can affect contaminant transport, which may lead to erroneous conclusions about temporal trends in groundwater quality, particularly if only a few samples have been collected over time (Alley 2001). For example, recharge during relatively dry periods may have a greater concentration of salts and total-dissolved solids (TDS), while recharge during relatively wet periods may have a relatively lower TDS concentration (Sukhija et al. 1998). Climate variability on interannual to multi-decadal timescales has been linked with changes in spatiotemporal-precipitation patterns that can result in substantial infiltration events that mobilise large, pore-water chloride and nitrate reservoirs in the vadose zone of aquifers in semiarid and arid regions (Gurdak et al. 2007, 2008). Groundwater quality may deteriorate substantially if these large chemical reservoirs reach the water table.

Coastal regions support approximately one-quarter of the global population, but contain less than 10 % of the global-renewable water supply and are undergoing rapid-population growth (Kundzewicz et al. 2007). Sea-level rise, spatiotemporal changes in precipitation and evapotranspiration, which affect recharge, and increased groundwater pumping will likely result in more groundwater salinisation in many coastal regions (Barrocu and Dahab 2010; Beuhler 2003; IPCC 2007a; Klein and Nicholls 1999; Kundzewicz et al. 2007; Moustadraf et al. 2008; Oude Essink 1996; Oude Essink 2001, 2004; Oude Essink et al. 2010; Pierson et al. 2001; Ranjan et al. 2006a, b; Sharif and Singh 1999; Yechieli et al. 2010). Vandenbohede et al. (2008) simulated a likely 15 % increase in recharge across a Belgian coastal aquifer over the next 100 years. A 0.4 m sea-level rise increased simulated groundwater flow of fresh water toward low-lying inland areas and decreased groundwater flow toward the sea, while the increase in recharge resulted in more groundwater flow toward both low-lying inland areas and the sea. Therefore, brackish and salt water present in low-lying areas will be pushed back. Salt-water intrusion may occur from the low-lying areas into dunes, which could affect the ecology of the dunes and the drainage system used in most low-lying areas (Vandenbohede et al. 2008).

Lambrakis and Kallergis (2001) showed that over-pumping, combined with a dry period, has led to a substantial decline in groundwater quality of many Greek coastal aquifers. When simulated groundwater pumping was discontinued, the reverse process of groundwater freshening was a relatively long process, ranging from 15 to 10,000 years depending on the local geochemical conditions and flow regime (Lambrakis and Kallergis 2001). Such long periods of groundwater freshening highlight the importance of minimising the initial saltwater intrusion. The salinisation of groundwater may, in turn, affect the water quality in many rivers and estuaries (Burkett et al. 2002). Due to increasing human population, agricultural development and economic activities, the shortage of fresh groundwater for domestic, agricultural, and industrial purposes becomes more striking in coastal low-lying deltaic areas like the Mississippi, Nile, Mekong, Ganges, Po, and Rhine-Scheldt deltas (Oude Essink 1996).

Reduced groundwater recharge and increased pumping may disrupt the current balance of the freshwater/saline water boundary, resulting in saline water intrusion in coastal basins, and even inland aquifers, such as the carbonate rock aquifer in the Winnipeg region of Canada (Chen et al. 2004; Grasby and Betcher 2002). Increased groundwater pumping could induce upward leakage of groundwater with poorer-water quality, such as in the High Plains aquifer (McMahon et al. 2007). Alley (2001) also noted that the combined effects of groundwater development and climate change may lead to less dilution of contaminants in streams during low flow (baseflow from groundwater) than was assumed in setting stream-discharge permits.

A wide range of additional climate-change effects on groundwater quality are possible. Kovalevskii (2007) showed that under projected climate change, many regions of Russia will likely have increased rates of recharge that may increase rates of contaminant transport and groundwater vulnerability to various distributed and point-source contamination. The combination of the heat-island effect from urbanisation and global warming on subsurface temperatures has implications for groundwater quality because of changes to subsurface biogeochemical reactions (Knorr et al. 2005; Taniguchi et al. 2008). Additional research is needed to understand and predict the full range of effects on groundwater quality from changes in the subsurface thermal regime and various biogeochemical reactions (Aureli and Taniguchi 2006). Climate change and the global trend of increasing urbanisation may also increase flood vulnerability (Aureli and Taniguchi 2006). Flooding in urban areas could increase loading of common urban contaminants like oil, solvents, and sewage to groundwater.

Nutrient transport rates beneath agricultural lands may also be sensitive to climate change. A study of nitrogen (N) and phosphorus (P) in Sweden (Destouni and Darracq 2009) illustrated subsurface controls on nutrient loading to coastal areas that were relatively insensitive to projected climate due to a lagged response to historical nutrient inputs. However, Destouni and Darracq (2009) noted groundwater-induced emissions of greenhouse gases such as N_2O as a neglected feedback mechanism.

Relatively few studies have explored climate-change effects on pesticide fate and transport in the subsurface. Bloomfield et al. (2006) identified that the main climate drivers for changing pesticide fate and behavior are changes in rainfall seasonality and intensity, and increased temperatures. However, indirect impacts, such as land-use change are likely to have a more substantial effect on pesticides than the direct effects of climate change on pesticide fate and transport. Bloomfield et al. (2006) noted the overall effect of climate change on pesticide fate and transport is likely to be highly variable and challenging to predict because of the uncertainties associated with climate predictions.

Long-term monitoring efforts will likely provide the necessary data to observe and understand climate-related spatiotemporal trends in groundwater quality (McMahon et al. 2007; Dragoni and Sukhija 2008). Groundwater-remediation practices may consider climate-change prediction in site design. Warner (2007) noted that climate change, including shifting rainfall patterns, rising sea levels, and fluctuating river levels may affect the potential failure of a fixed-in-place remediation strategy, such as in-situ permeable reactive barrier, to capture its intended plume. The relatively short life expectancy of most engineered groundwater-remediation systems precludes the development of economically viable remediation systems for the long-term and uncertain nature of climate predictions. Warner (2007) suggested that flexibility in design of remediation systems may account for future shifts in the hydraulic gradient caused by climate change, or more likely, from human activities and groundwater pumping.

5.4 Methods for Investigating Global Change Beneath the Surface

Green et al. (2011) explored and reviewed a range of techniques for exploring subsurface effects of climate change, which are summarized here. Methods available to detect temporal changes in groundwater quantity and quality are numerous and range markedly in observation scale and "directness" of observation. The most direct, but also smallest-scale observations are obtained from head measurements in piezometers and water quality measurements of water samples obtained in wells. While in-situ measurements arguably provide the most accurate and reliable measures to detect change, spatial variability and transfer of information across scales (i.e., scaling) must be considered. Moreover, observation networks do not exist across large parts of the globe, and installing and maintaining measurement systems is expensive and labor intensive. To evaluate temporal trends at regional to global scales and to study their relationship to change in regional to global climate and human activities, studies of extensive data sets (monitoring networks) of such "point-data" are required. Hydroclimatically similar regions can be explored using a global database of historical climate data. Similarity between historical climates in different regions is a necessary starting point but may not be sufficient to constitute analogous climate change scenarios.

Most hydrogeophysical methods have the advantage that they allow detection of change over larger volumes of the subsurface, but at the expense of detail, notably regarding water chemistry. Remote sensing of systematic change in the recent past and future across the globe has limited ability to "see" watershed-scale groundwater. The major benefit of remote sensing technologies is their ability to access spatial information in remote areas where in-situ monitoring is sparse or non-existent. Furthermore, conjunctive use of well data, hydrogeophysics and remote sensing is essential.

5.4.1 Age Dating and Chemical Proxies

Tracer methods are standard tools of hydrologists to obtain constraints on the age of groundwater and on the processes and conditions experienced during recharge and upon transit in the groundwater system (Clark and Fritz 1997; Cook and Herczeg 2000; Kooi 2008b; Loosli et al. 2001; Plummer 1993). Age dating refers to methods that aim to constrain the timing of recharge, often via the time since recharge. Groundwater ages can be estimated using radioactive isotopes with well-known, stable source concentrations (e.g., ^{14}C), radioactive isotopes with variable source concentration and a daughter isotope that can be fairly uniquely linked to the mother species (e.g., $^{3}H/^{3}He$), or conservative chemical species which exhibit negligible decay and which have a well-known, systematically changing source concentration (e.g., ^{85}Kr, CFC's, SF_6).

These "direct methods" of age dating, in principle, allow construction of a continuous record of water age with distance along a flow path, thereby potentially revealing temporal changes in recharge. Accuracy of age-dating methods covering time scales of 100–500 years is low, making temporal changes in this age-range difficult to resolve.

Several "indirect" age-dating methods provide additional useful constraints on groundwater age. These methods generally determine whether a water sample is recharged before or after a known event. An absolute age of a water sample can only be calculated when the sample corresponds to a distinct event marker. The nuclear bomb test peaks in ^{3}H, ^{14}C and ^{36}Cl are key examples. These indirect methods are most useful to study spatial variability in groundwater flow systems.

Several chemical proxies are used to trace changes in groundwater flow and changes in recharge conditions associated with climate change and surface environmental change in general. Key proxies are the stable isotopes of water (Clark and Fritz 1997) and noble gases dissolved in groundwater (Porcelli et al. 2002; Stute and Schlosser 1993). Also, chloride content of groundwater and, in particular in vertical SWC profiles collected in thick vadose zones in desert areas, have been exploited to infer changes in recharge conditions (e.g. Edmunds and Tyler 2002). Although noble gases have been applied primarily in paleohydrological reconstructions of long time scales (Kooi 2008a), they should also provide valuable constraints regarding changes in groundwater systems on timescales of decades to centuries.

5.4.2 Hydrogeophysical Techniques

Three hydrogeophysical methods are particularly relevant to the study of groundwater and the changes that arise from climate variability and change:

1. electrical/electromagnetic methods,
2. subsurface temperature logging, and
3. land-based gravity surveying.

A wide range of electrical/electromagnetic imaging and logging methods can be used to study groundwater systems and their responses to climate-related phenomena. This group of methods includes spontaneous/self potential (SP), electrical resistivity, induced polarisation (IP), a range of time and frequency domain electromagnetic methods, and ground-penetrating radar (GPR). Their advantage over point sampling is that large areas can be covered either in land-based surveys or airborne surveys. Borehole logging methods can be used in a similar fashion to provide vertical profiles of these properties with depth and to constrain survey data.

Perhaps the most common application of these methods is to studies of saline water in aquifers (Dent 2007). Climate change is expected to result in higher sea levels, posing an even greater threat to coastal aquifers. Thus, these hydrogeophysical methods are ideally suited for monitoring changes in groundwater salinity over large coastal areas due to the effects of sea level rise. These techniques may prove invaluable for detecting changes in salinity over broad agricultural areas.

Subsurface temperature can be used to reconstruct climate change and land cover change, because the signal of surface temperature change is preserved in subsurface environment (e.g., Chapman et al. 1992; Davis et al. 2010; González-Rouco et al. 2009). Changes in surface temperature associated with changes in air temperature (Smerdon et al. 2009) can propagate into the subsurface, and can be detected by measuring ground temperatures up to several hundred meters deep (Beltrami and Mareschal 1995; Čermák et al. 1992). Temperature-depth profiles collected in boreholes can reveal and be used to help reconstruct the surface temperature changes due to climate change and land cover change during a few to several hundred years (Beltrami 2002; Huang et al. 2000; Roy et al. 2002). Effects of global warming on subsurface temperature subsequently affect the ecology and water quality.

Land-based gravity measurements have been used to detect changes in groundwater storage. Pool and Eychaner (1995) observed that measured gravity changes of about 13 microGal represented storage changes of about 0.30 m of water. Gravity meters are now sufficiently accurate to measure variations of about 2 microGal, and finer instrumental precision with temporal averaging. Gravity measurements have also been used to detect the changes in groundwater storage in situ (gravity profiling) and using the GRACE satellite data as discussed in the next section.

5.4.3 Remote Sensing of Space-Time Trends

Satellite remote sensing (RS) represents the most powerful method for detection and monitoring of environmental and climate change on a global scale. However, capabilities of RS to “look below the ground surface” and to detect properties that directly bear on groundwater conditions are extremely limited. Notable exceptions to this are satellite-based observations of the gravity field associated with changes in groundwater storage.

Remote sensing and earth observation technologies provide an important means of collecting groundwater-related data on a regional scale and to assess the state of the resource. Satellite remote sensing, despite drawbacks of temporal frequency and estimation errors, offers the advantages of global coverage, availability of data, metadata, error statistics, and the ability to provide meaningful spatial averages.

Aerial thermal infrared imaging is being used for mapping groundwater discharge zones in estuaries, rivers and oceans. Peterson et al. (2009) used aerial thermal infrared imaging to reveal that submarine groundwater discharge (SGD) along the western coast of the Big Island of Hawaii is often focused as point-source discharges that create buoyant groundwater plumes that mix into the coastal ocean.

Landsat, the Moderate-resolution Imaging Spectroradiometer (MODIS), the Advanced Very High Resolution Radiometer (AVHRR), and certain other instruments can resolve the location and type of vegetation, which can be used to infer a shallow water table. Altimetry measurements and Interferometric Synthetic Aperture Radar (InSAR) over time can show where subsidence is occurring, which is often an indicator of groundwater depletion. Microwave radar and radiometry measurements can be used to estimate snow and surface soil water, which further constrain groundwater assessments.

Perhaps the most valuable remote sensing technology for groundwater investigations is satellite gravimetry employed by the Gravity Recovery and Climate Experiment (GRACE) – a satellite gravimetry technology that may be used to assess groundwater storage changes. Since its launch in 2002, the GRACE satellites have been employed to detect tiny temporal changes in the gravity field of the Earth (Ramillien et al. 2008). Temporal changes in measured gravity are primarily caused by changes in total water (mass) storage (TWS) in the atmosphere, ocean and at and below the surface of the continents. GRACE is being used to generate time series of total terrestrial water variations (Tapley et al. 2004), which can be used to assess groundwater storage changes. Wahr et al. (2006) presented the first technique for deriving terrestrial water storage variations from global gravity field solutions delivered by GRACE. Rodell and Famiglietti (2002) showed in a pre-GRACE-launch study that interannual variations and trends in the High Plains aquifer water storage would be detectable by GRACE, pointing to new opportunities for groundwater remote sensing. Rodell et al. (2007) developed time series of groundwater storage variations averaged over the Mississippi River basin and its four major sub-basins using in situ data, and used these to evaluate GRACE-based estimates in which SWC and snow water equivalent fields output from a sophisticated land surface model were used to isolate groundwater from the GRACE terrestrial water storage data. At the smaller spatial scale of Illinois (145,000 km^2), Swenson et al. (2006) showed that GRACE captures the signal of changes in total water storage very well, while Yeh et al. (2006) showed that GRACE-based estimates of groundwater storage variations compared well with borehole observations on seasonal timescales. Swenson et al. (2008) used Oklahoma Mesonet data and local groundwater level observations to further refine methods to remove the SWC signal from the total water storage change signal recorded by GRACE.

Post-launch studies using GRACE data have demonstrated that when combined with ancillary measurements of surface water and SWC, GRACE is capable of monitoring changes in groundwater storage with reasonable accuracy (temporal resolution 10 days to monthly, spatial resolution 400–500 km, mass change ~9 mm water equivalent). Syed et al. (2008) also found agreement between the storage changes estimated by GRACE and the Global Land Data Assimilation System (GLDAS), where GLDAS was used to disaggregate terrestrial water storage between soil, vegetation canopy and snow.

The need to better quantify potential changes in the water cycle associated with climate change (GEWEX[1]; WATCH program[2]) has provided a major stimulus for improvement of techniques to monitor key variables and components of the hydrological cycle using space-based platforms. Advances and new developments in monitoring of soil moisture (de Jeu et al. 2008; Liu et al. 2009), precipitation, and evapotranspiration (Anderson and Kustas 2008; Kalma et al. 2008) provide crucial elements to help constrain space-time trends in groundwater recharge. Future research will undoubtedly focus on the further integration of these multi-platform and multi-parameter observations, including GRACE data, in extensive hydrological models. Recent dedicated hydrological missions for improved monitoring of soil moisture (2009: SMOS/ESA; 2011: SMAP/NASA) and precipitation (2012: GPM/NASA) enhance RS capabilities of groundwater resources assessment.

The monthly temporal resolution of GRACE is an issue for many applications, but it should be sufficient for regional groundwater assessments. To address such scale issues, Zaitchik et al. (2008) used an advanced data assimilation approach to incorporate GRACE data into a land surface model, and hence merge them with other datasets and our knowledge of physical processes as represented in the model. In simulations over the Mississippi River basin, the GRACE-assimilation groundwater storage output fit observations better than output from the open loop, and they were of much higher spatial and temporal resolution than GRACE alone. Yamamoto et al. (2008) reported the larger difference, in particular at low latitude regions, between current terrestrial water models of global river basins and GRACE data. This technique may be the key to maximising the value of GRACE data for groundwater resources studies (e.g., Fukuda et al. 2009).

5.5 Assessments of Subsurface Hydrology: Numerical Simulations

Mathematical groundwater models play a central role, both for interpreting and integrating data and for generating general insight to the response of groundwater systems to climate change and other forcings on multiple spatial and temporal scales. While observations are essential to explore and document subsurface global change, numerical models provide key tools, not only to assist in developing a process-based understanding of observed changes (i.e., hindcasting), but also

[1] http://www.gewex.org/

[2] http://www.eu-watch.org/

predict the future response of the subsurface parameters to climate change, land-use change and water management scenarios (forecasting). Distributed groundwater models simulate flow in the subsurface, both in saturated and unsaturated conditions, as well as for porous and fractured media. Specialised codes are used to simulate chemical processes, such as solute transport and reactions, heat transport, and density-dependent flow (e.g., for coastal regions). In addition to groundwater models, which form the basis for groundwater assessment, other potential models include coupled land surface-atmospheric models, biogeochemical models, surface-water hydrological models, coupled surface-water/groundwater models, and coupled land surface and variable-saturated groundwater models.

Process-based continental or global-scale hydrological models are rare. Thus, most studies develop watershed or smaller scale models, which are better constrained by available data and, thus, more easily calibrated. However, there remain challenges for coupling GCM predictions with hydrological models (Scibek and Allen 2006b; Toews and Allen 2009; Xu 1999), including issues discussed in the section Global Climate Projection.

The appropriate level of model complexity for a given problem may remain subjective, but some level of process interaction within the plant-soil-groundwater-atmospheric system must be present. Tietjen et al. (2009) made a case for at least two soil layers in a soil-vegetation model that simulated soil-water dynamics under different climatic conditions. Others have applied relatively complex, spatially distributed subsurface models and coupled surface-groundwater models (Goderniaux et al. 2009; Hunt et al. 2013; van Roosmalen et al. 2007, 2009).

Numerical model-based studies continue to improve, but for the most part, the approaches are similar to the limited examples given above and more comprehensive case studies discussed by Green et al. (2011). Models used to predict terrestrial and subsurface effects of climate change must incorporate appropriate processes and their interactions in space and time. Integration studies encompassing changes in human or socio-economic scenarios (apart from emissions scenarios), such as land use and water demand are generally lacking (Holman 2006).

5.6 The Role of Groundwater in the Water-Food-Energy-Climate Nexus

Food and energy are inextricably linked through water in many important ways (see also Chap. 4). In most regions, agriculture uses a dominant share of water, often based on senior (possibly "grandfathered") water rights. Urban areas and industries, including the energy sector, have growing water demands and substantial financial resources that often lead to purchases of water rights from agricultural stakeholders. Thus, the price of water tends to rise from the demand side. In many water limited areas, projected reductions in supply will further raise prices. In this way, climate change enters the water-food-energy nexus as an additional complicating factor.

Various organisations and funding agencies are aiming to address the water-food-energy nexus, by this or another term, including integrated modeling (Bazilian et al. 2011). A book by the World Economic Forum (Waughray 2011) covers the water-food-energy-climate nexus, including some discussion of groundwater issues. The interactive nature of problems related to this nexus will continue to spawn interest and exploration, hopefully with new innovations.

5.7 Adapting to Climate Change: Integrated Groundwater Management

Climate adaptation measures are developed to cope with the consequences of a changing climate and reduce future risks. Adaptation encompasses both national and regional strategies as well as practical measures taken at all political levels and by individuals.

In many parts of the world, groundwater is crucial to sustainable development through provision of low-cost, reliable and high-quality water supplies. About 70 % of drinking water in the European Union, 80 % of rural water supply in sub-Saharan Africa and 60 % of agricultural irrigation in India depend on groundwater (IAH 2006). Groundwater also sustains ecosystems and landscapes in humid regions in supporting wetlands and riparian areas, and also supports unique aquatic ecosystems in more arid regions and in coastal environments. The largely hidden nature of groundwater means that development is often untallied and thus uncontrolled and not incorporated into overall water resource management, resulting in over-exploitation and contamination. Thus, even without considering climate change, sustainable management of groundwater is a major challenge. Groundwater is a widely distributed resource responding at basin scales, and local stakeholders (e.g., municipalities, industrial enterprises and farmers) are influenced by national policies determining land and water use. In general, governance systems, resource policies, innovation incentives, data collection and information provision need to relate to a wide range of scales (see Chap. 6), with different adaptive management approaches in rural and urban environments (IAH 2006).

Climate change challenges the traditional assumption that past hydrological experience provides a good guide to future conditions. In times of surface-water shortages during droughts, a typical response is for groundwater resources to be abstracted as an emergency supply. Under conditions of climate change, this response could be unsustainable, especially in areas expected to experience an increase in drought frequency and duration. Also, rising sea levels under climate change will further threaten coastal freshwater aquifers, especially those already experiencing salinisation due to over-exploitation.

Alley (2006) suggested that the effects of discharge and groundwater development often take many years to become evident. Thus, governments tend to neglect the data collection and analysis needed to support informed groundwater management until problems materialize. This type of reactionary stance to groundwater management is flawed because, although some groundwater systems are renewable, many groundwater resources contain “fossil” groundwater and thus are nonrenewable natural resources on human time scales. For example, the groundwater that is removed from storage in many arid and semiarid regions was recharged during wetter periods under paleoclimate conditions (Alley et al. 2002).

Adaptation approaches can be preventative or reactive and apply to natural and social systems. Ensuring the sustainability of investments in groundwater resources planning and development, over the entire lifetime of a scheme and taking explicit account of changing climate, is referred to as *climate proofing* (CEC 2007). At a minimum, and in the absence of reliable projections of future changes in hydrological variables, adaptation processes and methods can be implemented, such as improved water use efficiency and water demand management, offering no-regrets options to cope with climate change.

The Netherlands are investing in "climate proofing" (Kabat et al. 2005) that uses hard infrastructure and softer measures, such as insurance schemes or evacuation planning, to reduce the risks of climate change and hydrologic variability to a quantifiable level that is acceptable by the society or economy. The Netherlands and the rest of the world's coastal delta regions are vulnerable to climate change and sea-level rise. Rather than coping with extreme climatic events, as people from all over the world have done over human history, climate proofing is a proactive approach to develop precautionary measures to address the low-probability but high-magnitude hydroclimatologic events forecasted under climate change and variability (Kabat et al. 2005). Climate proofing should be driven by opportunities for technological, institutional, and societal innovations, rather than by the fear of climate-change induced threats. The climate-proofing approach could be used by water-resource scientists, engineers, and managers to develop forward-thinking, innovative solutions and precautionary measures for a range of probable hydroclimatic events under future climate change. The discredited stationarity of hydroclimatology (Milly et al. 2008) may promote innovation and suitable precautionary measures to protect the sustainability of groundwater resources under projected hydroclimatic regimes. Thus the process of adaptation to climate change must itself be adaptive over time.

Potential adaptive responses include some combination of technological (e.g., deepening of existing boreholes), behavioral (e.g., altered groundwater use), managerial (e.g., altered farm irrigation practices), and policy oriented (e.g., groundwater abstractions licensing regulations) approaches. The IPCC (2007a) argued that while most technologies and strategies are studied and developed in certain countries, the effectiveness of various options to substantially reduce risks for vulnerable water-stressed areas is not yet known, particularly at higher levels of warming and related impacts. Shah (2009) noted an indirect feedback of pumping on climate change due to energy use and associated carbon emissions. This is one obvious example of the interactions between potential groundwater-atmosphere feedbacks and adaptation to global change that must be considered.

For integrated water resources management, two types of decisions deal with: (1) new investments, and (2) the operation and maintenance of existing systems. Information is needed about future water availability and demand, both of which are affected by climate change at the river-basin scale (Ballentine and Stakhiv

1993). As explained by the IPCC (2008), supply-side options generally involve increases in storage capacity or water abstraction. Demand-side adaptation options rely on the combined actions of individuals (industry users, farmers and individual consumers) and may be less reliable. Some options, such as those incurring increased pumping and treatment costs, may be inconsistent with climate change mitigation measures because they involve high energy consumption.

One of the major challenges facing water resources managers is coping with climate change uncertainty, particularly where expensive investment in infrastructure such as well-field design, construction and testing and laying of pipelines is required (Brekke et al. 2004; Taylor et al. 2013). Dessai and Hulme (2007) discussed this challenge and related questions, including: To what amount of uncertainty in climate change should we adapt? Are robust adaptation options socially, environmentally and economically acceptable and how do climate change uncertainties compare with other uncertainties such as changes in demand? The answers to these questions leading to robust adaptation decisions will require the development of probability distributions of specified outcomes (Wilby and Harris 2006) and negotiation between decision-makers and stakeholders involved in the adaptation process (Dessai and Hulme 2007). For lower income countries, availability of resources and building adaptive capacity are particularly important in order to meet water shortages and salinisation of fresh waters.

Examples of current adaptation to observed and anticipated climate change in the management of groundwater resources are few, with groundwater typically considered as part of an integrated water-supply system. Here, three examples serve to highlight the difference in approach in technically-advanced and developing country contexts. The ability of California's water supply system to adapt to long-term climate and demographic changes was examined by Tanaka et al. (2006) using a state-wide economic-engineering optimisation model of water supply management and considering two climate warming scenarios for the year 2100. However, recent drought conditions[3] raised concerns regarding long-standing issues of groundwater quality and management in California. Even so, the prediction by Tanaka et al. (2006) that California's water supply system appears physically capable of adapting to significant changes in climate and population may remain valid, albeit at significant cost. Such adaptations would entail large changes in the operation of California's large groundwater storage capacity, significant transfers of water among water users and some adoption of new technologies. In the Sacramento Valley, California, Purkey et al. (2007) used four climate time series to simulate agricultural water management with adaptation in terms of improvements in irrigation efficiency and shifts in cropping patterns during dry periods leading to lower overall water demands in the agricultural sector with associated reductions in groundwater pumping and increases in surface-water allocations to other water use sectors. Land-use adaptation to projected climate change may include management changes within land-use classes (e.g., alternative

[3] http://www.water.ca.gov/waterconditions/declaration.cfm

crop rotations) or changes in land classification (e.g., converting annual cropping systems to perennial grasslands or forests). Soil and water conservation programs already encourage some of these types of land-use changes.

A similar technological approach to that demonstrated for California is presented for the Mediterranean region of Europe. This region is experiencing rapid social and environmental changes with increasing water scarcity problems that will worsen with climate change. Iglesias et al. (2007) found that these pressures are heterogeneous across the region or water use sectors and adaptation strategies to cope with water scarcity include technology, use of strategic groundwater and better management based on preparedness rather than a crisis approach. Iglesias et al. (2007) also promoted the importance of local management at the basin level but with the potential benefits dependent on the appropriate multi-institutional and multi-stakeholder coordination.

In contrast to the examples from North America and Europe, Ojo et al. (2003) discussed the downward trends in rainfall and groundwater levels, and increases in water deficits and drought events affecting water resources availability in West Africa. There, the response strategies needed to adapt to climate change emphasize the need for water supply-demand adaptations. The mechanisms needed to implement adaptation measures include: building the capacity and manpower of water institutions in the region for hydro-climatological data collection and monitoring; the public participation and involvement of stakeholders; and the establishment of both national and regional cooperation.

Furthermore, water resources management has a clear association with many other policy areas such as energy, land use and nature conservation. In this context, groundwater is part of an emerging integrated water resources management approach that recognises society's views, reshapes planning processes, coordinates land and water resources management, recognises water quantity and quality linkages, manages surface-water and groundwater resources conjunctively, and protects and restores natural systems while considering of climate change. Also, biofuel production has implications for groundwater recharge quantity and quality (IPCC 2008).

In summary, groundwater resources stored in aquifers can be managed given reasonable scientific knowledge, adequate monitoring and sustained political commitment and provision of institutional arrangements. Although there is no single approach to relieving pressures on groundwater resources, incremental improvements in resource management and protection can be achieved now and in the future under climate change. Sustainable management of groundwater will only be possible by approaching adaptation through the effective engagement of individuals and stakeholders at community, local government and national policy levels. Adaptative decision processes in the face of global change should be addressed even to improve management and decision making in an otherwise unchanging world. That is, natural and human-induced variability under historical conditions will be better quantified and managed using new scientific advances gained under the auspices of global change research, making such work a "win-win" proposition.

5.8 Future Directions

Future work must build upon progress to date, and 12 key issues have been identified to improve understanding and guide integrated groundwater management (IGM) in light of climate change:

1. Knowledge of biophysical processes and their interactions must continue to increase, so that systems will be better understood, and estimates of projected groundwater changes and their potential feedbacks on climate will be refined, including quantification of uncertainty and associated risks.
2. Effects of projected climate change on hydrological fluxes (e.g., groundwater recharge) vary with different combinations of soils/aquifer materials, vegetation, and climate zone.
3. Long-term monitoring of terrestrial systems (groundwater, surface water, vegetation and land-use patterns) must be maintained and fortified to quantify baseline properties.
4. Shifts (versus gradual changes and linear trends) in the temporal means and variances of climate variables are probable forms of climate and groundwater changes which should be evaluated.
5. Higher spatial resolution is needed to make satellite-based gravity measurements more practical for regional groundwater management.
6. Long-term (multidecadal or greater) feedback from groundwater to atmospheric processes constitutes a knowledge gap. Paleohydrology indicates that contemporary groundwater-climate systems are not in equilibrium, due to the long memory of deep groundwater with long flow paths and large storage. Contemporary and projected climate change will have lagged and potentially amplified effects on many groundwater systems.
7. The nexus of climate change with food, water and energy security is linked directly to groundwater in many systems.
8. Issues of food and energy security, environmental protection, and social welfare all interact and depend upon improved understanding of terrestrial responses to climate change and feedback mechanisms.
9. Scaling fluxes of water and its constituents to the domains of interest for management and policy is an overarching theme for projecting groundwater responses and feedbacks with climate.
10. Information from intensive study areas must be transferred across the globe to other areas where monitoring infrastructure and research resources are not available. Mapping of global analogues in terms of climatic and terrestrial properties is a promising first-order approach.
11. Artificial recharge and managed storage and recovery projects may become more important components of many local water systems to bank excess renewable-water supplies
12. IGM needs to be both strategic and flexible over time (tactical) as projected climate-groundwater interactions become certainties, or otherwise unexpected realities. Climate proofing may offer no-regrets options to cope with climate

change by developing precautionary measures that address low-probability but high-magnitude hydroclimatologic events.

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References

Aguilera H, Murillo J (2009) The effect of possible climate change on natural groundwater recharge based on a simple model: a study of four karstic aquifers in SE Spain. Environ Geol 57(5):963–974

Allen DM, Mackie DC, Wei M (2004) Groundwater and climate change: a sensitivity analysis for the Grand Forks aquifer, southern British Columbia, Canada. Hydrogeol J 12(3):270–290

Allen DM, Cannon AJ, Toews MW, Scibek J (2010) Variabiltiy in simulated recharge using different GCMs. Water Resour Res 46:W00F03. doi:10.1029/2009WR008932

Alley WM (1993) Regional ground-water quality. Wiley, New York, 634 pp. ISBN: 978-0-471-28453-6

Alley WM (2001) Ground water and climate. Ground Water 39(2):161

Alley WM (2006) Tracking U.S. groundwater reserves for the future? Environment 48(3):10–25

Alley WM, Healy RW, LaBaugh JW, Reilly TE (2002) Flow and storage in groundwater systems. Science 296:1985–1990

Alley WM, Reilly TE, Franke OL (1999) Sustainability of ground-water resources, U.S. geological survey circular 1186. U.S. Geological Survey, Denver, 79 pp

Anderson M, Kustas W (2008) Thermal remote sensing of drought and evapotranspiration. Eos 89(26):233–234

Aureli A, Taniguchi M (2006) Groundwater assessment under the pressures of humanity and climate changes – GRAPHIC. United Nations Educational Scientific and Cultural Organization, Paris

Bajjali W, Abu-Jaber N (2001) Climatological signals of the paleogroundwater in Jordan. J Hydrol 243(1–2):133–147

Ballentine TM, Stakhiv EZ (eds) (1993) Climate change and water resources management. In: Proceedings of the national conference on climate change and water resources management

(1st), Albuquerque, New Mexico, November 4–7, 1991. US Army Corps of Engineers Institute of Water Resources, Washington D.C.
Barco J, Hogue TS, Girotto M, Kendall D, Putti M (2010) Climate signal propagation in southern California aquifers. Water Resour Res 46:W00F05. doi:10.1029/2009WR008376
Barnett TP, Pierce DW, Hidalgo HG et al (2008) Human-induced changes in the hydrology of western United States. Science 319(5866):1080–1083
Barrocu G, Dahab K (2010) Changing climate and saltwater intrusion in the Nile Delta, Egypt. In: Taniguchi M, Holman IP (eds) Groundwater response to changing climate, International Association of Hydrogeologists selected paper. CRC Press/Taylor and Francis Group, London, pp 11–25
Barthel R, Sonneveld BGJS, Goetzinger J, Keyzer MA, Pande S, Printz A, Gaiser T (2009) Integrated assessment of groundwater resources in the Oueme basin, Benin, West Africa. Phys Chem Earth 34(4–5):236–250
Bates B, Kundzewicz ZW, Wu S, Palutikof JP (2008) Climate change and water, Technical paper VI of the Intergovernmental Panel on Climate Change. Intergovernmental Panel on Climate Change Secretariat, Geneva, 210 pp
Bazilian M, Rogner H, Howells M et al (2011) Considering the energy, water and food nexus: towards an integrated modelling approach. Energy Policy 39(12):7896–7906
Bear J, Cheng HD (1999) Seawater intrusion in coastal aquifers – concepts methods and practices. Kluwer, Dordrecht/Boston/London, 625 pp
Beltrami H (2002) Climate from borehole data: energy fluxes and temperatures since 1500. Geophys Res Lett 29(23):2111. doi:10.1029/2002GL015702
Beltrami H, Mareschal JC (1995) Resolution of ground temperature histories inverted from borehole temperature data. Global Planet Chang 11(1–2):57–70
Bense VF, Ferguson G, Kooi H (2009) Evolution of shallow groundwater flow systems in areas of degrading permafrost. Geophys Res Lett 36, L22401
Berg MA, Allen DM (2007) Low flow variability in groundwater-fed streams. Can Water Resour J 32(3):227–246
Beuhler M (2003) Potential impacts of global warming on water resources in southern California. Water Sci Technol 47(7–8):165–168
Bloomfield JP, Williams RJ, Gooddy DC, Cape JN, Guha P (2006) Impacts of climate change on the fate and behaviour of pesticides in surface and groundwater – a UK perspective. Sci Total Environ 369(1–3):163–177
Bouraoui F, Vachaud G, Li LZX, Le Treut H, Chen T (1999) Evaluation of the impact of climate changes on water storage and groundwater recharge at the watershed scale. Climate Dynam 15(2):153–161
Brekke LD, Miller NL, Bashford KE, Quinn NWT, Dracup JA (2004) Climate change impacts uncertainty for water resources in the San Joaquin River Basin California. J Am Water Resour Assoc 40(1):149–164
Brikowski TH (2008) Doomed reservoirs in Kansas, USA? Climate change and groundwater mining on the Great Plains lead to unsustainable surface water storage. J Hydrol 354(1–4): 90–101
Brouyere S, Carabin G, Dassargues A (2004) Climate change impacts on groundwater resources: modelled deficits in a chalky aquifer, Geer basin, Belgium. Hydrogeol J 12(2):123–134
Brown L (2001) Running on empty. Forum Appl Res Public Policy 16:1–3
Bultot F, Coppens A, Dupriez GL, Gellens D, Meulenberghs F (1988) Repercussions of a CO_2 doubling on the water cycle and on the water balance – a case study for Belgium. J Hydrol 99(3–4):319–347
Burkett VR, Zilkoski DB, Hart DA (2002) Sea-level rise and subsidence: implications for flooding in New Orleans, Louisiana, U.S. geological survey subsidence interest group conference. U.S. Geological Survey, Galveston, pp 63–71
Burnett WC, Aggarwal PK, Aureli A et al (2006) Quantifying submarine groundwater discharge in the coastal zone via multiple methods. Sci Total Environ 367(2–3):498–543

Cambi C, Dragoni W (2000) Groundwater yield, climate changes and recharge variability: considerations arising from the modelling of a spring in the Umbria-Marche Apennines. Hydrogeologie 4:11–25

Candela L, von Igel W, Javier Elorza F, Aronica G (2009) Impact assessment of combined climate and management scenarios on groundwater resources and associated wetland (Majorca, Spain). J Hydrol 376(3–4):510–527

Caruso BS (2002) Temporal and spatial patterns of extreme low flows and effects on stream ecosystems in Otago, New Zealand. J Hydrol 257(1–4):115–133

Castro MC, Hall CM, Patriarche D, Goblet P, Ellis BR (2007) A new noble gas paleoclimate record in Texas – basic assumptions revisited. Earth Planet Sci Lett 257(1–2):170–187

Cayan DR, Kammerdiener SA, Dettinger MD, Caprio JM, Peterson DH (2001) Changes in the onset of spring in the Western United States. Bull Am Meteorol Soc 82:399–415

CEC (2007) Adapting to climate change in Europe – options for EU action. Green paper from the Commission to the Council, the European Parliament, the European Economic and Social Committee and the Committee of the Regions, Commission of the European Communities, Brussels

Čermák V, Bodri L, Šafanda J (1992) Underground temperature fields and changing climate: evidence from Cuba. Global Planet Chang 5(4):325–337

Chapman DS, Chisholm TJ, Harris RN (1992) Combining borehole temperature and meteorologic data to constrain past climate change. Global Planet Chang 6(2–4):269–281

Chen Z, Grasby SE, Osadetz KG (2002) Predicting average annual groundwater levels from climatic variables: an empirical model. J Hydrol 260(1–4):102–117

Chen Z, Grasby SE, Osadetz KG (2004) Relation between climate variability and groundwater levels in the upper carbonate aquifer, southern Manitoba. Can J Hydrol 290(1–2):43–62

Cheng G, Wu T (2007) Responses of permafrost to climate change and their environmental significance, Qinghai-Tibet Plateau. J Geophys Res 112:F02S03. doi:10.1029/2006JF000631

Chiew FHS, McMahon TA (2002) Modelling the impacts of climate change on Australian streamflow. Hydrol Process 16(6):1235–1245

Christensen JH, Hewitson B, Busuioc A et al (2007) Regional climate projections. In: Solomon S et al (eds) Climate change 2007: the physical science basis, Contribution of working group I to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge/New York

Clark I, Fritz P (1997) Environmental isotopes in hydrogeology. Lewis Publishers, Boca Raton/New York, 328 pp

Cohen D, Person M, Daannen R et al (2006) Groundwater-supported evapotranspiration within glaciated watersheds under conditions of climate change. J Hydrol 320(3–4):484–500

Cook P, Herczeg AL (2000) Environmental tracers in subsurface hydrology. Kluwer, Norwell, 529 pp

Cook BI, Smerdon JE, Seager R, Coats S (2014) Global warming and 21st century drying. Climate Dynam 43:1–21

Dams J, Woldeamlak ST, Batelaan O (2007) Forecasting land-use change and its impact on the groundwater system of the Kleine Nete catchment, Belgium. Hydrol Earth Syst Sci Discuss 4 (6):4265–4295

Davis MG, Harris RN, Chapman DS (2010) Repeat temperature measurements in boreholes from northwestern Utah link ground and air temperature changes at the decadal time scale. J Geophys Res 115:B05203. doi:10.1029/2009JB006875

de Jeu R, Wagner W, Holmes T, Dolman A, van de Giesen N, Friesen J (2008) Global soil moisture patterns observed by space borne microwave radiometers and scatterometers. Surv Geophys 29(4):399–420

de Vries JJ, Simmers I (2002) Groundwater recharge: an overview of processes and challenges. Hydrogeol J 10(1):5–17

Dent D (2007) Environmental geophysics mapping salinity and water resources. Int J Appl Earth Obs Geoinf 9(2):130–136

Dessai S, Hulme M (2007) Assessing the robustness of adaptation decisions to climate change uncertainties: a case study on water resources management in the East of England. Glob Environ Chang 17(1):59–72

Destouni G, Darracq A (2009) Nutrient cycling and N_2O emissions in a changing climate: the subsurface water system role. Environ Res Lett 4(3):035008

Dettinger MD, Earman S (2007) Western ground water and climate change – pivotal to supply sustainability or vulnerable in its own right? Ground Water 4(1):4–5

Döll P (2009) Vulnerability to the impact of climate change on renewable groundwater resources: a global-scale assessment. Environ Res Lett 4(3):035006

Dragoni W, Sukhija BS (2008) Climate change and groundwater: a short review, Geological Society special publication. Geological Society, London, pp 1–12

Dzhamalov RG, Zektser IS, Krichevets GN, Safronova TI, Sotnikova LF, Gromova YV (2008) Changes in groundwater runoff under the effect of climate and anthropogenic impact. Water Resour 35(1):15–22

Eckhardt K, Ulbrich U (2003) Potential impacts of climate change on groundwater recharge and streamflow in a central European low mountain range. J Hydrol 284(1–4):244–252

Edmunds WM, Tyler SW (2002) Unsaturated zones as archives of past climates: toward a new proxy for continental regions. Hydrogeol J 10:216–228

Eltahir EAB (1989) A feedback mechanism in annual rainfall in Central Sudan. J Hydrol 110: 323–334

Eltahir EAB (1993) Interactions of hydrology and climate in the Amazon basin. Doctorate thesis, Massachusetts Institute of Technology, Cambridge, MA, 188 pp

Eltahir EAB, Bras RL (1994) Precipitation recycling in the Amazon basin. Q J Roy Meteorol Soc 120:861–880

Eltahir EAB, Bras RL (1996) Precipitation recycling. AGU Rev Geophys 34(3):367–378

Emori S, Brown SJ (2005) Dynamic and thermodynamic changes in mean and extreme precipitation under changed climate. Geophys Res Lett 32(17):L17706. doi:10.1029/2005GL023272

Fan Y, Miguez-Macho G, Weaver CP, Walko R, Robock A (2007) Incorporating water table dynamics in climate modeling: 1. Water table observations and equilibrium water table simulations. J Geophys Res 112:D10125

Faure H, Walter RC, Grant DR (2002) The coastal oasis: ice age springs on emerged continental shelves. Global Planet Chang 33(1–2):47–56

Ferguson IM, Maxwell RM (2010) Role of groundwater in watershed response and land surface feedbacks under climate change. Water Resour Res 46:W00F02. doi:10.29/2009WR008616

Fukuda Y, Yamamoto K, Hasegawa T, Nakaegawa T, Nishijima J, Taniguchi M (2009) Monitoring groundwater variation by satellite and implications for in-situ gravity measurements. Sci Total Environ 407(9):3173–3180

Garbrecht JD, Rossel FE (2002) Decade-scale precipitation increase in Great Plains at end of 20th century. J Hydrol Eng 7(1):64–75

Gasse F (2000) Hydrological changes in the African tropics since the last glacial maximum. Quat Sci Rev 19(1–5):189–211

Ghil M (2002) Natural climate variability. In: MacCracken MC, Perry JS (eds) Encyclopedia of global environmental change. Wiley, Chichester

Giertz S, Diekkruger B, Jaeger A, Schopp M (2006) An interdisciplinary scenario analysis to assess the water availability and water consumption in the Upper Oueme catchment in Benin. Adv Geosci 9:3–13

Glassley WE, Nitao JJ, Grant CW, Johnson JW, Steefel CI, Kercher JR (2003) The impact of climate change on vadose zone pore waters and its implication for long-term monitoring. Comput Geosci 29(3):399–411

Gleeson T, Alley WM, Allen DM, Sophocleous MA, Zhou Y, Taniguchi M, Vandersteen J (2012) Towards sustainable groundwater use: setting long-term goals, backcasting, and managing adaptively. Ground Water 50:19–26

Gleick PH (1986) Methods for evaluating the regional hydrologic impacts of global climatic changes. J Hydrol 88(1–2):97–116
Goderniaux P, Brouyère S, Fowler HJ, Blenkinsop S, Therrien R, Orban P, Dassargues A (2009) Large scale surface-subsurface hydrological model to assess climate change impacts on groundwater reserves. J Hydrol 373(1–2):122–138
González-Rouco JF, Beltrami H, Zorita E, Stevens MB (2009) Borehole climatology: a discussion based on contributions from climate modeling. Clim Past 5(1):97–127
Gosling SM, Taylor RG, Arnell NW, Todd MC (2010) A comparative analysis of projected impacts of climate change on river runoff from global and catchment-scale hydrological model. Hydrol Earth Syst Sci Discuss 7:7191–7229
Grasby SE, Betcher RN (2002) Regional hydrogeochemistry of the carbonate rock aquifer, southern Manitoba. Can J Earth Sci 39:1053–1063
Green TR, van Schilfgaarde J (2006) Watershed approach, encyclopedia of soil science. Taylor & Francis, London, pp 1874–1878
Green TR, Bates BC, Charles SP, Fleming PM (2007) Physically based simulation of potential effects of carbon dioxide-altered climates on groundwater recharge. Vadose Zone J 6(3): 597–609
Green TR, Taniguchi M, Kooi H, Gurdak JJ, Allen DM, Hiscock KM, Treidel H, Aureli A (2011) Beneath the surface of global change: impacts of climate change on groundwater. J Hydrol 405 (3–4):532–560
Gurdak JJ (2008) Ground-water vulnerability: nonpoint-source contamination, climate variability, and the High Plains aquifer. VDM Verlag Publishing, Saarbrucken
Gurdak JJ, Roe CD (2010) Review: recharge rates and chemistry beneath playas of the High Plains aquifer USA. Hydrogeol J 18(8):1747–1772
Gurdak JJ, Hanson RT, McMahon PB, Bruce BW, McCray JE, Thyne GD, Reedy RC (2007) Climate variability controls on unsaturated water and chemical movement, High Plains aquifer, USA. Vadose Zone J 6(3):533–547
Gurdak JJ, Walvoord MA, McMahon PB (2008) Susceptibility to enhanced chemical migration from depression-focused preferential flow, High Plains aquifer. Vadose Zone J 7(4): 1172–1184
Gurdak JJ, Hanson RT, Green TR (2009) Effects of climate variability and change on groundwater resources of the United States, U.S. Geological Survey fact sheet 2009-3074. U.S. Geological Survey, Idaho Falls, Idaho, 4 pp.
Gutowski WJ, Vorosmarty CJ, Person M, Otles Z, Fekete B, York J (2002) A coupled land-atmosphere simulation program (CLASP): calibration and validation. J Geophys Res 107(D16, 4283):1–17
Haldorsen S, Heim M, Dale B, Landvik JY, van der Ploeg M, Leijnse A, Salvigsen O, Hagen JO, Banks D (2010) Sensitivity to long-term climate change of subpermafrost groundwater systems in Svalbard. Quatern Res 73:393–402
Hanson RT, Dettinger MD (2005) Ground water/surface water responses to global climate simulations, Santa Clara-Calleguas Basin, Ventura. Calif J Am Water Resour Assoc 41(3): 517–536
Hanson RT, Newhouse MW, Dettinger MD (2004) A methodology to assess relations between climatic variability and variations in hydrologic time series in the southwestern United States. J Hydrol 287(1–4):252–269
Hanson RT, Dettinger MD, Newhouse MW (2006) Relations between climatic variability and hydrologic time series from four alluvial basins across the southwestern United States. Hydrogeol J 14(7):1122–1146
Healy RW (2010) Estimating groundwater recharge. Cambridge University Press, Cambridge, 245 pp
Hendry MJ, Woodbury AD (2007) Clay aquitards as archives of Holocene paleoclimate: ^{18}O and thermal profiling. Ground Water 45(6):683–691

Herrera-Pantoja M, Hiscock KM (2008) The effects of climate change on potential groundwater recharge in Great Britain. Hydrol Process 22(1):73–86

Hewitson BC, Crane RG (2006) Consensus between GCM climate change projections with empirical downscaling: precipitation downscaling over South Africa. Int J Climatol 26(10): 1315–1337

Hiscock KM, Lloyd JW (1992) Palaeohydrogeological reconstructions of the North Lincolnshire Chalk, UK, for the last 140,000 years. J Hydrol 133(3–4):313–342

Holman IP (2006) Climate change impacts on groundwater recharge-uncertainty, shortcomings, and the way forward? Hydrogeol J 14(5):637–647

Hsu KC, Wang CH, Chen KC, Chen CT, Ma KW (2007) Climate-induced hydrological impacts on the groundwater system of the Pingtung Plain, Taiwan. Hydrogeol J 15(5):903–913

Huang S, Pollack HN, Po-Yu S (2000) Temperature trends over the past five centuries reconstructed from borehole temperatures. Nature 403:756–758

Hunt RJ, Prudic DE, Walker JF, Anderson MP (2008) Importance of unsaturated zone flow for simulating recharge in a humid climate. Ground Water 46(4):551–560

Hunt RJ, Walker JF, Selbig WR, Westenbroek SM, Regan RS (2013) Simulation of climate-change effects on streamflow, lake water budgets, and stream temperature using GSFLOW and SNTEMP. Trout Lake Watershed. http://pubs.usgs.gov/sir/2013/5159/

IAH (2006) Groundwater for life and livelihoods – the framework for sustainable use. In: 4th World water forum invitation and briefing, Kenilworth

Iglesias A, Garotte L, Flores F, Moneo M (2007) Challenges to manage the risk of water scarcity and climate change in the Mediterranean. Water Resour Manag 21:775–788

IPCC (2007a) Climate change 2007: impacts, adaptation and vulnerability. In: Parry ML, Canziani OF, Palutikof JP, van der Linden PJ, Hanson CE (eds) Contribution of working group II to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge/New York

IPCC (2007b) Climate change 2007: the physical science basis. In: Solomon S et al. (eds) Contribution of working group I to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge/New York, p 996

IPCC (2008) Technical paper on climate change and water. Finalized at the 37th Session of the IPCC Bureau, Geneva

IPCC (2013) Climate change 2013: the physical science basis. In: Stocker TF, Qin D, Plattner G-K, Tignor M, Allen SK, Boschung J, Nauels A, Xia Y, Bex V, Midgley PM (eds) Contribution of working group I to the fifth assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge/New York, p 1535

Jasper K, Calanca P, Fuhrer J (2006) Changes in summertime soil water patterns in complex terrain due to climatic change. J Hydrol 327(3–4):550–563

Jorgensen DG, Yasin al-Tikiriti W (2003) A hydrologic and archeologic study of climate change in Al Ain, United Arab Emirates. Global Planet Chang 35(1–2):37–49

Juckem PF, Hunt RJ, Anderson MP, Robertson DM (2008) Effects of climate and land management change on streamflow in the driftless area of Wisconsin. J Hydrol 355(1–4):123–130

Jungkunst HF, Flessa H, Scherber C, Fiedler S (2008) Groundwater level controls CO_2, N_2O and CH_4 fluxes of three different hydromorphic soil types of a temperate forest ecosystem. Soil Biol Biochem 40(8):2047–2054

Jyrkama MI, Sykes JF (2007) The impact of climate change on spatially varying groundwater recharge in the grand river watershed (Ontario). J Hydrol 338(3–4):237–250

Kabat P, van Vierssen W, Veraart J, Vellinga P, Aerts J (2005) Climate proofing the Netherlands. Nature 438(7066):283–284

Kalma J, McVicar T, McCabe M (2008) Estimating land surface evaporation: a review of methods using remotely sensed surface temperature data. Surv Geophys 29(4):421–469

Keeling CD, Bacastow RB, Bainbridge AE (1976) Atmospheric carbon dioxide variations at Mauna Loa Observatory, Hawaii. TELLUS 28(6):538–551

Keeling CD, Brix H, Gruber N (2004) Seasonal and long-term dynamics of the upper ocean carbon cycle at station ALOHA near Hawaii. Global Biogeochem Cycles 18(4):1–26
Kerr RA (2000) A North Atlantic climate pacemaker for the centuries. Science 288(5473): 1984–1986
Kertesz A, Mika J (1999) Aridification – climate change in South-Eastern Europe. Phys Chem Earth Solid Earth Geod 24(10):913–920
Kitabata H, Nishizawa K, Yoshida Y, Maruyama K (2006) Permafrost thawing in Circum-Artic and Highlands under climate change scenario projected by community climate system model (CCSM3). Sci Online Lett Atmosphere 2:53–56
Klein RJT, Nicholls RJ (1999) Assessment of coastal vulnerability to climate change. Ambio 28(2):182–187
Knorr W, Prentice IC, House JI, Holland EA (2005) Long-term sensitivity of soil carbon turnover to warming. Nature 433(7023):298–301
Kooi H (2008a) Groundwater palaeohydrology. In: Bierkens MFP, Dolman AJ, Troch PA (eds) Climate and the hydrological cycle, vol 8, IAHS special publication. International Association of Hydrological Sciences, Wallingford, UK, pp 235–254
Kooi H (2008b) Spatial variability in subsurface warming over the last three decades; insight from repeated borehole temperature measurements in the Netherlands. Earth Planet Sci Lett 270: 86–94
Koster RD, 25 others (2006) GLACE: the global land–atmosphere coupling experiment. Part I: overview. J Hydrometeorol 7:590–610
Kovalevskii VS (2007) Effect of climate changes on groundwater. Water Resour 34(2):140–152
Kundzewicz ZW, Döll P (2009) Will groundwater ease freshwater stress under climate change? Hydrol Sci J 54(4):665–675
Kundzewicz ZW, Mata LJ, Arnell NW et al (2007) Freshwater resources and their management. In: Parry ML, Canziani OF, Palutikof JP, van der Linden PJ, Hanson CE (eds) Climate change 2007: impacts adaptation and vulnerability. Cambridge University Press, Cambridge, pp 173–210
Lambrakis N, Kallergis G (2001) Reaction of subsurface coastal aquifers to climate and land use changes in Greece: modelling of groundwater refreshening patterns under natural recharge conditions. J Hydrol 245(1–4):19–31
Laroque M, Mangin A, Razack M, Banton O (1998) Contribution of correlation and spectral analyses to the regional study of a large karst aquifer (Charente, France). J Hydrol 205(3–4): 217–231
Le Treut H, Somerville R, Cubasch U, Ding Y, Mauritzen C, Mokssit A, Peterson T, Prather M (2007) Historical overview of climate change. In: Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, Tignor M, Miller HL (eds) Climate change 2007: the physical science basis, Contribution of working group I to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge/New York
Lee KS, Chung ES (2007) Hydrological effects of climate change, groundwater withdrawal, and land use in a small Korean watershed. Hydrol Process 21(22):3046–3056
Leith RMM, Whitfield PH (1998) Evidence of climate change effects on the hydrology of streams in South-Central B.C. Can Water Resour J 23(3):219–230
Liu YY, van Dijk AIJM, de Jeu RAM, Holmes TRH (2009) An analysis of spatiotemporal variations of soil and vegetation moisture from a 29-year satellite-derived data set over mainland Australia. Water Resour Res 45(7):W07405
Loaiciga HA (2003) Climate change and ground water. Ann Assoc Am Geogr 93(1):30–41
Loaiciga HA (2009) Long-term climatic change and sustainable ground water resources management. Environ Res Lett 4(3):035004
Loaiciga HA, Valdes JB, Vogel R, Garvey J, Schwarz H (1996) Global warming and the hydrologic cycle. J Hydrol 174(1–2):83–127
Loaiciga HA, Maidment DR, Valdes JB (2000) Climate-change impacts in a regional karst aquifer, Texas, USA. J Hydrol 227(1–4):173–194

Loosli HH, Aeschbach-Hertig W, others (2001) Isotopic methods and their hydrogeochemical context in the investigation of palaeowaters. In: Edmunds WM, Milne CJ (eds) Palaeowater in coastal Europe: evolution of groundwater since the late Pleistocene, Geological Society of London special publication 189. Geological Society of London, London, pp 193–212

Lorenz EN (1963) Deterministic nonperiodic flow. J Atmos Sci 20:130–141

Lorenz EN (1975) Climate predictability. In: The physical basis of climate and climate modelling, WMO GARP publication series no 16. WMO, Geneva, pp 132–136

Mantua NJ, Hare SR (2002) The Pacific decadal oscillation. J Oceanogr 58(1):35–44

Martin-Rosales W, Pulido-Bosch A, Vallejos A, Gisbert J, Andreu JM, Sanchez-Martos F (2007) Hydrological implications of desertification in southeastern Spain. Hydrol Sci J 52(6): 1146–1161

Mayer TD, Congdon RD (2008) Evaluating climate variability and pumping effects in statistical analyses. Ground Water 46(2):212–227

McCabe GJ, Palecki MA, Betancourt JL (2004) Pacifica and Atlantic Ocean influences on multidecadal drought frequency in the United States. Proc Natl Acad Sci 101(12):4136–4141

McGuire VL (2011) Water-level changes in the High Plains aquifer, predevelopment to 2009, 2007–08, and 2008–09, and change in water in storage, predevelopment to 2009, U.S. Geological Survey Scientific Investigations report 2011–5089, 13 p, available on the web at http://pubs.usgs.gov/sir/2011/5089/

McMahon PB, Dennehy KF, Bruce BW, Bohlke JK, Michel RL, Gurdak JJ, Hurlbut DB (2006) Storage and transit time of chemicals in thick unsaturated zones under rangeland and irrigated cropland, High Plains United States. Water Resour Res 42:W03314

McMahon PB, Dennehy KF, Bruce BW, Gurdak JJ, Qi SL (2007) Water-quality assessment of the High Plains aquifer, 1999–2004. U.S. Geological Survey. Professional Paper 1749, 136 pp., Reston, Virginia.

Mearns LO, Hulme M, Carter TR, Leemans R, Lal M, Whetton P (2007) Climate scenario development. In: Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, Tignor M, Miller HL (eds) Climate change 2007: the physical science basis, Contribution of working group I to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge/New York, pp 739–768

Milly PCD, Dunne KA, Vecchia AV (2005) Global pattern of trends in streamflow and water availability in a changing climate. Nature 438:347–350

Milly PCD, Betancourt JL, Falkenmark M, Hirsch RM, Kundzewicz ZW, Lettenmaier DP, Stouffer RJ (2008) Stationarity is dead: whither water management. Science 319:573–574

Miyakoshi A, Taniguchi M, Okubo Y, Uemura T (2005) Evaluations of subsurface flow for reconstructions of climate change using borehole temperature and isotope data in Kamchatka. Phys Earth Planet In 152(4):335–342

Monnin E, Indermuhle A, Dallenbach A, Fluckiger J, Stauffer B, Stocker TF, Raynaud D, Barnola J-M (2001) Atmospheric CO_2 concentrations over the last glacial termination. Science 291:112–114

Mote PW, Hamlet AF, Clark MP, Lettenmaier DP (2005) Declining mountain snowpack in western North America. Am Meteorol Soc 86(1):39–49

Moustadraf J, Razack M, Sinan M (2008) Evaluation of the impacts of climate changes on the coastal Chaouia aquifer, Morocco, using numerical modeling. Hydrogeol J 16:1411–1426

Mudelsee M (2001) The phase relations among atmospheric CO_2 content, temperature and global ice volume over the past 420 ka. Quat Sci Rev 20:583–589

Nakićenović N, Swart R (2000) Special report on emissions scenarios, A special report of working group III of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge/New York

Neelin JD, Muennich M, Su H, Meyerson JE, Holloway CE (2006) Tropical drying trends in global warming models and observations. Proc Natl Acad Sci U S A 103(16):6110–6115

Ngongondo CS (2006) An analysis of long-term rainfall variability, trends and groundwater availability in the Mulunguzi river catchment area, Zomba mountain, southern Malawia. Quat Int 148:45–50

Novicky O, Kasparek L, Uhlik J (2010) Vulnerability of groundwater resources in different hydrogeology conditions to climate change. In: Taniguchi M, Holman IP (eds) Groundwater response to changing climate, International Association of Hydrogeologists. CRC Press/Taylor & Francis Group, London, pp 1–10

Ojo O, Oni F, Ogunkunle O (2003) Implications of climate variability and climate change on water resources availability and water resources management in West Africa. In: Franks S, Bloschl S, Kumagai M, Musiake K, Rosbjerg D (eds) Water resources systems – water availability and global change. International Association of Hydrological Sciences, Wallingford, pp 37–47

Okkonen J, Kløve B (2010) A conceptual and statistical approach for the analysis of climate impact on ground water table fluctuation patterns in cold conditions. J Hydrol 388:1–12

Okkonen J, Jyrkama M, Kløve B (2009) A conceptual approach for assessing the impact of climate change on groundwater and related surface waters in cold regions (Finland). Hydrogeol J 18(2):429–439

Oude Essink GHP (1996) Impact of sea level rise on groundwater flow regimes, a sensitivity analysis for the Netherlands. Delft University of Technology, Delft, 411 pp

Oude Essink GHP (2001) Salt water intrusion in a three-dimensional groundwater system in the Netherlands: a numerical study. Transp Porous Media 43:137–158

Oude Essink GHP (2004) Modeling three-dimensional density dependent groundwater flow at the island of Texel, the Netherlands. In: Cheng AHD, Ouazar D (eds) Coastal aquifer management: monitoring, modeling, and case studies. Lewis Publisher, New York, pp 77–94

Oude Essink GHP, van Baaren ES, de Louw PGB (2010) Effects of climate change on coastal groundwater systems: a modeling study in the Netherlands. Water Resour Res 46:W00F04. doi:10.1029/2009WR008719

Ouysse S, Laftouhi NE, Tajeddine K (2010) Impacts of climate variability on the water resources in the Draa basin (Morocco): analysis of the rainfall regime and groundwater recharge. In: Taniguchi M, Holman IP (eds) Groundwater response to changing climate, International Association of Hydrogeologists selected paper. CRC Press/Taylor & Francis Group, London, pp 27–48

Payne JT, Wood AW, Hamlet AF, Palmer RN, Lettenmaier DP (2004) Mitigating the effects of climate change on the water resources of the Columbia River Basin. Clim Chang 62:233–256

Peterson RN, Burnett WC, Glenn C, Johnson A (2009) Quantification of point-source groundwater discharges to the ocean from the shoreline of the Big Island, Hawaii. Limnol Oceanogr 54(3): 890–904

Petit JR, Jouzel J, Raynaud D et al (1999) Climate and atmospheric history of the past 420,000 years from the Vostok ice core, Antarctica. Nature 399(6735):429–436

Phillips FM (1994) Environmental tracers for water in desert soils of the American Southwest. Soil Sci Soc Am J 58:15–24

Pielke RA Sr (2001) Influence of the spatial distribution of vegetation and soils on the prediction of cumulus convective rainfall. Rev Geophys 39(2):151–177

Pierson WL, Nittim R, Chadwick MJ, Bishop KA, Horton PR (2001) Assessment of changes to saltwater/freshwater habitat from reductions in flow to the Richmond River estuary. Aust Water Sci Technol 43(9):89–97

Plummer LN (1993) Stable isotope enrichment in paleowaters of the Southeast Atlantic Coastal Plain, United States. Sci Total Environ 262:2016–2020

Pool DR, Eychaner JH (1995) Measurement of aquifer-storage change and specific yield using gravity surveys. Ground Water 33:425–432

Porcelli D, Ballentine CJ, Wieler R (2002) Noble gases in geochemistry and cosmochemistry, vol 47, Reviews in mineralogy and geochemistry. Geochemical Society and Mineralogical Society of America, Washington, DC

Postel S (2001) Growing more food with less water. Sci Am 284(2):46–51

Purkey DR, Joyce B, Vicuna S, Hanemann MW, Dale LL, Yates D, Dracup JA (2007) Robust analysis of future climate change impacts on water for agriculture and other sectors: a case study in the Sacramento Valley. Clim Chang 87:S109–S122 Supplement

Ramillien G, Famiglietti JS, Wahr J (2008) Detection of continental hydrology and glaciology signals from GRACE: a review. Surv Geophys 29:361–374

Randall DA, Wood RA, Bony S et al (2007) Climate models and their evaluation. In: Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, Tignor M, Miller HL (eds) Climate change 2007: the physical science basis, Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge/New York, pp 589–662

Ranjan P, Kazama S, Sawamoto M (2006a) Effects of climate change on coastal fresh groundwater resources. Glob Environ Chang 16(4):388–399

Ranjan SP, Kazama S, Sawamoto M (2006b) Effects of climate and land use changes on groundwater resources in coastal aquifers. J Environ Manage 80(1):25–35

Reilly TE, Dennehy KF, Alley WM, Cunningham WL (2008) Ground-water availability in the United States, vol 1323. US Geological Survey, 70 pp

Rivard C, Paniconi C, Gauthier MJ, François G, Sulis M, Camporese M, Larocque M, Chaumont D (2008) A modeling study of climate change impacts on recharge and surface-groundwater interactions for the Thomas Brook catchment (Annapolis Valley, Nova Scotia). In: Proceedings of the GeoEdmonton, Canadian Gotechnical Society, International Association of Hydrogeologists, Canadian national chapter joint annual conference, Edmonton

Rodell M, Famiglietti JS (2002) The potential for satellite-based monitoring of groundwater storage changes using GRACE: the High Plains aquifer Central US. J Hydrol 263(1–4): 245–256

Rodell M, Chen J, Kato H, Famiglietti JS, Nigro J, Wilson CR (2007) Estimating groundwater storage changes in the Mississippi River basin (USA) using GRACE. Hydrogeol J 15(1): 159–166

Rosenberg NJ, Epstein DJ, Wang D, Vail L, Srinivasan R, Arnold JG (1999) Possible impacts of global warming on the hydrology of the Ogallala aquifer region. Clim Change 42(4):677–692

Rowell D, Jones R (2006) Causes and uncertainty of future summer drying over Europe. Climate Dynam 27(2):281–299

Roy S, Harris RN, Rao RUM, Chapman DS (2002) Climate change in India inferred from geothermal observations. J Geophys Res B: Solid Earth 107(7):5–1

Ruud N, Harter T, Naugle A (2004) Estimation of groundwater pumping as closure to the water balance of a semi-arid, irrigated agricultural basin. J Hydrol 297(1–4):51–73

Sahagian DL, Schwartz FW, Jacobs DK (1994) Direct anthropogenic contributions to sea level rise in the twentieth century. Nature 367:54–57

Salas JD, Boes DC (1980) Shifting level modeling of hydrologic series. Adv Water Resour 3: 59–63

Salas JD, Rao GS, Anderson M, Arabi M, Francés F, Suarez W, Lavado-Casimiro WS, Green TR (2014) Introduction to hydrology. In: Wang LK, Yang CT (eds) Modern water resources engineering, Handbook of environmental engineering. Humana Press, New York, pp 1–126

Sandstrom K (1995) Modeling the effects of rainfall variability on groundwater recharge in semi-arid Tanzania. Nord Hydrol 26:313–330

Scanlon BR, Healy RW, Cook PG (2002) Choosing appropriate techniques for quantifying groundwater recharge. Hydrogeol J 10(1):18–29

Scanlon BR, Keese KE, Flint AL, Flint LE, Gaye CB, Edmunds WM, Simmers I (2006) Global synthesis of groundwater recharge in semiarid and arid regions. Hydrol Process 20:3335–3370

Scibek J, Allen DM (2006a) Comparing modelled responses of two high-permeability, unconfined aquifers to predicted climate change. Global Planet Chang 50(1–2):50–62

Scibek J, Allen DM (2006b) Modeled impacts of predicted climate change on recharge and groundwater levels. Water Resour Res 42(11):W11405. doi:10.1029/2005WR004742

Scibek J, Allen DM, Cannon AJ, Whitfield PH (2007) Groundwater-surface water interaction under scenarios of climate change using a high-resolution transient groundwater model. J Hydrol 333(2–4):165–181
Seneviratne SI, Corti T, Davin EL, Hirschi M, Jaeger EB, Lehner I, Orlowsky B, Teuling AJ (2010) Investigating soil moisture-climate interactions in a changing climate: a review. Earth Sci Rev 99(3–4):125–161
Serrat-Capdevila A, Valdes JB, Perez JG, Baird K, Mata LJ, Maddock Iii T (2007) Modeling climate change impacts – and uncertainty – on the hydrology of a riparian system: the San Pedro Basin (Arizona/Sonora). J Hydrol 347(1–2):48–66
Shah T (2009) Climate change and groundwater: India's opportunities for mitigation and adaptation. Environ Res Lett 4(3):035005
Sharda VN, Kurothe RS, Sena DR, Pande VC, Tiwari SP (2006) Estimation of groundwater recharge from water storage structures in a semi-arid climate of India. J Hydrol 329(1–2): 224–243
Sharif MM, Singh VP (1999) Effect of climate change on sea water intrusion in coastal aquifers. Hydrol Process 13(8):1277–1287
Sharma ML (1989) Impact of climate change on groundwater recharge conference on climate and water. Academy of Finland, Helsinki, pp 511–519
Singleton MJ, Moran JE (2010) Dissolved noble gas and isotopic tracers reveal vulnerability of groundwater in a small, high elevation catchment to predicted climate change. Water Resour Res 46:W00F06. doi:10.1029/2009WR008718
Skinner AC (2008) Groundwater: still out of sight but less out of mind. Q J Eng Geol Hydrogeol 41(1):5–19
Slomp CP, Van Cappellen P (2004) Nutrient inputs to the coastal ocean through submarine groundwater discharge: controls and potential impact. J Hydrol 295(1–4):64–86
Small EE (2005) Climatic controls on diffuse groundwater recharge in semiarid environments of the southwestern United States. Water Resour Res 41:W04012
Smerdon JE, Beltrami H, Creelman C, Stevens MB (2009) Characterizing land surface processes: a quantitative analysis using air-ground thermal orbits. J Geophys Res 114(D15):D15102
Sophocleous M (2004) Climate change: why should water professionals care? Ground Water 42(5):637–637
Speidel DH, Agnew AF (1988) The world water budget. In: Speidel DH, Ruedisili LC, Agnew AF (eds) Perspectives on water: uses and abuses. Oxford University Press, New York, pp 27–36
St.Jacques JM, Sauchyn DJ (2009) Increasing winter baseflow and mean annual streamflow from possible permafrost thawing in the Northwest Territories, Canada. Geophys Res Lett 36: L01401
Stewart IT, Cayan DR, Dettinger MD (2004) Changes in snowmelt runoff timing in western North America under a 'business as usual' climate change scenario. Clim Chang 62:217–232
Stute M, Schlosser P (1993) Principles and applications of the noble gas paleothermometer. In: Swart PK, Lohmann KC, McKenzie J, Savin S (eds) Climate change in continental isotopic records, vol 78, Geophysical monograph. American Geophysical Union, Washington, DC, pp 89–100
Sukhija BS, Reddy DV, Nagabhushanam P (1998) Isotopic fingerprints of paleoclimates during the last 30,000 years in deep confined groundwaters of southern India. Quatern Res 50(3): 252–260
Swenson S, Yeh PJF, Wahr J, Famiglietti J (2006) A comparison of terrestrial water storage variations from GRACE with in situ measurements from Illinois. Geophys Res Lett 33(16): L16401. doi:10.1029/2006GL026962
Swenson S, Famiglietti J, Basara J, Wahr J (2008) Estimating profile soil moisture and groundwater variations using GRACE and Oklahoma Mesonet soil moisture data. Water Resour Res 44:W01413. doi:10.1029/2007WR006057
Syed TH, Famiglietti JS, Rodell M, Chen J, Wilson CR (2008) Analysis of terrestrial water storage changes from GRACE and GLDAS. Water Resour Res 44(2):W02433

Tague C, Grant GE (2009) Groundwater dynamics mediate low-flow response to global warming in snow-dominated alpine regions. Water Resour Res 45(7):W07421

Tague C, Grant G, Farrell M, Choate J, Jefferson A (2008) Deep groundwater mediates streamflow response to climate warming in the Oregon Cascades. Clim Change 86(1–2):189–210

Tanaka SK, Zhu TJ, Lund JR et al (2006) Climate warming and water management adaptation for California. Clim Change 76:361–378

Taniguchi M (2000) Evaluation of the saltwater-groundwater interface from borehole temperature in a coastal region. Geophys Res Lett 27(5):713–716

Taniguchi M (2002) Estimations of the past groundwater recharge rate from deep borehole temperature data. Catena 48(1–2):39–51

Taniguchi M, Burnett WC, Ness GD (2008) Integrated research on subsurface environments in Asian urban areas. Sci Total Environ 404(2–3):377–392

Tapley BD, Bettadpur S, Ries JC, Thompson PF, Watkins MM (2004) GRACE measurements of mass variability in the Earth system. Science 305:503–505

Taylor KE (2001) Summarizing multiple aspects of model performance in a single diagram. J Geophys Res 106:7183–7192

Taylor RG, Scanlon B, Döll P et al (2013) Ground water and climate change. Nat Clim Chang 3(4): 322–329

Thomsen R (1989) The effects of climate variability and change on groundwater in Europe. In: Conference on climate and water. Academy of Finland, Helsinki, pp 486–500

Thoning KW, Tans PP, Komhyr WD (1989) Atmospheric carbon dioxide at Mauna Loa Observatory. 2. Analysis of the NOAA GMCC data, 1974–1985. J Geophys Res 94(D6):8549–8565

Tietjen B, Zehe E, Jeltsch F (2009) Simulating plant water availability in dry lands under climate change: a generic model of two soil layers. Water Resour Res 45(1):W01418

Toews MW, Allen DM (2009) Evaluating different GCMs for predicting spatial recharge in an irrigated arid region. J Hydrol 374(3–4):265–281

Vaccaro JJ (1992) Sensitivity of groundwater recharge estimates to climate variability and change, Columbia Plateau, Washington. J Geophys Res 97(D3):2821–2833

van der Gun JAM (2010) Climate change and alluvial aquifers in arid regions: examples from Yemen. In: Ludwig F, Kabat P, Schaik H, Valk M (eds) Climate change adaptation in the water sector. Earthscan Publishing, London, pp 159–176

Van Dijck SJE, Laouina A, Carvalho AV et al (2006) Desertification in northern Morocco due to effects of climate change on groundwater recharge. In: Kepner WG, Rubio JL, Mouat DA, Pedrazzini F (eds) Desertification in the Mediterranean region: a security issue. Springer, Dordrecht, pp 549–577

van Roosmalen L, Christensen BSB, Sonnenborg TO (2007) Regional differences in climate change impacts on groundwater and stream discharge in Denmark. Vadose Zone J 6(3): 554–571

van Roosmalen L, Sonnenborg TO, Jensen KH (2009) Impact of climate and land use change on the hydrology of a large-scale agricultural catchment. Water Resour Res 45(7):W00A15

Vandenbohede A, Luyten K, Lebbe L (2008) Effects of global change on heterogeneous coastal aquifers: a case study in Belgium. J Coast Res 24(2 Suppl B):160–170

Wahr J, Swenson S, Velicogna I (2006) The accuracy of GRACE mass estimates. Geophys Res Lett 33:L06401. doi:10.1029/2005GL025305

Walvoord MA, Striegl RG (2007) Increased groundwater to stream discharge from permafrost thawing in the Yukon River basin: potential impacts on lateral export of carbon and nitrogen. Geophys Res Lett 34:L12402

Wang G (2005) Agricultural drought in a future climate: results from 15 global climate models participating in the IPCC 4th assessment. Climate Dynam 25(7):739–753

Wang T, Istanbulluoglu E, Lenters J, Scott D (2009) On the role of groundwater and soil texture in the regional water balance: an investigation of the Nebraska Sand Hills, USA. Water Resour Res 45(10):W10413

Warner SD (2007) Climate change, sustainability, and ground water remediation: the connection. Ground Water Monit Rem 27(4):50–52

Waughray D (ed) (2011) Water security: the water-food-energy-climate nexus, World Economic Forum. Island Press, Washington, DC

White I, Falkland T, Metutera T, Metai E, Overmars M, Perez P, Dray A, Falkland AC (2007) Climatic and human influences on groundwater in low atolls. Vadose Zone J 6(3):581–590

Wilby RL, Harris I (2006) A framework for assessing uncertainties in climate change: low-flow scenarios for the River Thames, UK. Water Resour Res 42:W02419. doi:10.1029/2005WR00406

Wilby RL, Wigley TML (1997) Downscaling general circulation model output: a review of methods and limitations. Prog Phys Geogr 21:530–548

Windom HL, Moore WS, Niencheski LFH, Jahnke RA (2006) Submarine groundwater discharge: a large, previously unrecognized source of dissolved iron to the South Atlantic Ocean. Mar Chem 102:252–266

Winter TC (1983) The interaction of lakes with variably saturated porous media. Water Resour Res 19(5):1203–1218

Winter TC (1999) Relation of streams, lakes, and wetlands to groundwater flow systems. Hydrogeol J 7(1):28–45

Woldeamlak ST, Batelaan O, De Smedt F (2007) Effects of climate change on the groundwater system in the Grote-Nete catchment, Belgium. Hydrogeol J 15(5):891–901

Xu C-Y (1999) From GCM's to river flow: a review of downscaling methods and hydrologic modelling approaches. Prog Phys Geogr 23(2):229–249

Yamamoto K, Hasegawa T, Fukuda Y, Nakaegawa T, Taniguchi M (2008) Improvement of JLG terrestrial water storage model using GRACE satellite gravity data. In: Taniguchi M, others (eds) From headwater to the ocean. CRC Press/Taylor and Francis Group, London, pp 369–374

Yang C, Chandler RE, Isham VS, Annoni C, Wheater HS (2005) Simulation and downscaling models for potential evaporation. J Hydrol 302(1–4):239–254

Yechieli Y, Shalev E, Wollman S, Kiro Y, Kafri U (2010) Response of the Mediterranean and Dead Sea coastal aquifers to sea level variations. Water Resour Res 46:W12550. doi:10.1029/2009WR008708

Yeh PJ-F, Swenson SC, Famiglietti JS, Rodell M (2006) Remote sensing of groundwater storage changes in Illinois using the Gravity Recovery and Climate Experiment (GRACE). Water Resour Res 42:W12203. doi:10.1029/2006WR005374

Yusoff I, Hiscock KM, Conway D (2002) Simulation of the impacts of climate change on groundwater resources in eastern England. In: Hiscock KM, Rivett MO, Davison RM (eds) Sustainable groundwater development. Geological Survey of London, London, pp 325–344

Zaitchik BF, Rodell M, Reichle RH (2008) Assimilation of GRACE terrestrial water storage data into a land surface model. J Hydrometeorol 9:535–548

Zektser IS, Loaiciga HA (1993) Groundwater fluxes in the global hydrologic cycle: past, present and future. J Hydrol 144(1–4):405–427

Zuppi GM, Sacchi E (2004) Hydrogeology as a climate recorder: Sahara-Sahel (North Africa) and the Po Plain (Northern Italy). Global Planet Chang 40(1–2):79–91

Part II

Governance