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Disentangling the Complexity of Groundwater Dependent Social-ecological Systems

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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.

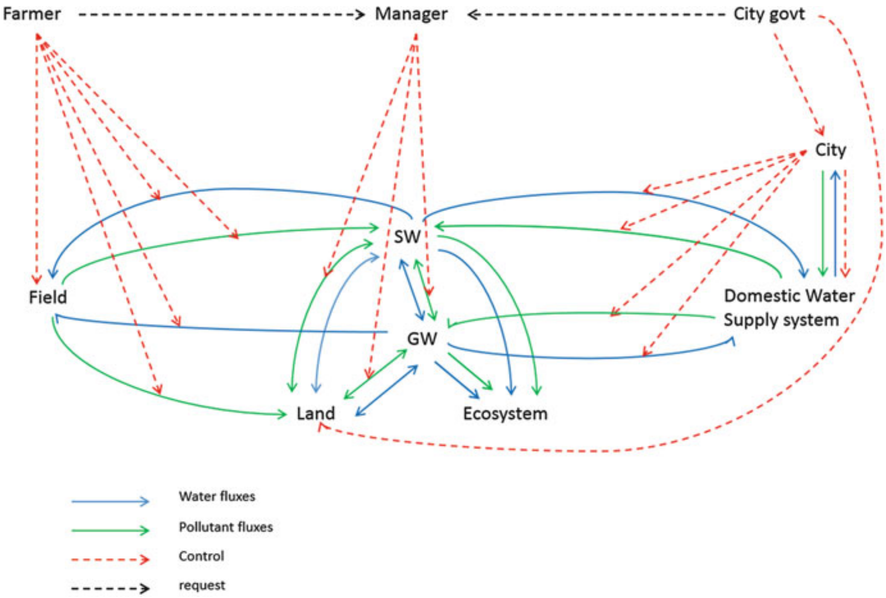

Fig. 3.1 Components at stake in groundwater management – The "IGM-scape"

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).

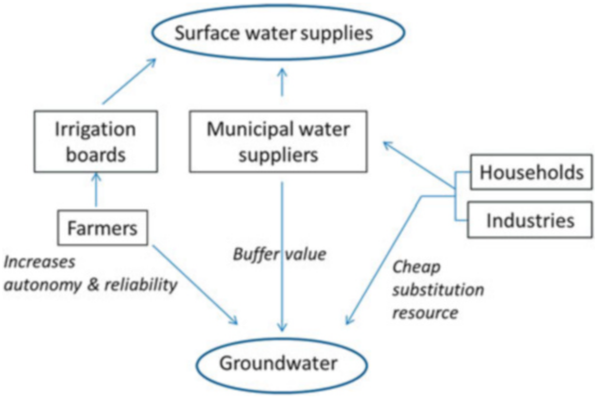

Fig. 3.2 Motivations for substituting surface with groundwater

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

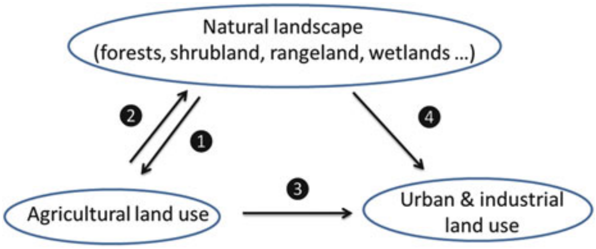

Fig. 3.3 Land use changes affecting groundwater recharge and quality

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