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Climate Change Impacts on Groundwater and Dependent Ecosystems

Bjørn Kløve[1,2*], Pertti Ala-Aho[1], Guillaume Bertrand[3], Jason J. Gurdak[4], Hans Kupfersberger[5], Jens Kvœrner[2], Timo Muotka[1], Heikki Mykrä[1], Elena Preda[6], Pekka Rossi[1], Cintia Bertacchi Uvo[7], Elzie Velasco[4], Manuel Pulido-Velázquez[8]

[1]University of Oulu, 90014 University of Oulu, Finland – tel. +358 40 5944514 -fax 358 8 553 4507 - bjorn.klove@oulu.fi

[2]Bioforsk – Norwegian Institute for Agricultural and Environmental Research, Frederik A. Dahls vei 20, N-1432 Ås, Norway - jens.kvarner@bioforsk.no

[3]University of Neuchâtel, Rue Emile-Argand 11 – CP 158 CH - 2009 Neuchâtel, Switzerland - guillaume.bertrand@email.com

[4]Department of Geosciences, San Francisco State University, 1600 Holloway Avenue, San Francisco, CA 94132, U.S.A.

[5]Joanneum Research Forschungsgesellschaft mbH. Elisabethsr. 16/II, A- 8010 Graz, Austria - hans.kupfersberger@joanneum.at

[6]University of Bucharest – Splaiul Independentei 91-95, 050095 Bucharest, Romania

[7]Water Resources Engineering, Lund University Box 118, 221 00 Lund, Sweden

[8]Research Institute of Water and Environmental Engineering, Universitat Politècnica de València, Cami de Vera s/n, 46022 Valencia, Spain

*Corresponding author

Bjørn Kløve, University of Oulu, 90014 University of Oulu, Finland – tel. +358 40 5944514 - fax 358 8 553 4507 - bjorn.klove@oulu.fi

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Abstract

Aquifers and groundwater-dependent ecosystems (GDE) are facing increasing pressure from water consumption, irrigation and climate change. These pressures modify groundwater levels and their temporal patterns and threaten vital ecosystem services such as arable land irrigation and ecosystem water requirements, especially during droughts. This review examines climate change effects on groundwater and dependent ecosystems. The mechanisms affecting natural variability in the global climate and the consequences of climate and land use changes due to anthropogenic influences are summarised based on studies from different hydrogeological strata and climate zones. The impacts on ecosystems are discussed based on current findings on factors influencing the biodiversity and functioning of aquatic and terrestrial ecosystems. The influence of changes to groundwater on GDE biodiversity and future threats posed by climate change is reviewed, using information mainly from surface water studies and knowledge of aquifer and groundwater ecosystems. Several gaps in research are identified. Due to lack of understanding of several key processes, the uncertainty associated with management techniques such as numerical modelling is high. The possibilities and roles of new methodologies such as indicators and modelling methods are discussed in the context of integrated groundwater resources management. Examples are provided of management impacts on groundwater, with recommendations on sustainable management of groundwater.

1. Introduction

Groundwater is the major freshwater store acting in the hydrological cycle. It provides water for human consumption, agriculture, industry and many groundwater-dependent ecosystems, especially during droughts. In recent decades the increasing use of groundwater for human consumption and irrigation has resulted in groundwater lowering in large parts of the world (Wada et al., 2010; Treidel et al., 2012). It is well recognised that regional depletion of groundwater resources is a global-scale problem (Konikow and Kendy, 2005). Many groundwater resources are non-renewable on meaningful time scales for both human society and ecosystems. The predicted climate change will exacerbate these concerns in many parts of the world by reducing precipitation and increasing evapotranspiration, both of which will reduce recharge and possibly increase groundwater withdrawal rates (Treidel et al., 2012). Thus, increasing awareness of the importance of wetlands and other groundwater-dependent ecosystems (GDE) has led to emphasis being placed on a better understanding of groundwater-ecosystem interactions in a changing climate (Kløve et al., 2011a, 2011b).

While the impacts of groundwater withdrawal and land use on groundwater have been investigated in numerous studies, climate change impacts on groundwater and dependent ecosystems have received less attention (Taylor et al., in review). Hydrological studies of climate change often address surface water, but fewer studies focus on groundwater (Kundzewicz and Döll, 2009; Green et al., 2011). The predicted impacts of climate warming on groundwater include changes in the magnitude and timing of recharge (e.g. Hiscock et al., 2012), typically with a shift in seasonal mean and annual groundwater levels depending on changes in the distribution of rainfall (Liu, 2011) and snow melt (Jyrkama and Sykes, 2007; Okkonen and Kløve, 2010). The predicted changes in recharge may be larger than the changes in precipitation (Ng et al., 2010). Land use and urbanisation may suppress or amplify groundwater responses to climate change. For example, afforestation can increase recharge (Chaves et al., 2012) and urbanisation can increase consumption (Taylor and Tindimugaya, 2012). In addition to human impacts, natural long-term fluctuations in groundwater levels caused by climate variability must be considered (Hanson et al., 2004; Gurdak et al., 2007; Anderson and Emanuel, 2008).

Sustainable groundwater management in the future requires groundwater to be used in a manner that can be maintained for an indefinite time without having unacceptable environmental, economic or social consequences (Alley et al., 1999). Groundwater sustainability is a value-driven process of intra- and inter-generational equity that balances the environment, society and the economy (Gleeson et al., 2010, 2012). This requires groundwater management to be approached in a holistic way, where all water uses are seen in the context of socio-economic development and protection of ecosystems and ecosystem services (Constanza et al., 1997). The current lack of knowledge on groundwater-ecosystem interactions can be seen as reflecting a neglect of groundwater in integrated watershed management plans (UNEP/CBD, 2010). The European Commission (EC) Groundwater Directive and Water Framework Directive raise concerns about how groundwater use may affect ecosystems. Re-balancing of water allocation between various human uses, as well as to biodiversity and ecosystem functioning, is clearly needed (Showstack, 2004).

This paper focuses on groundwater and associated dependent aquatic and terrestrial ecosystems; climate change effects on groundwater hydrology and geochemistry; and the processes affecting global climate, which in turn influence hydrology, groundwater ecosystem interactions and adaptation policies for groundwater and GDE management. The objective of the paper was to synthesise current knowledge on the complex interactions between climate, groundwater and ecosystems, and to examine integrated groundwater management strategies that account for human and ecosystems needs. Although there are other recent reviews on climate change and groundwater (Earman and Dettinger, 2011; Green et al., 2011; Treidel et al., 2012; Taylor et al., in review), this is the first to synthesise the effects of climate change on GDE.

2. Review of climate change impacts on GDE

2.1 Climate change and climate variability

Climate change may be perceived as alterations in the local or global climate on different time scales. Cyclical climate changes in a relative short time perspective are called climate variability. For groundwater, this variability can be illustrated as oscillating changes in recharge (P-ET), where the annual recharge varies in a regular or irregular manner that can resemble oscillations (Fig. 1). Several natural phenomena related to atmospheric and (or) oceanic circulation can affect the climate locally or globally, causing changes and (or) variability. Many of these phenomena are related to the circulation of the oceans and (or) of the atmosphere. The Gulf Stream, the North Atlantic Oscillation (NAO) and the El Niño-Southern Oscillation (ENSO) are among the best known, but other phenomena such as the Pacific Decadal Oscillation (PDO) and Atlantic Multidecadal Oscillation (AMO) have been described more recently (e.g. Huss et al., 2010).

ENSO is the result of an interaction between the Pacific Ocean and the atmosphere whereby anomalies in sea surface temperature (SST) co-vary with the intensity of the Southern Oscillation (Rasmusson and Carpenter, 1982; McPhaden et al., 2006), while NAO is an atmospheric phenomenon centred over the North Atlantic (Hurrell et al., 2003). ENSO has a typical quasi-periodic oscillation of 2-7 years, while NAO displays a yearly variability and a decadal quasi-periodic oscillation. PDO has a 10-25 year quasi-periodic cycle that is associated with decadal variability in atmospheric circulation prominent in the North Pacific, where variations in the strength of the wintertime Aleutian Low pressure system co-vary with SST from 20ºN polewards (Mantua et al., 1997). AMO is an oceanic-atmospheric phenomenon with a periodicity of 50-70 years that arises from variations in SST in the Atlantic Ocean (Kerr, 2000; Enfield et al., 2001). All these phenomena change the yearly climate regionally and seasonally, so that some regions of the world become seasonally warmer or colder, or drier or wetter, than normal. Associated with the effects of climate variability, oscillations in river runoff have been extensively described in rivers worldwide (e.g. Cullen et al., 2002; Ionita et al., 2012).

The effects of climate variability on groundwater have been less well explored than those on surface water (Green et al., 2011). However, climate variability on interannual to multidecadal time scales, including ENSO, NAO, PDO, and AMO, has also been shown to affect groundwater levels and recharge (Hanson et al., 2004; Pool, 2005; Fleming and Quilty, 2007; Gurdak et al., 2007, 2009; Anderson and Emanuel, 2008; Holman et al., 2009, 2011; Trembley et al., 2011; Venencio and Garcia, 2011). It is likely that the signals seen in recharge are also seen in groundwater levels, but as aquifers differ in size, the response to the input signal variability will be more evident in smaller aquifers (Fig. 1).

The increase in greenhouse gas emissions since the industrial revolution has also affected the climate of the Earth. For example, a small but constant increase in mean atmospheric temperature has been observed since then (IPCC, 2007). Human activities can also cause climate change locally by changing land use, water use and vegetation, with consequent impacts on hydrology (e.g. Collischon et al., 2001). These causes of climate change and variability are continuously acting and interacting with each other. The result of such a complex system is that in some periods their impacts are additive and enhance each other, while in other periods they counteract each other and their impacts decline regionally (see Fig. 1).

Coupled global climate models (GCMs), which describe the circulation of the atmosphere and the oceans, are frequently used to develop scenarios of future climate (rainfall, temperature, radiation, etc.) taking into consideration different scenarios for increases in greenhouse gases. Such scenarios include a constant increase in greenhouse gases for the next 100 years (scenario A2 of IPCC) or a reduction in emissions (scenario B1 of IPCC), or anything in between. In any case, future climate scenarios projected by GCMs in terms of precipitation and temperature may be used to force hydrological models and numerical groundwater flow models in a sequential (e.g. Okkonen and Kløve, 2011) or fully coupled manner (Therrien et al., 2007), in order to predict the impacts of future climate on recharge, groundwater flow and interactions with associated ecosystems.

2.2 Impact of climate change on the variability of groundwater quantity and quality

Climate change and variability have directly and indirectly affected, and will continue to affect, groundwater quantity and quality in many complex and unprecedented ways (Holman, 2006; Dettinger and Earman, 2007; Earman and Dettinger, 2011; Treidel et al., 2012; Taylor et al., in review). Future climate change will affect recharge rates and, in turn, the depth of groundwater levels and the amount of available groundwater (Ludwig and Moench, 2009). Much of the research to date has focused on climate change effects on the magnitude and timing of recharge (Döll, 2009; Green et al., 2011; Treidel et al., 2012), with less emphasis on whether recharge mechanisms may change, possibly from more diffuse to focused recharge mechanisms in some regions (Gurdak et al., 2007). Moreover, few papers have addressed how groundwater will be indirectly affected by the changing patterns of groundwater abstraction and (or) land use (Treidel et al., 2012). Increasing abstraction with reduced recharge can reduce groundwater levels significantly, as demonstrated conceptually in Fig. 1. More studies have addressed the potential effects of climate variability and change on recharge than natural or human-induced changes in groundwater discharge. Furthermore, groundwater quality has received far less attention than groundwater quantity (Treidel et al., 2012).

Groundwater recharge depends on the distribution, amount and timing of precipitation, evapotranspiration losses, snow cover thickness and snow melt characteristics, and land use/land cover. Warmer winter temperatures can reduced the amount of ground frost and allow more water to infiltrate into the ground, resulting in increased groundwater recharge. The potential recharge rate can increase by approximately 100 mm/year over a period of 40 years in Canada (Jyrkama and Sykes 2007). Warmer winter shift the river peak flow earlier in a year resulting in a similar shift in aquifer water levels (Scibek et al. 2007). Earlier snow melt can reduce summer low flows (Okkonen and Kløve 2011). Summer low flows will also change due to melting and retreating glaciers; leading to first an increased summer flow as more ice is melted (Kumar and Singh 1997), but eventually resulting in lower summer flow as glaciers retreat (Huss 2011). This will influence glacier fed rivers such as Po that is linked to large confined aquifers and Glomma which is linked to unconfined floodplain aquifers. Reduced river flow in dry periods will influence the groundwater exchange directly and can also lead to more groundwater abstraction as river water is less available. Lower groundwater tables can promote surface water recharge in loosing streams as hydraulic gradients increase.

Recharge to an aquifer depends on the groundwater level, with lower positions normally increasing the capture zone and recharge. The properties of the aquifer are also essential; small, shallow unconfined aquifers respond more rapidly to climate change, whereas larger and confined systems show a slower response. Unconfined aquifers, especially surficial and shallow aquifers, are more likely to have renewable groundwater on meaningful time scales and will be particularly sensitive to changes in variability and climatic conditions (Winter, 1999; Healy and Cook, 2002; Sophocleous, 2002; Lee et al., 2006). Confined and deeper aquifers are more likely to have non-renewable groundwater and will be less sensitive to the direct effects of climate variability and change. Non-renewable groundwater is vulnerable to the indirect effects of increased human abstraction to meet current water requirements (Wada et al., 2012) and future water demand under a changing climate (Treidel et al., 2012).

Predicting spatiotemporal changes in the magnitude, timing and mechanism of recharge is complex in most climate regions. For example, in semi-arid regions, only heavy rainfall events result in groundwater recharge, whereas in humid regions an increase in heavy rainfall events can reduce recharge rates because most water may be lost through runoff (Bates et al., 2008).

In cold climates, seasonal variations in water level are common where a permanent snow cover hinders groundwater recharge in winter, while snow melt water replenishes aquifers in spring (Kuusisto, 1984; Rutulis, 1989; Van der Kamp and Maathuis, 1991). It is expected that in snow-dominated regions, warmer winters will cause snow melt and groundwater recharge (e.g. Jyrkama and Sykes, 2007; Sutinen et al., 2007) and runoff to occur over longer periods and earlier in the year (e.g. Veijalainen et al., 2010). Increased aquifer recharge will increase wintertime groundwater levels (see Mäkinen et al., 2008; Okkonen and Kløve, 2010), whereas in spring and summer the groundwater levels may decrease with a warmer climate (Okkonen and Kløve, 2010).

Some studies, for example that by Hiscock et al. (2012), have used GCMs to simulate future precipitation and temperature trends based on a ‘high’ (SRES A1F1) gas emissions scenario by the end of the 21st century, and report that northern Europe will receive more winter rainfall, leading to increased groundwater recharge but during a shorter time period, and that summers will be drier, with longer periods of limited or no groundwater recharge. Dams et al. (2011) showed for a catchment in Belgium that future climate change can reduce groundwater levels, particularly in late summer-early autumn, and reduce groundwater discharge in regions with little discharge. Southern Europe will have less recharge overall and the region may become more water stressed than at present, with any increase in winter recharge unable to compensate for the reduced autumn recharge (Hiscock et al., 2012). Southern Spain is predicted to be among the worst affected regions in Europe, with almost total disappearance of recharge (Hiscock et al., 2012).

Groundwater quality changes will be a consequence of changed recharge patterns and land-use. Reduced soil frost result in more recharge and less overland flow (Okkonen and Kløve 2011).

This can increases groundwater availability (Jyrkama and Sykes 2007) but also increase risk of leaching of contaminants during winter (Okkonen et al. 2010). Warmer climate increase might influence pesticide leaching to groundwater, but the processes are complex and mainly related to land use changes driven by changes in climate (Bloomfield et al. 2007) and increased pest pressures e.g. due to lower winter mortality (Noyes et al. 2006). In cold regions, a milder climate with temperatures around freezing melting point increase the use of salt application for slippery control (Balderacchi et al. 2013). In warmer climate, less recharge can lead to further decline of groundwater levels. Reduced groundwater level increase the risk of contamination mainly from sea water intrusion in coastal aquifers (Werner et al. 2013). Increased flood can lead to river water being more polluted (e.g. Hrdinka et al. 2012) and reduced minimum flow can lead to increase riverine concentration in wastewater effluents as waters are less diluted posing a risk to groundwater in loosing streams with a direct contact to aquifers.

Changes to both groundwater and surface water levels may ultimately alter the interaction between groundwater and surface water, as well as the interaction between natural and societal water supply and demand (Hanson et al., 2012). Groundwater storage acts as a moderator of surface water response and climate feedback (Maxwell and Kollet, 2008). For example, Hanson et al. (2004) identified temporal changes in response to the low frequency variability of the Pacific Decadal Oscillation (PDO) in groundwater-surface water interactions from a small watershed in the south-western USA. Temporal changes in the PDO range of streamflow (resulting from changes in precipitation and temperature due to PDO) at a downstream location lagged behind those at an upstream location by about three-quarters of a year, which may represent a delay in sustained downstream flows owing to streamflow infiltration to the floodplain aquifer (Hanson et al., 2004). Changes in stream base flow and groundwater levels tended to precede changes in streamflow at some locations by about 1 or 2 years, which may suggest that streamflow infiltration dominates prior to sustained streamflow during wet periods (Hanson et al., 2004).

Climate-induced changes in groundwater/surface water interactions will directly affect wetlands and other GDE (Earman and Dettinger, 2011; Kløve et al., 2012; Candela et al., 2012; Tujchneider et al., 2012). It is likely that impacts on GDE will depend on changes in groundwater and surface water levels and that they will vary depending on location in the landscape and land use changes, as shown in Fig. 2. For local and intermediate scale systems (Fig. 2), it likely that the spatial extension of GDE will diminish with decreasing groundwater levels and surface water levels at increasing temperatures. Simulation results show that short-flow-path groundwater systems, such as those providing baseflow to many headwater streams, will likely have substantial changes in the timing of discharge in response changes in seasonality of recharge; whereas regional-scale aquifer systems with flow paths on the order of many tens of kilometers, in contrast, are much less affected by changes in seasonality of recharge (Waibel et al. 2013). These effects are uncertain, however, and depend on local hydrogeology. More studies should focus on changes in both groundwater and surface water, as well as their interactions with climate change.

For terrestrial and riparian vegetation, a shift in location, as well as in species composition, can occur. Changes in groundwater can change the wetland water balance, leading to lowered water level and reduced groundwater inflow. For example, Candela et al. (2012) use downscaled climate and groundwater model simulations to project a 17% reduction in recharge for the first quarter of the 21st century, most likely reducing groundwater discharge into wetlands of Majorca, Spain. Ecosystems in coastal regions can be severely negatively affected by salt water intrusion at future higher sea water levels and reduced groundwater inflow (Fig. 2). Drexler et al. (2013) observed a decrease of fen area of 10% to 16% from areal photos in Sierra Nevada due to changes in groundwater inflow in 50-80 years. Losses of biodiversity in GDEs of Santa Fe (Argentina) is related to decreasing discharge caused by increasing demand for groundwater and decreasing recharge rates (Tujchneider et al., 2012). Treidel et al. (2012) suggest that the future preservation of many wetlands and other GDE requires adaptive management actions that decrease groundwater abstraction for irrigated agriculture and that re-locate wells with detrimental effects on groundwater discharge to dependent ecosystems.

2.3 Climate change in GDE

To understand the impacts of climate change on ecosystems, we must understand all pressures and their potential impacts in the ecosystem and their potential feedbacks. All external pressures can change the ecosystem status, with changes typically becoming more severe with increasing pressure (Fig. 3). The response will be scale-dependent, which is a source of uncertainty as these responses are not well understood on smaller scales. Large-scale changes in hydrology are not always seen at the aquifer scale, where the local hydrogeology is dominant (Fig. 4). For groundwater systems, the natural variability in groundwater quantity and quality will depend on the size of the capture zone and the scale of the groundwater system (Toth, 1963; Fig. 5A). From an ecological point of view, ecosystems fed by local groundwater systems will show a more contrasted variation in temperature and nutrient concentrations than regional capture zones (Bertrand et al., 2012; Fig. 5B). As a consequence, it is likely that larger systems will be more resilient to climate change (Fig. 5C). In GDE, land use changes can alter abiotic conditions, with potentially rapid responses in biological communities and processes. Land use changes may even override changes caused by large-scale changes in climate, as reflected in regional hydrology (Fig. 4).

2.3.1 Climate change impacts in terrestrial ecosystems: Tree growth and distribution

Studies using GCMs indicate a warmer climate, with an increase in precipitation with increased CO_2, for the 21st century (Kundzewicz et al., 2007). Previous studies on the impacts of climate change on terrestrial ecosystems focus on changes caused by predicted future changes in precipitation, temperature and CO_2 on evapotranspiration, growth (assimilation) and distribution of vegetation, particularly young trees (Brolsma et al., 2010). The few modelling studies done so far with fully coupled vegetation-hydrology models show complex interactions and feedbacks from the combined effects of increased temperature, precipitation and CO_2. Increased CO_2 reduces stomatal conductance, which reduces transpiration and counteracts a potential increase in evapotranspiration caused by warming; Increased CO_2 also increases assimilation and plant growth, which results in higher biomass and transpiration. Increased temperature could also lengthen the growing season, although the impact of daylight is important (Saxe, 2001).

The main responses of GDE plants to modifications in groundwater resources and hydrology can be summarised in a conceptual scheme (Fig. 6). At larger scales, a shift in zonation is expected, with vegetation moving towards the poles and higher altitude. At the landscape scale, drought- and wet-tolerant species will shift uphill and downslope (Brolsma and Birkens, 2007). In the case of a general piezometric decrease, the effects on trees may be negligible or, conversely, it may provoke a total extinction of the original ecosystem (Naumburg et al., 2005). These effects depend on the interactions between biological (e.g. development or vegetative pause of the root system) and physical processes (soil water circulation, hydric potential differences between roots and leaves) and tree adaptation abilities (root development rate). If the tree cannot develop a deeper root system to keep in contact with the groundwater (rapid lowering), this can be temporarily compensated for by soil moisture (e.g. Meinzer et al., 1999). The resilience of ecosystems to resource abstraction is thus dependent on very local meteorological conditions (meteorological water supply) and the yield capacity of the soil layers (soil texture influencing soil water flow paths and accessibility to roots). In dry conditions, rainfall frequency may decrease but average rainfall depth may increase, resulting in increased recharge, which along with more deep-rooted vegetation can partly counterbalance the impacts of climate change (Liu, 2011). Simulations for a temperate (wet climate) hillslope, with reduced yearly rainfall and increased winter rainfall, show increased upslope recharge due to decreased upslope biomass and increased winter rainfall, resulting in increased groundwater levels and wetter conditions downslope and enlargement of wet-adapted vegetation cover (Brolsma et al., 2010). However, the impacts of other growth-limiting factors such as nutrients, pH, light and air humidity are not well known, rendering modelling results uncertain. For example, vegetation disease is probably the most obvious consequence of groundwater lowering (Scott et al., 1999). Water stress leads to reduced photosynthesis and transpiration, stomata closure (Leffler et al., 2000; Sperry et al., 2002; Cooper et al., 2003) and sometimes to xylem cavitation, especially in phreatophytes (Groeneveld et al., 1994). Xylem cavitation may lead in turn to the death and disappearance of some branches, initially the most distal ones. In snow and glacier-fed systems featuring high latitude and high altitude areas, a general piezometric level increase may occur due to climate change (e.g. Beniston, 2006). In such conditions, tree resiliency mostly depends on the capacity of the species to adapt to anoxic conditions. Even among phreatophytes, this capacity varies and is still difficult to evaluate (Ganskopp, 1986; Groeneveld, 1990). In the case of prolonged anoxia, some trees may lose their deepest roots and produce either shallower roots or roots adapted to anoxia (Groeneveld and Crowley, 1988).

2.3.2 Aquatic ecosystems: Lessons from surface waters

Climate change is expected to impose environmental regimes that will exceed the resilience capacity of most aquatic organisms (Poff et al., 2002). For example, shifts in the distributional ranges of freshwater taxa will be equally obvious to, or may even exceed, those predicted for most terrestrial organisms (Hickling et al., 2006). Given that inland waters are already among the most heavily human-impacted environments, climate change represents an additional and severe threat to freshwater ecosystems, altering their fundamental ecological processes and species distributions (Poff et al., 2002; Woodward et al., 2010).

Water temperature is an important environmental variable in freshwater ecosystems that directly influence organisms and ecosystem processes. Thermal regime regulates the growth and development of aquatic organisms and therefore directly affects species distributions and assemblage structure (Daufresne et al., 2004; Bertrand et al., 2012), as well as primary production and organic matter decomposition (e.g. Richardson, 1992). Temperature in lakes shows a correlation with air temperature and using this proxy Trumpickas et al. (2009) predicted a considerable increase for lake temperature of the great lakes of USA. For rivers, the increase in surface water temperature is mainly caused by reduced low flow and heat capacity for as shown by modelling for the United States, Europe, eastern China, and parts of southern Africa and Australia (Kane et al. 2013). Upto a 26 % increase is expected for seasonal rivers due to changes in low flow (Kane et al. 2013). Air temperature fluctuations are seen to a depth of 10-15 meters in groundwater, and a constant increase in soil mean temperature can be seen as an increase in mean groundwater temperature upto 4 ^{o}C in temperate climate in simulations using a considerable warming scenario (Taylor et al. 2009).

In addition to temperature, climate change also affects precipitation patterns and, consequently, the hydrological regime, and these effects can sometimes be even more detrimental to freshwater organisms than the direct effects of modified temperatures. Biota with low dispersal abilities and long generation times (K-strategists according to Mac Arthur and Wilson, 2001) are expected to be more common in permanently flowing springs, whereas biota with strong dispersal ability (R-strategists) will be favoured in non-permanent discharge habitats (e.g. Erman and Erman, 1995; Smith and Wood, 2002). Floods and droughts act as external disturbances, causing displacement of organisms and their resources, while indirect effects of discharge variation arise from interactions with the fluvial geomorphology and local stream habitat structure (Poff et al., 1997). Site-specific conditions such as current velocity and stability of sediments are likely to be modified by climatic-induced processes, which may alter the species distribution (Bertrand et al., 2012). Furthermore, the effects of temperature and discharge variability must be distinguished from land use-related environmental stressors such as eutrophication, acidification and sedimentation (e.g. Evans, 2005). Thus far, only a few attempts have been made to assess how changes in broad-scale climate factors will alter hydrological regimes and how these interactions will affect biological communities (Daufresne and Boët, 2007; Durance and Ormerod, 2007).

Freshwater springs are dependent on continuous discharge of groundwater and form subsurface-surface water and aquatic-terrestrial ecotones, which are important components of riverine landscape biodiversity (Ward and Tockner, 2001). Springs and spring-fed streams are considered physically stable environments that support stable biological communities (Barquin and Death, 2006). Given that the thermal regime of groundwater systems is less dependent on air temperature patterns than that of surface waters, the effects of altered air temperatures are likely to be less pronounced in springs and other GDE. However, climate change-induced modifications of recharge may have a profound impact on spring communities. Such changes may by reflected in decreased groundwater level in summer, but increased winter level and associated flooding can affect biological communities even more through changes in water chemistry caused by intensified links between aquatic and terrestrial environments (Green et al., 2011). In addition to intensity, the timing of disturbance events may be critical for biological communities. Freshwater organisms in boreal areas are evolutionarily adapted to a highly predictable seasonal flow regime, and alteration of the hydrological regime to more unpredictably occurring extreme flow events may result in serious problems for freshwater biota. Spring organisms, however, are reported to have remarkable resilience to human-induced disturbances. For example, Ilmonen et al. (2012) showed that invertebrate communities in springs affected by forestry approximately 30 years prior to sampling did not differ appreciably from those in non-modified reference springs.

3. Identification of research and data gaps

3.1 Impact of climate change on the variability of groundwater quantity and quality

For groundwater quantity, the fundamental issue is how recharge will be altered with climate change. The response of plant transpiration to increased CO_2, climate warming and changes in soil moisture and groundwater elevation must be understood and included in recharge models. More information is needed on groundwater recharge mechanisms, storage capacity and residence times in cold and alpine conditions (Singleton and Moran, 2010; Treidel et al., 2012). Most studies of climate change effects on surface hydrology in alpine, mountainous and snow-dominated regions do not explore subsurface hydrological responses (Green et al., 2011). The impacts of frost on soil hydraulic conductivity and recharge are large, but not fully understood (Okkonen and Kløve, 2010). These mechanisms need to be included in numerical models. The interactions between climate, groundwater and surface water must also be understood in order to predict changes in groundwater recharge (Okkonen et al., 2010).

Only few studies have addressed the potential effects of climate change on groundwater quality (Treidel et al., 2012). Even if climate change has no direct effect on local groundwater quality, changes in the volume of groundwater entering GDE may change the quality of the receiving waters (Earman and Dettinger, 2011). The limited number of studies conducted to date on groundwater quality have primarily addressed seawater intrusion into coastal aquifers, and some studies indicate that groundwater pumping is expected to have more of an effect than climate change and sea level rise on seawater intrusion in some coastal aquifers (Treidel et al., 2012; Ferguson and Gleeson, 2012). However, the effect of climate change on air temperature may influence groundwater temperatures and dissolved oxygen concentrations (Kløve et al., 2012; Haldorsen et al., 2012). This would have important implications for reaction rates and reduction-oxidation (redox) reactions that directly affect the nitrogen and carbon cycle in soil and groundwater, non-point source and point source contamination, and the fate of many groundwater contaminants. Climate-induced changes that alter biogeochemical processes may make groundwater less suitable for drinking (Figura et al., 2011). The quality of groundwater may be a limiting factor for some intended uses, such as drinking or irrigation, and for the long-term sustainability of groundwater resources worldwide (Gurdak et al., 2012), and therefore additional research is needed on regulation of groundwater quality. Changes in recharge rates and mechanisms may also increase the mobilisation of pesticides and other pollutants in the unsaturated zone and reduce groundwater quality (e.g. Gooddy et al., 2001; Johnson et al., 2001; Bloomfield et al., 2006; Sugita and Nakane, 2007). In some semiarid and arid regions, climate change may mobilise naturally occurring salts, such as nitrate and chloride porewater reserves, or enhance denitrification and removal of nitrate from the unsaturated zone prior to recharge (Gurdak et al., 2007). Stuart et al. (2011) noted that nitrate leaching to groundwater as a result of climate change is not sufficiently well understood to make useful predictions without additional monitoring data. Studies on natural soil and agricultural processes in the United Kingdom report a range of nitrate leaching rates from a slight increase to possibly high nitrate concentrations in groundwater by 2100 because of climate change (Stuart et al., 2011). In addition, a possible increase in surface water intrusion and flooding poses a risk to groundwater quality because of contamination by bacteria and organic matter from wetlands (Silander et al., 2006).

3.2 Impacts of climate change and groundwater change on ecosystems

Groundwater ecology as a scientific discipline is in its infancy (Larned 2012), and little is known about how climate change will affect GDE and their biota. Considering the importance of the ecosystem services provided by GDE to humankind, this lack of knowledge is unfortunate, as it hinders the adaptive management of GDE in the face of global environmental change. Any management decisions need to deal with potential conflicts between human resource use and GDE biodiversity. Many GDE support surprisingly high biodiversity and levels of endemism (Boulton et al., 2008), thus being of considerable conservation value. However, as they have suffered from human disturbance around the world, their unique biota is rapidly becoming threatened (Heino et al., 2006; Barquín and Scarsbrook, 2008; Boulton, 2009).

Changes in groundwater input can influence water quality in ecosystems in several, partly unknown, ways. A reduction in the average groundwater level tends to enhance soil aeration and thus organic matter oxidation. This can lead to nutritive enrichment, mostly through production of NO_3^- and PO_4^{3-}, which are generally the limiting nutrients in GDE (Wassen et al., 2005). In aerobic conditions, PO_4^{3-} may become toxic due to its fixation with the oxidised form of iron (Fe^{3+}) in the root zone (Boomer and Bedford, 2008). An increase in groundwater flux may result in waterlogged conditions, anoxic processes and associated fluxes of contaminants (Werner and Zedler, 2002; Olde Venterink et al., 2006). This may unbalance the nutritive equilibrium through the production of reduced species such as Fe^2+ (which might also release PO_4^{3-} bound to Fe^{3+}), Mn^{2+} (important nutrient but often at toxic concentrations in acid soils; El-Jaloual and Cox, 1998), or N_2 (which can only be taken up by roots in symbiosis with particular nitrogen-fixing bacteria).

If rainfall increases, acidification of the superficial zones of ecosystems may occur (Wassen et al., 1996; Grootjans et al., 2006; Bertrand et al., 2008). This process should be perceptible after several years of hydrological modifications (van Diggelen et al., 1996; van der Hoek and Sýkora, 2006; van Belle et al., 2006). Furthermore, acidification rate depends on organic acid production in situ (Kooijman and Paulissen, 2006), sulphur dynamics (Devito and Hill, 1997), and the acid-buffering ability of soils (van Bremen and Buurman, 2002).

Secondary hydrological changes in GDE due to altered water balance and groundwater levels have so far received little attention in climate change studies. The generation and maintenance of peat soils over time depend on hydrological conditions, and in recent studies of peatlands exposed to groundwater lowering, soil cracking, peat subsidence and secondary changes in water flow and storage patterns have been observed (Kværner and Snilsberg, 2008, 2011).

3.3 Impact of land use and water management

Groundwater is necessary for many human and natural systems and is a substantial economic resource in most developed and developing countries (Hiscock et al., 2012). The management of groundwater resources has many policy implications outside the immediate water sector (Ludwig and Moench, 2009). These include implications for agriculture and food security, energy, human health and safety (White and Falkland, 2012), and the conservation of groundwater-dependent ecosystems (Chaves et al., 2012; Kløve et al., 2012). Many policy and management decisions directly affect groundwater and (or) climate, which in turn further modifies groundwater resources. Examples of such policies are self-sufficency policy leading which in arid regions leads to cultivation of crops with high water requirement instead of crops with less water needs (Khalid 2013). In cold climate, agriculture is supported with European Union subsidence despite surface and groundwater being vulnerable due to high runoff and pristine water quality of that are easily contaminated. Also bioenergy crops expansion to marginal agricultural soils leads more pollution. Thus, policy decisions must carefully assess implications to the climate-water-society complex and the sustainability of groundwater resources (Treidel et al., 2012).

While most studies have addressed the response of recharge and groundwater/surface water interactions to climate change, quantifying groundwater withdrawals and use remains a difficult but necessary challenge (Treidel et al., 2012). Groundwater withdrawals for drinking water, agriculture and industry have a major effect on most groundwater resources and are a component of the groundwater budget that can be controlled directly by adaptive management practices and policy decisions. Treidel et al. (2012) conclude that additional scientific studies are needed in most aquifers of the world to quantify spatial and temporal patterns of groundwater discharge, withdrawals and uses in response to present and future climate.

3.4 Modelling gaps

The quantification of climate change impacts on groundwater systems and GDE can be explored by running groundwater models with future meteorological boundary conditions, which may be derived from future climate scenarios computed with climate models. However, there is a vast range of different GCMs with differing assumptions on ocean-atmosphere interaction, initial conditions and emission scenarios. Some of the GCMs are dynamically refined to regional climate models (RCMs). For various regions in particular, the predictions on changes in precipitation differ between different climate models. Furthermore, a statistical bias correction is sometimes performed based on station data (Themeßl et al., 2011), which draws on the idea that a good fit on historical data proves the adequacy of the climate model for prediction of future climate changes. However, this involves the assumption of stationarity and, in addition, it is questionable whether available data records are long enough to reliably represent natural climate variability and anthropogenically induced climate change (Kiem and Verdon-Kidd, 2011).

Typically, the spatial discretisation of climate models is in the order of tens of square kilometres. Thus, climate model results cannot reflect processes on a smaller spatial scale, which is particularly problematic in regions with strongly varying topography. Moreover, an aquifer or even more a GDE and its catchment might be of substantially smaller proportions. As input for hydrological models, time series of temperature and precipitation measurements are needed at the very least. While the resulting temperature data might be in line with observed data, this is rarely the case for precipitation. Seasonal patterns or shorter wet or dry periods are often poorly predicted in climate model outputs. This reduces the applicability of predicted precipitation time series for hydrological impact studies.

One way to get around this is the Delta approach, where a factor is added to observed meteorological variables to mimic the future time series affected by change (Taylor and Tindimugaya, 2012). This change factor can be derived from climate model outputs or can include stochastic or soft paleoclimatic components to account for climate variability components not included in observations. Goderniaux et al. (2011) describe an approach where they combine the change factor with a transient stochastic weather generator to address the uncertainty from different model structures and parameterisations in driving GCMs and RCMs.

Hydrological models run for climate change impact studies should be integrated with simultaneous consideration of processes in the unsaturated and saturated zones, overland and channel flow, soil-atmosphere interactions and, when relevant, the effects of snow and frozen soil. In general, climate change impacts on groundwater systems and GDEs are indirect consequences of changes in precipitation, evapotranspiration and surface runoff. Groundwater systems then face altered patterns and magnitudes of recharge from water that has moved through the unsaturated zone and/or surface water levels that lead to different exchange conditions. Depending on the focus of a study, emphasis can be placed on modelling sub-processes and treating the other water balance components as boundary conditions with or without feedback. Thus, a decision has to be made on the level of hydrological model complexity justified given the existing hydrological data for model calibration.

To model climate change impacts on GDE, hydraulic aspects (e.g. extent of capture zone or critical groundwater level conditions) need to be complemented by data on biological and geochemical processes. For example, groundwater temperature is an important driver of all biological activities, which in turn might influence water quality in multiple ways. Thus, modelling processes within GDEs may become highly nonlinear. Finally, future scenarios also need to include transient assumptions about land use change, socioeconomic developments and, in particular, water abstraction, as these affect groundwater systems and thus GDE.

Overall, a consistent strategy has to be developed to link relevant GDE processes to the surrounding groundwater system, considering strongly diverging temporal and spatial application scales. One crucial component is to disentangle the sequence of nonlinear feedbacks between hydraulic and biological processes. Moreover, the prospects for numerical modelling of the most relevant GDE problems need to be clearly identified and further described. If numerical models are to be used as a tool to provide the link between climate change impacts on GDE, a number of challenges must be met in a consistent manner. In this context, the propagation of uncertainty from the climate model outputs through hydrological models must be taken into account.

4. New approaches

4.1 Integrated multidisciplinary monitoring of groundwater and GDE and new methods in the field of ecohydrology

In the future, groundwater systems and use of these resources need to be studied in a multidisciplinary way in order to better understand the interaction between processes on the soil surface related to hydrology and land use, and the relationship between groundwater and ecosystems. Modelling is needed to link the complex natural processes to groundwater extraction, land use and management effects. For such integrated studies, data are required on land use changes and water extraction, groundwater-dependent ecosystems, and groundwater-surface water interactions. We also need to understand how ecosystems depend on hydrological drivers and how they respond to predicted changes in hydrology (Fig. 4, 6). Monitoring data should include information on geomorphology, ecology and hydrology. Such data are usually unavailable in national monitoring efforts, which generally focus on groundwater levels or river discharges. Smaller-scale monitoring is needed for systems relevant for future ecosystem protection and legislation (e.g. NATURA 2000 ecosystems).

Several new methods could be introduced in GDE research. Plant responses to water stress could be followed on short and longer time scales (Table 1) to verify the predicted changes in plants. As ecosystem responses will vary spatially, the monitoring networks need to be spatially distributed within a catchment. In many cases groundwater flowpaths are not well known, and tracer methodologies could be used to obtain this information. New tracers such as Nobel gases are also available for assessing changes in climate. It will be important to carry out climate change assessments that consider all reasons for climate change, including climate variability and the impact of urbanisation on climate. For this, long series of records are crucial, as climate (and recharge) can oscillate with wavelengths of more than 50 years. However, many long-term weather monitoring stations are affected by the urban heat island effect (Hamdi, 2010) and temperature data must be used with care.

4.2 Modelling as a future management tool

Given the numerous challenges in combining climate model results and groundwater and GDE model applications, and the varying importance of different processes in different and complex GW-GDE systems, a promising approach is to use elaborated conceptual models to pinpoint the most important steps and drivers and set up a problem-specific model chain. Alternatively, the fully integrated hydrological modelling approach can be pursued (e.g. Goderniaux et al., 2011), but this requires either powerful computers (due to long CPU response times) or modelling compromises within the spatial model resolution.

The conceptual model/model chain procedure may involve establishment of a list of indicators that describe GDE vulnerability, followed by identification of the linkages between these indicators and climate change-related impacts. The latter aspects should also include different levels of model proficiency, e.g. numerical modelling of water temperature is understood quite well, whereas other (in particular GDE-related) processes and their temporal and spatial variations are currently less well described.

Furthermore, the role of the unsaturated zone in transferring climate change signals to groundwater systems and GDE needs to be better clarified (Treidel et al., 2012). Goderniaux et al. (2011) state that the unsaturated zone smooths groundwater recharge flux variations so that groundwater levels become insensitive to seasonal fluctuations in the weather, but rather reflect multiannual fluctuations. However, Ng et al. (2010) found that changes in average precipitation are amplified by changes in average recharge, leading to nonlinearities due to the temporal distribution of precipitation change.

Uncertainty in hydrological modelling should not be overlooked, although it has been shown to be smaller than that in climate modelling. It is dependent on the kind of hydrological system investigated and the specific hydrological models used. Along that line, Ng et al. (2010) present a combined probabilistic approach that explicitly accounts for uncertainties in meteorological forcing by applying a stochastic weather generator and soil and vegetation properties by generating realisations that are conditioned on soil moisture and soil water chloride observations.

4.3 Groundwater indicators in future status and risk assessment

Indicators are widely seen as a means of bridging the gap between scientific research and political needs. Political organisations often recommend the development of indicators and teams of experts and academics carry out this task (Hinkel, 2011). Indicators represent a tool for ecological assessment, while indices (e.g. multimetric indices, composite indices) are highly integrative, allowing broad-scale questions to be addressed using a few carefully selected parameters (Innis et al., 2000). The use of indicators requires a comprehensive understanding of the structural components of an ecosystem and the interactions between, and responses to, various stresses of these components, as well as their spatial and temporal variability. Rigorous testing and validation are necessary to establish such an understanding (Innis et al., 2000). Although the messages of different indicators are complex, careful assessment may allow policy makers to identify aspects and stressors of ecosystems requiring improvements of existing policies.

Despite financial and operational constraints, there is a need for monitoring and evaluation of groundwater status. To this end, chemical and sometimes microbial indicators have been analysed in many GDE. For example, groundwater pollution due to agricultural activities is shown by NO_3-N and PO_4-P content, which may result in failure to comply with national and EU water quality regulations. Heavy metals and metalloids, as well as hydrocarbons and pesticides, must meet the standard values, being generally under the detection limits (Cruz et al., 2009).

Groundwater has been selected as one of the terrestrial Essential Climatic Variables (ECVs) by GTOS (Global Terrestrial Observation System). Several critical variables have been considered under the heading “groundwater”: groundwater level, groundwater recharge and discharge, and water quality. Of these, groundwater level is a direct indicator of groundwater supply and withdrawal rates, while the GTOS Panel on Climate recognises that groundwater discharge and recharge are critical indicators of climate change. Despite the priority given to chemical status, biological, physical and radiological factors are also considered when assessing groundwater quality.

Current indicator systems (e.g. groundwater indicators or water quality indicators according to the EC Water Framework Directive, WFD) need to be strengthened and applied more consistently and universally. However, much more remains to be done to produce or to aggregate indicators relevant to assessing the vulnerability of GDE. All existing indices of vulnerability to climate change show substantial conceptual, methodological and empirical weaknesses, including lack of focus, lack of a sound conceptual framework, methodological flaws, high sensitivity to alternative methods for data aggregation and limited data availability (Füssel, 2009).

A common approach to GDE assessment requires scientific consensus. Improvements in data availability can be realised through better organisation and interoperability of databases. Many indicators have been developed under national and EU programmes (in projects providing scientific support for WFD or GWD implementation in different EU countries), but have not reached their potential.

The selection of elements in an indicator set is critical. An appropriate indicator must incorporate several spatial and temporal scales and multiple environmental factors (physical, chemical and biotic) to provide robust results. The use of carefully selected indicators maximises the amount and quality of information about the ecological integrity of a system, while minimising the time and expense involved.

Whether, and which, indicators are useful for vulnerability assessment or climate change adaptation policy remains an open question. Before this question can be addressed, goals and targets of specific indicators in climate-relevant policy fields need to be evaluated and refined frequently (Hinkel, 2011).

4.4 Integrated management of groundwater and GDE

The accelerating trend for withdrawal and use of groundwater over recent decades has been essential in the development of many regions of the world, producing large social and economic benefits through the provision of low-cost, drought-reliable and high-quality water supplies. Many regions have large groundwater-dependent economies. This fast expansion has been referred to as the “silent revolution”, in the sense that in many regions it has followed a bottom-up approach, driven by the personal initiative of millions of individual farmers in pursuit of the significant short-term benefits usually provided by groundwater (Llamas and Martinez-Santos, 2005). Such developments are often uncontrolled and not incorporated into a comprehensive land and water management plan at the basin scale, resulting in overexploitation and groundwater degradation and drainage impacts on GDE.

Sustainable development of groundwater is a major challenge that is expected to be exacerbated by the potential impact of climate change. The expected increase in the frequency and intensity of dry periods might lead to increased and unsustainable abstraction of groundwater resources (Green et al., 2011). Groundwater is bound to play a decisive role in adapting water resource management to climate change. Forward planning for adaptation of groundwater management to global (climate and land-use) change is essential in order to develop sustainable practices to cope with the impacts of future climate change. This adaptation should consider the local context, the dominant drivers and their projected impact on groundwater resources in the future (World Bank, 2009).

Integrated Water Resources Management (IWRM) requires the coordinated development of water, land and related resources in order to maximise the resultant economic and social welfare in an equitable manner, without compromising the sustainability of vital ecosystems (GWP, 2000). Groundwater management within the general IWRM framework requires integration of an appropriate policy and regulatory framework, institutional arrangements, social participation and economic instruments to be fully effective (Foster and Ait-Kadi, 2012). In future IWRM, GDE have to be seen as an integral part of groundwater resources.

Some of the challenges for the adaptation of integrated groundwater management to climate change are:

- Appropriate institutional/regulatory framework for groundwater appropriation and use. Groundwater management is best carried out through the collaborative efforts of a regulatory agency and aquifer management organisation involving representatives of local associations of water users and other stakeholder groups (Garduno and Foster 2010). The decisive role of collective action in groundwater governance is being increasingly recognised (Lopez-Gunn, 2003; Lopez-Gunn and Martinez-Cortina, 2006).
- Economic management instruments. Groundwater and GDE services are often undervalued (economic externalities and groundwater economic value are scarcely recognised), which has often led to inefficient patterns of groundwater use, resulting in overexploitation and pollution problems. With increasing water scarcity, the economic value of groundwater is rising. It is essential to study the total economic value of the resource in order to assess the net benefits of management actions (NRC, 1997). There is an array of economic instruments that can provide the appropriate incentives for efficient groundwater extraction and management. Although the economic instruments to manage surface water and groundwater are similar, they are not identical due to certain special characteristics associated with groundwater, including the relatively high cost and complexity of assessing groundwater, the highly decentralised nature of resource use and the high monitoring costs, and the long time-lags and near irreversibility of most aquifer contamination. The selection and use of a particular economic instrument will depend on hydrological, economic, social and political considerations. Abstraction charges provide direct incentives for water saving. There are

two alternatives: pricing through resource abstraction fees or indirect pricing through increasing energy tariffs. Water markets have been advocated to improve resource management, especially with regard to more efficient water use and allocation within and between sectors. Groundwater banks also offer new perspectives for water management in drought conditions (Howitt, 2004; Pulido-Velazquez et al., 2004).

- Integrated conjunctive management of surface and groundwater and GDE. Conjunctive use (CU) is the coordinated management of surface and groundwater resources, taking advantage of their complementary properties. Jointly operating all manageable water resources in a river basin or region can increase the yield, efficiency, supply reliability and cost-effectiveness of a system. CU not only refers to artificial recharge practices, but to a broad range of options (including alternating use of surface water and groundwater, managed stream/aquifer interaction, etc.) which can occur in different temporal patterns and within active or passive management, according to the region's development status and planning objectives (Pulido-Velazquez et al., 2003; Sahuquillo and Lluria, 2003). Tanaka et al. (2006) used a state-wide hydroeconomic optimisation model to analyse water supply in California under climate change scenarios for the year 2100. They found that the system can adapt to significant changes in climate and land use through major changes in the operation of the large groundwater storage capacity and conjunctive use management, significant transfers of water among users (water markets) and the adoption of new technologies.
- Land use regulations to protect groundwater resources and GDE. Management of groundwater quality requires the protection of aquifers and groundwater from ingress of pollutants and also the remediation/treatment of polluted resources. However, treatment

of polluted groundwater is complex, expensive and often only partially successful, and it may take many years before groundwater quality is restored. The protection of aquifers against pollution requires constrained land use, effluent discharge and waste disposal practices. One widely used strategy has been the establishment of groundwater protection zones. Improved coordination among the governments and agencies engaged in land-water management is needed.

- Building adaptive capacity for groundwater and GDE management. This requires undertaking research to better understand the risks faced and the system's vulnerability to climate change, and to improve or extend the range of adaptations. Education and communication programmes could be developed to improve stakeholders' and communities' understanding of risks and management responses and empower groups to develop new adaptations or apply existing adaptations more effectively or extensively (World Bank, 2009).

5. Conclusions

Climate processes influence groundwater patterns in a complex way, with a number of direct and indirect effects. Future recharge can reflect normal climate variability, human-induced warming and local land and water management. These changes may counteract or amplify each other. The influence of past climate must be studied over a long time scale (e.g. 70-100 years) to reveal natural variability.

Climatic variables influence hydrological processes, so any change in precipitation, evapotranspiration, snow accumulation and snow melt will influence recharge and groundwater formation. It is not fully understood how evapotranspiration changes with increased temperature, CO_2 and alterations in rainfall patterns. In cold climates, the response in terms of snow melt, frost and winter hydrology and recharge needs to be better quantified. Changes in groundwater interaction with surface water are important for groundwater recharge and must be better understood. Studies on groundwater are needed from regions with different socio-economic development, land uses and hydrogeological settings.

Climate model outputs have been used to assess changes in hydrology. So far these climate models do not estimate changes in climate at a scale useful for studies of changes in small groundwater deposits and in GDE. Future scenarios of precipitation, wind speed and radiation are all highly uncertain compared with the predicted changes in temperature. Changes in land use patterns, irrigation, vegetation cover and water use are not well understood and documented. The present state of climate response modelling must therefore include a proper assessment of uncertainties in input variables and predicted changes in water consumption and land use. As drought is a major threat to GDE, it is important that impact studies include an assessment of drought on the water supply. Typically, drought effects can be seen only after several years of drought, when groundwater levels are lowered.

The impacts on ecosystems will vary depending on the type of ecosystem, amount of water input and changes in water input. For some groundwater systems, the changes in temperature may be smaller than for surface waters. The expected change will depend on the existing quantitative and qualitative stresses on these ecosystems. Conceptual and numerical models can be useful for predicting future changes in ecosystems, but the effect response in these systems is not well known. Many processes are highly non-linear and should be included in numerical models. In major aquifers used for drinking water production, data on groundwater levels are available, but for smaller systems information is scarce. In most cases, monitoring on ecosystem scale is lacking in national monitoring programmes. Generalising the effects on groundwater quantity and quality for any particular region is challenging and subject to considerable uncertainty.

Groundwater is important for both economies and ecosystems. In future, groundwater should be managed in a multidisciplinary way in order to provide efficient solutions. Numerical models will be essential for understanding the complex interactions in GDE, while simple indicators will be helpful for monitoring the results of policy practices in GDE. As the study of GDE is new, more research and development is needed. This should include the development of scientific methodologies and national monitoring activities.

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