Chapter 10 groundwater assessment answers

This chapter begins with a brief discussion on the three main classes of groundwater-dependent ecosystems (GDEs), namely: (I) GDEs located in groundwater (e.B karst; Stygofauna); (II) GDEs requiring surface expression of groundwater (e.B. springs, wetlands); and (III) GDIt depends on the availability of groundwater on the underground surface within the rooting depth of vegetation (e.B forests, adjacent forests). We will then discuss a set of techniques available to identify the location of GDEs in a landscape, with a focus on Class III GDEs and a secondary focus of Class II GDEs. These techniques include inferential methods using hydrological, geochemical and geomorphological indicators, biotic assemblages, historical documentation and remote sensing methods. The techniques available to quantify groundwater use by GDE are briefly described, including the use of simple modelling tools, remote sensing methods and complex modelling applications. This chapter also describes the current threats to the persistence of GDEs around the world. It describes the natural hydrological attributes relevant to GDEs and the processes that lead to disturbances of natural hydrological properties as a result of human activities (e.B. groundwater extraction). Two case studies, (1) Class III: terrestrial vegetation and (2) Class II: sources, are discussed in relation to these issues. Water Table Leaf Area Index Unconfined Aquifer Groundwater Deep Groundwater Abstraction These keywords were added by machine and not by the authors. This process is experimental and the keywords can be updated as the learning algorithm improves. In order to manage groundwater in a truly integrated manner, account must be taken of the interaction of groundwater with ecology. Groundwater interacts with several classes of biomes, including stygofauna of aquifers, rivers based on basic currents (the discharge of groundwater into rivers) and terrestrial ecosystems. Management plans that do not include such consideration are likely to have a negative impact on these groundwater-dependent ecosystems. In this chapter, we focus on the links between ecology and groundwater availability, not groundwater resources and human demand. The reason for this is that we believe that the environmental allocations of groundwater have generally received less attention than the allocations to human requirements, and because we have four important knowledge gaps for the sustainable management of environmental allocations of Identify. These are:1. How do we know where a GDE is in the landscape? If we do not know where they are, we cannot manage them and distribute groundwater resources adequately. 2. How much groundwater is used by a GDE? If we do not know how much groundwater is being consumed, we cannot allocate an adequate amount of the resource. 3. What are the to the GDEs? Only by understanding the threats to companies can we ensure their sustainable management. 4. What are likely reactions to groundwater extraction? Without knowing what to measure, we cannot regulate groundwater extraction in a way that does not have a negative impact on businesses. Hatton and Evans (1998) were perhaps the first to systematically categorize THE GDEs. They identified five classes of ecosystem dependence on groundwater:1. Ecosystems that are completely dependent on groundwater; or obligatory GDEs. In these communities, only minor changes in the availability or quality of groundwater lead to a total loss of the current ecosystem structure and function. Examples of fully dependent ecosystems include the hill spring systems of the Great Artesian Basin in eastern Australia, karstic groundwater ecosystems in Western Australia, and the adjacent vegetation along streams in Central Australia. 2. Ecosystems that are highly dependent on groundwater. In these communities, small to moderate changes in groundwater availability lead to significant changes in ecosystem structure and function. Examples of highly dependent ecosystems in Australia include: Melaleuca swamp forests and forests of tropical northern Australia, base-flow-dependent ecosystems of temperate Australia and the wet lands of the Swan Coastal Plain. 3. Ecosystems with proportional dependence on groundwater. Such ecosystems do not have the threshold-type responses of (1) and (2) above. As the availability or quality of groundwater changes, there is a proportional response to the structure and function and distribution of the ecosystem. Examples are the basic flow and permanent lake ecosystems. 4. Ecosystems that are opportunistic users of groundwater. In these ecosystems, groundwater sometimes plays an important role in their water balance and dependence is not mandatory (so-called optional dependence). Examples of opportunistic ecosystems include swamp forests of coastal floodplains on the edge of the southeastern highlands and Jarrah forests and banksia forests in Western Australia. 5. Ecosystems that seem to be dependent on groundwater but are in fact fully fed by surface water streams or are only dependent on surface water flows. Examples of this species are seasonal floodplain lakes at small streams in northern Australia and end basin lakes in the Central Lowlands. There are two major problems with this classification system. First, determining the degree of dependency is difficult and requires many years of studying a site. The finding that an ecosystem is only an opportunistic user of the may require a decade of waiting before a drought occurs and dependence on groundwater arises. Secondly, it is extremely difficult and time-consuming to determine the existence or absence of a threshold response. Consequently, a simplified classification system was proposed by Eamus et al. Proposed. Aquifers and cave ecosystems where stygofauna live. This class also includes the hyporheic zones of rivers and floodplains. (II) Ecosystems that depend on the surface expression of groundwater. These include river flows, streams and wetlands, springs and estuary grasses. (III) Ecosystems that rely on the underground presence of groundwater within the rooting depth of the ecosystem (usually via the capillary edge). The application of this simple classification system helps managers identify the correct techniques for evaluating the GDE structure, function and management regime (Eamus et al. 2006). This classification scheme was recently adopted in the Australian National Atlas of Groundwater Dependent Ecosystems.Springs in geomorphic environments that are much more complex than most wetlands emerging from hills, cliffs and under other waters. In addition to their complex development environment, sources often support a variety of microhabitats that are not observed in wetlands. The sphere into which the aquifers flow was first described by Meinzer (1923) and then simplified by Hynes (1970) into three classes: Rheocren (channel formation), limnocren (pool generation) and Helocren (wet meadow formation). Springer et al. (2008) and Springer and Stevens (2009) reviewed the literature and extended this historical scheme by 12 areas of discharge of terrestrial sources, including: (1) sources that arise in caves, (2) Sources of exposure, (3) artesian wells, (4) geysers, (5) Gushets, (6) Contact hanging gardens, (7) Helocrene wet meadows, (8) slope springs, (9) Hypocrene buried springs, (10) limnocrene surficial lentic pools , (11) mineralized hills and (12) rheocrene lotilot canal. This classification provides a more accurate lexicon that describes the groundwater generation function in terms of ecosystem configuration and distribution. Geomorphological variations between the 12 terrestrial source types Von Springer and Stevens (2009) lead to predictable fluctuations in the vegetation of spring, habitat structure, plant and fauna diversity and the structure and function of the ecosystem (Griffiths et al. 2008). For example, Helocrene springs are usually dominated by wetland graminoids and shrub species, with little canopy cover by trees. Many mountainside springs typically occupy a position on the landscape where groundwater drainage is due to low discharge rates, which extract fine-grained sediments or sappen groundwater to create source-dependent spring water theatres for canals (Laity and Malin 1985; Meinzer 1923). The persistence of GDEs is based on appropriate groundwater attributes. The these attributes are essential as this can help to set targets for groundwater management and monitoring strategies (Kreamer et al. 2014). In general, the following Attributes are important for GDEs (Clifton and Evans 2001):1. Groundwater depth, for ungifted aquifers; 2. groundwater pressure – hydraulic head and its expression in aquifer, for constricted aquifers; 3. Groundwater flow - flow rate and volume of groundwater supply; direction of flow; 4. Groundwater quality, including groundwater salt content, acidity and concentrations of nutrients and pollutants. The significance of these characteristics for the GDE is summarised in Fig. 13.1. Groundwater depth (from the land surface) is perhaps one of the most important groundwater attributes for GDEs (Eamus et al. 2006). This is particularly true for terrestrial ecosystems that depend on the underground supply of groundwater. The depth to the groundwater directly determines the availability of groundwater for vegetation, with particular regard to the distance between the capillary edge above the groundwater level and the plant roots. Increased groundwater depth can lead to reduced plant growth, mortality and a change in species composition (Shafroth et al. 2000). Lowering a groundwater level can also lead to the loss of habitat for cave and groundwater ecosystems (Boulton et al. 2003; Heitmuller and Reece 2007). On the other hand, a rising groundwater level can disadvantage species that are susceptible to water logging and lead to the succession of various plant communities (Naumburg et al. 2005). Changes in water level depth in conjunction with other environmental factors can also lead to groundwater contamination. For example, lowering a groundwater level under acidic sulphate soils leads to the oxidation of pyrite and subsequent acidification of the shallow aquifer (Ritsema et al. 1992; Nath et al. 2013).Fig. 13.1 Importance of the groundwater regime (deep groundwater and groundwater pressure and flow) and quality on different GdEs classes and the anthropogenic threats of groundwater flow is important for Classes II and III GDEs, as it maintains water absorption through vegetation (Shafroth et al. 2000). Lower groundwater pressure and flow lead to lower groundwater drainage and consequently to a lower availability of surface water for wetlands and GDEs that depend on groundwater currents and sources (Zektser et al. 2005). In estuaries or coastal areas, a reduced groundwater flow leads to the penetration of seawater and contamination of coastal freshwater aquifers (Jayasekera et al. 2011; Lambrakis 1998), which reduces groundwater quality. For cave and groundwater ecosystems, an appropriate groundwater flow is important to improve the supply of organic matter and oxygen et al. 2005) to stygofauna contained in these systems. Groundwater quality is essential for all types of GdEs in order to maintain an appropriate chemical composition in the water supply and/or in the living environment. In some areas, groundwater is hydrochemically stratified. A disturbance of the stratification can cause the chemical composition to be unsuitable for the associated aquifer. Is. Groundwater pressure and flow naturally fluctuate. In unlimited aquifers, of course, short-term fluctuations occur in response to the time-changing absorption of water by vegetation; whereas longer-term fluctuations often reflect time-changing groundwater charging as a result of wet and dry cycle cycles. GDEs developed in naturally highly volatile areas (e.B. areas with strong climatic seasonality) have generally adapted to the fluctuations of the groundwater regime and may therefore be more resistant to changes in the groundwater regime than those developed from areas with more consistent regimes. In the Howard River basin in the Northern Territory of Australia, for example, the natural intraannual variation of groundwater depth is about 8 m (Cook et al. 1998). This large variation (caused by a combination of wet and dry-time fluctuations in precipitation, the lateral underground flow of groundwater to the Howard River and the evapotranspirational discharge) is taken into account by changes in the Landscape Leaf Area Index (LAI) and the root depth. These groundwater properties can be altered due to human activities. The current threats to the persistence of GDEs, including processes that lead to disturbances of natural hydrological properties as a result of human activities (e..B groundwater abstraction), are described in section 13.5. However, identifying their location in a landscape is often difficult, time-consuming and therefore expensive, and always requires a high level of technical know-how. This section explains a number of techniques that can be supported. Early assessments of groundwater dependence were often based on conclusions (Clifton and Evans 2001; Eamus et al. 2006). Therefore, responses affirming one or more of the following points may be seen as support for the hypothesis that at least some species in an ecosystem use groundwater.1. Does a stream/river flow all year round, despite long periods of low or zero rainfall (and thus zero surface flows)? 2. Do the salinity levels of estuary systems without surface water inputs fall below the dessal content of seawater? 3. Does the total flow in a river increase downstream if there is no inflow or surface flow? 4. Are water levels maintained in a wetland during prolonged dry periods? 5. Is groundwater discharged to the surface for a longer period of time each year? If such a resource exists, evolution will ensure that some species will use it. 6. If the which is associated with the surface discharge of groundwater (in terms of species composition, phenological pattern, leaf area index or vegetation structure), by the vegetation in the vicinity, but which does not access this groundwater? 7. If the annual transpiration rate is transpiration with a suspected GDE significantly greater than the annual rainfall on the site and the location is not a run-on site? 8. Are the relationships between plant water (especially before dawn and midday water potentials and transpiration rates) to less water stress (water potentials closer to zero; Transpiration rate greater) than vegetation nearby, but not access to groundwater on the surface? The best time to judge this is during rainless periods. 9. Does the water balance of a site indicate that the sum of the water consumption plus runoff plus the outflow plus deep run-off is significantly greater than the annual precipitation plus start-up? 10. Is the occasional (or usual) groundwater release on the surface associated with important stages of vegetation development (e.B. flowering, germination, seedling) 11. Do groundwater and hydrological modelling indicate that groundwater either discharges to the surface or is within the probable rooting depth of vegetation? 12. Is the groundwater or capillary edge above the groundwater level in the rooting depth of one of the vegetation? 13. Does some of the vegetation remain green and physiologically active during prolonged dry periods of the year (mainly transpiring and fixing carbon, although trunk diameter growth or leaf growth are also good indicators)? 14. Do some ecosystems in a small region (and thus in an area with the same annual rainfall, temperature and vapour pressure deficit) and in an area that does not have access to up or stream or river water show major seasonal changes in the leaf area index, while others do not? 15. Are seasonal changes in groundwater depth greater than the sum of lateral currents and percolation to depth (i.e. vegetation is a significant drain for groundwater); (Cook et al. 1998)) If the error terms used to estimate lateral flow and percolation to depth are similar in size or size to the vegetation water rate, this method may not be appropriate. Affirmative responses to one or more of these questions lead to the conclusion that the system is a GDE. However, this does not include information on the type of dependence (obligate or optional) or on the groundwater regime (e.B. time of groundwater availability, volume used, surface expression location, pressure of the aquifer required to support the surface discharge of groundwater) necessary to support the ecosystem. In shallow, unlimited aquifers, where the roots of the vegetation directly access the groundwater table usually via the capillary zone), it is possible to recognize the daily pattern of vegetation water use in subdaily fluctuations in groundwater (Gribovszki et al. 2010). Although the air pressure or temperature is meeting (which lead to changes in the volume of water, and condensation) and inputs of precipitation can cause changes in groundwater depth, it is still possible to identify and sometimes quantify the extraction of groundwater by transpiration (Gribovszki et al. 2010). White may have been the first to make subday changes in groundwater depth to quantify the transpiration use of groundwater in 1932 (White 1932). An idealized representation of the deil pattern of groundwater depth in a shallow, unlimited aquifer is shown in Fig. 13.2.Fig. 13.2 A schematic representation of changes in the depth-to-groundwater by vegetation transpiration The solid continuous oscillating curve represents the cycle of groundwater decline (due to ET) during the day, followed by the recovery of the water level when ET returns to zero at night (provided that the nighttime transpiration). The dashed straight line (with inclination = r) is used to estimate the amount of water passed through vegetation in 24 h (0:00 h to 0:00 24 h later; indicated by the horizontal dotted arrow) through the vegetation. This is represented by the vertical arrow, which represents the difference between the groundwater depth that would have occurred without vegetation water use and the observed groundwater depth. By applying this method, it is possible to identify the location of a GDE and thus to enable the first step in the management of both groundwater and dependent ecology. Lautz (2008) provides a detailed analysis of groundwater use using the White method for analyzing subday changes in groundwater depth. It shows that spatial differences in groundwater use can be explained by differences in the type of vegetation (wetland and grassland) and the specific yield of the aquifer. As expected, the ratio of groundwater-soil-water extraction increased as soil moisture levels decreased depending on the time since the rain. Geochemical studies, in particular isotope-based analyses of water samples, can be used to distinguish groundwater sources from other water sources (e.B. atmospheric, soil or stream water sources) and to identify the time of source and groundwater residence time (e.B. Winograd et al. 1998; Monroe et al. 2005). Mineral deposits and helium isotope expression due to groundwater discharge may also indicate groundwater discharge (Crossey and Karlstrom 2012), as the presence of certain plant species and invertebrates confirms. In the case of basic flow systems (i.e. rivers and streams showing significant currents in periods of zero surface or lateral currents), measurements of the Identify and quantify magnesium or radon concentrations of river and groundwater water supply the amount and timing of groundwater inflows into the river (Cook et al. 2003). Stable isotopes (such as deuterium (2H) and 18O) can also be used for these systems, as well as artificial marking with tracers such as lithium. When tracers enter groundwater Inclusion in the vegetation is usually conclusive proof that the access occurs through this vegetation. However, the presence of a tracer in a flat-rooted species can occur when neighboring deep-rooted species have hydraulic lifts and the flat-rooted plants then harvest this water (Caldwell et al. 1998). If a close agreement is established between the composition of the groundwater isotope and the xylemisotope composition, we can conclude that the vegetation uses groundwater. Direct evidence that vegetation uses groundwater can be used by comparing the stable isotope composition of groundwater, soil water, surface water (if relevant) and vegetation xylem water (Kray et al. 2012; Lamontagne et al. 2005; O'Grady et al. 2006; Thorburn et al. 1993; Zencich et al. 2002; Spaek and Prokow 2011). A direct comparison of periodic measurements was carried out by Hunt et al. (1996), which showed that time integration by measuring isotope composition was a valuable tool that provides insights that are not available from non-isotope techniques. If a sufficient variation of the isotope composition occurs between these sources, it is possible to identify the only or most dominant water source used by different species at different times of the year (Zencich et al. 2002). An example of the use of 18O isotope analyses of xylem water, soil water and groundwater is shown in Fig. 13.3.Fig. 13.3 as an example of the use of 18O analyses of xylem water, soil water and groundwater in a study of several species growing in northern Yucatan (Mexico). The 18O content of the soil decreases with the depth through the soil profile and finally the groundwater is reached (at 3 m; brown square). The Xylem 18O content of three species (Ficus spp. green triangle; Spondias spp. purple circle; and Talisia spp. black diamond) is also presented. Ficus was least dependent on groundwater, while Talisia was most dependent (drawn from Querejeta et al. 2007) mixed member models that allow an assessment of the relative contribution of several water sources to the water absorbed by roots (Phillips and Gregg 2003; Kolb et al. 1997). Thus, the use of stable isotopes can provide information on spatial and temporal fluctuations in groundwater dependence and the rates of use of groundwater within and between species and ecosystems. The application of stable isotope analyses to quantify the rate of water use is set out in section 13.4.4.The various source areas of the discharge (source types) produce characteristic geomorphology and soils indicating groundwater dependence Travertine mounds forming springs and hanging gardens are obvious examples of distinctive GDE geomorphology. The aerial analysis of spring channels is often used for the planning of spring restoration projects (e.B. Ramstead et al. 2012). Since the geometry of the spring channels is often irregular and not et al. 2008) is the detection of such a channel configuration an indication of spring flow control and not of surface flow control (Springer et al. 2008). In hypocrenes, excavating shallow wells or ground pits/cores can help identify groundwater sources, and among other source types, discrete particle size arrays can result from the consistency of the discharge from certain source types. Geochemical deposits such as travertine often indicate groundwater dependence in hill-forming, hypocrene, geyser and other sources. Montezuma Well, the massive travertine along the Colorado River and the collapsed travertine hills in the Tierra Amarilla region in northern New Mexico, are examples of source-related land forms (Crossey and Karlstrom 2012; Johnson et al. 2011; Newell et al. 2005). In dry regions, organic soil development on densprings can be extensive, unmistakable and can be called with radiocarbon techniques. Groundwater-dependent peat deposits can be massive and persist for thousands of years (e..B. Haynes 2008). In the Upper Carson Slough in Ash Meadows, a tributary of the upper Amargosa River basin in southern Nevada (McCracken 1992), peat-thick deposits of more than 2 m thick were commercially mined. If the geomorphology of the site has not been substantially altered, these characteristic land forms and soil characteristics generated by groundwater can remain identifiable, even if the aquifer has been largely drained. All over the world, both in terrestrial and subaqueous environments, springs are widely known to support unique aquatic and wetland plant species and unique assemblages. In one of hundreds of examples of unusual source-dependent plant species, Spaek and Prokov (2011) reported a highly isolated population of source-dependent Batrachium baudotii (Ranunculaceae) in a karst spring in central Poland. The few remaining hill springs between Guildford and Muchea in Western Australia support limited wetland graminoid plant relationships with Cyperaceae, Juncaceae and Restionaceae, as well as flooded chewing gum (Eucalyptus rudis) and Brackenfern (Pteridium esculentum) (Blyth and English 1996). In addition to source-dependent water and wetland species, dendrochronology of trees from the periphery of springs can also be useful for determining flow congestion. Melis et al. (1996) used such data to assess the flow variability of the spring-loaded Havasu Creek in the Grand Canyon, and reported that the Fraxinus velutina cores showed complacency of growth, suggesting a multi-year flow over 80 years. Surface-dwelling groundwater-dependent species that are based on a long-term of groundwater flow include several groups of plants, invertebrates, fish and amphibians. Among the plants in North America, such source-dependent species are selected sedges (Caryophyllaceae), Rushes (Juncaceae) and herbaceous taxa (e..B. some Primulaceae, Toxicoscordion spp., Flaveria mcdougallii). Under Hydrobiid spring snails are often limited to sources and canals, in particular the Pyrgulopsis and Tryonia (Hershler 1998, 2014), as well as some members of the water beetle families Elmidae and Dryopidae (Shepard 1993). In our studies of mountain springs in the American Southwest, chloroperlid stone flies and turbellare flatworms are often source-dependent species in cool-cold natural waters. Among North American fish, the puppy fish (Cyprinodontidae) and Goodeid topminnows (Goodeidae) are often source-dependent and often narrowly limited to individual sources (e.B Minckley and Deacon 1991; Unmack and Minckley 2008). Among the southwestern amphibians, populations of native ranid frogs of the genus Lithobate (Rana) are often associated with groundwater-dependent wet meadows (Cienegas, GDE-Fen). The huge aquatic light-bent salamander Cryptobranchus alleganiensis bishopi occurs only in clear, water-fed stream segments in the Ozarks. Several turtle species in eastern North America hibernate on the edge of cold water springs, where they are cooled but protected from freezing (Nickerson and Mays 1973; Ernst and Lovich 2009). Historical documentation is often useful for determining gDE status and the perennial of spring flow. Many sources of historical information can be available for such documentation, such as historical photographs and diaries, and interviews with longtime stewards and community elders. Such historical information can be very valuable in understanding changes over time; However, finding, determining the validity of such information, and compiling and interpreting the information can be challenging. The detection of GDEs by remote sensing (RS) involves the use of infrared and other thermal imaging cameras and has been successfully used to locate groundwater sources, especially in seasons with the greatest temperature differences between air and groundwater temperatures. Remote sensing (RS) provides a fast and spatially comprehensive technique for assessing the vegetation structure (e.B. leafy area index, basal area), vegetation function (e..B tree internal temperature, evapotranspiration rates and greenness) and relationships between climate variables, vegetation function and vegetation structure. An underlying conceptual model for the application of RS to the identification of the location of GDEs was that of the green islands. This approach compares the structure or function of a pixel in an RS image with that of a neighboring pixel. If one GDE covers a significant part of the area of one pixel, but not the other, it is assumed that during prolonged dry periods the structure/function of the two types of vegetation will deviate. This is because the vegetation that accesses groundwater is not soil-dry (if at all) like the vegetation that does not access groundwater. Accesses. the conceptual model of the green islands, the assessments of the vegetation structure or function are determined for the site and compared with adjacent control areas, either at a single time or preferred, over several contrast times (comparisons over wet and dry periods usually). In the United States, air thermal investigations of Florida's largest sources, Silver Springs, were conducted along the spring-fed outlet channel and new fed openings were discovered more than 1,200 m below the first source (Munch et al. 2006). Remote sensing techniques can be successfully used in areas with low gradient behavior that is not covered by dense vegetation. The U.S. Forest Service conducted remote sensing analysis for bogs in the Rocky Mountains to identify Fens (U.S. Forest Service 2012), and reported good results in the search for large fens that were exposed. However, similar remote sensing in the topographically complex Spring Mountains in southern Nevada detected less than 50% of the more than 200 sources in the area (U.S. Forest Service 2012). Münch and Conrad (2007) examined three catchment areas in the northern Sandveld of South Africa. They used Landsat images to identify the presence/absence of wetlands, and combined this with GIS terrain modeling to determine whether GDEs could be identified from a landscape wetness potential. It is important to note that this application focused on Class II GDEs – those that rely on surface expression of groundwater. They applied the philosophy of the green island and compared the attributes of potential GDEs with the attributes of the surrounding land cover at three contrasting times: July, when the rains began at the end of a dry year, August, in the winter of a wet year, and at the end of a dry summer. They concluded that RS data could be used to classify landscapes, and if combined with a spatial GIS-based model using landscape characteristics, they could create a map of the distributions of GDEs on a regional scale. However, it is not known whether this approach could be applied to Class III GDEs (which rely on access to groundwater below the surface). In dry and semi-dry regions, plant density often correlates with water availability. When groundwater is available to vegetation, plant density tends to be greater than adjacent areas where groundwater is not available. Lv et al. (2012) used remote-controlled images of a vegetation index (the Normalized Difference Vegetation Index; NDVI) to make changes in NDVI function of groundwater depth in northern China. A digital elevation model with a resolution of 25 m resolution and groundwater drilling data were used to create a contour map of the groundwater depths in the 2600 km2 long catchment area. About 29,000 pixels with a resolution of 300 m ANm. D.V. were then used and the following relationship was determined (Fig. 13.4):Fig. 13.4 The relationship NDVI and groundwater depth for the hailiutu River basin in northern China (Redrawn from Lv et al. 2012) This study showed that the largest NDVI, a reliable measure of vegetation cover, occurred in the shallowest depths of groundwater and which decreases in curvy as groundwater depth increases. They further analyzed NDVI data and identified five land classes, including water bodies and naked soil, as a land class with zero vegetation coverage; and arable land and adjacent areas as another class with the largest NDVI. The other three classes had intermediate values of NDVI. They then showed that the vegetated classes showed different reactions to the depth sand sit down to groundwater. A shutdown of about 10 m depth to groundwater was obvious; when the groundwater level was below 10 m, the vegetation cover was insensitive to a further increase in groundwater depth. A similar method was used by Jin et al. (2011) for the Ejina area in northwest China. Although much of the region is located in the Gobi Desert, with about 40 mm of annual rainfall, an oasis in the northern part of Ejina supports extensive agricultural and native vegetation. The NDVI was used by Jin and colleagues together with 13 groundwater boreholes, from which relationships between NDVI and groundwater depth were established for three vegetation classes (grassland, forest and bushland). Surprisingly, the maximum NDVI at the shallowest groundwater sites for each vegetation class, but at medium (2.5 – 3.5 m) depths, were observed. A demarcation of 4.4 m depth to groundwater was observed in such a way that the vegetation was missing in regions where the groundwater depth exceeded 5.5 m. Dresel et al. (2010) used geological, hydrogeological and environmental data to define regions with common physical and climatic profiles, which should therefore have similar RS signals. MODIS eVI and Landsat NDVI data were used and dry heginess thresholds (calculated as the Thornthwaite index) were developed for individual regions based on a correlation analysis of Landsat summer NDVI images and MODIS eVI. Both are substitutes for productivity, with eVI generally performing better than NDVI (Campos et al. 2013). Dresel et al. (2010) used three methods. In the first, the MODIS eVI images showed pixels with consistent photosynthetic activity throughout the year, and pixels with a variation over the year that was less than a standard deviation of the mean were considered consistent productivity throughout the year. For the second method, Landsat NDVI images were used to create areas with contrasting photosynthetic activity for a wet and a dry year. The third method used an unattended classification of landsat spectral data to identify spectral signatures of pixels that are considered highly likely for groundwater use with local expertise and other pixels with similar spectral signatures. Species-specific differences in spectral signatures have already been identified (Nagler et al. 2004). By combining all three methods within a GIS and finding pixels with consistent productivity throughout the year, as well as a high contrast between other local pixels and a similar spectral signature to known GDIt, it was possible to identify all pixels in a service area that had a very high probability of being a GDE. Then a ground truth was required. An alternative approach to mapping the location of GDEs involves mapping unloading zones, in particular discharge by refining vegetation and introducing them to the soil surface. The discharge of groundwater to the surface (in swamps, wetlands and rivers) or through transpiration has a profound effect on the ecology of these systems that use groundwater. In order to define the spatial extent of discharge over a landscape, a multidisciplinary approach is needed, incorporateing knowledge of geology, hydrology, ecology and climate (Tweed et al. 2007). Leblanc et al. (2003a, b) for example, used thermal, Landsat optical and MODIS NDVI data coupled with digital elevation models and deep groundwater data to locate runoff areas in a large semi-arid basin in Lake Chad in Africa. Tweed et al. (2007) investigated the discharge (and charge) of the Glenelg Hopkins Basin in southeastern Australia. The discharge was carried out by direct evaporation of the groundwater level with a probable limit of 5 m depth from which evaporation could occur; by vegetation from regions that overlay a shallow, unlimited aquifer, and discharge to the ground surface to localized depression, slope breaks and wetlands, rivers and the ocean. The methodology they use is summarized in this way (from Tweed et al. 2007, Fig. 13.5).Fig. 13.5 A scheme of the methodology used by Tweed et al. (2007) in the use of RS and GIS to identify the location of GDEs in a landscape The indicators for groundwater discharge used in this study include:1. Low variability of vegetation activity over wet and dry periods (seasons or years) with the NDVI as a measure of photosynthetic activity. 2. Topographic depressions and slope breaks across the catchment area, derived from a digital elevation model for the catchment area, to identify potential sites for surface discharge. A topographical wet index (w) has been calculated from: w = ln(1/tan) where β is the slope of the land area gradient. The identification of concave slopes by the identification of negative of slopes has been used to identify areas where potential saturation zones (created by groundwater drainage) may occur throughout the landscape. 3. Groundwater depth data were used to generate a groundwater flow, and these were combined with the digital elevation map to create a deep-to-groundwater map. From this A detailed map of potential unloading zones across the 11,000 + km2 catchment area was created, which far exceeded the capability if only the limited drilling data had been used. A map of the NDVI's standard deviation was able to identify locations where groundwater supported vegetation activity, thus identifying GDEs in the catchment area. One possible limitation of this method was that it was most accurate in drier parts of the catchment area, where rainfall is more likely to limit vegetation activity. It was also found that the identification of topographical depressions was a more reliable indicator of groundwater drainage than the identification of slope fractures. The energy balance equation for land areas can be written as: LE + H = R n – G, where LE is latent energy flow (=ET), H is a meaningful heat flux, R n grid radiation and G soil heat flux. Temperature differences between the limit air temperature and the treetop temperature can be used to estimate a meaningful heat flux. Assuming a 24-hour cycle G = 0, and R n is either measured or derived from remote sensing data, then LE (i.e. ET) is calculated by difference. Li and Lyons (1999) used three models based on surface temperatures to estimate ET. The first model used only differences in surface and air temperature to calculate ET, the second model required NDVI data and surface temperature. This model requires that the four extreme values of surface temperature and NDVI are present in the examination area (i.e. patches of dry, bare soil, wet bare soils, wet, fully immersible spots and dry (water-contaminated) fully contemplative surfaces). This makes its application problematic. The third method simply used the Priestley-Taylor equation (see Li and Lyons 1999) to estimate potential ET (Ep). Two of the most important functional properties of terrestrial ecosystems are rates of water use (either transpiration or evapotranspiration) and rates of carbon fixation. The currents of transpired water and carbon uptake are coupled by the effect of stomata, through which both gases must flow. Due to the close coupling of water and carbon flows, vegetation indices such as NDVI or eVI, which are good proxies of productivity and thus carbon flow, can be successfully used in the search for GDEs, where it is an increase in water supply that drives their structural and functional differences (compared to neighbouring No-GDEs). Information management is a serious challenge for understanding and managing GTE. Accurate geo-referencing and data on the distribution and ecohydrology of feathers and other GDEs initially involves the development of an appropriate database framework (Springs Stewardship Institute 2012). Some or many of the above methods for determining the GDE distribution allow the development of a georeferenced map of sources within landscapes. However, A common problem with such mapping efforts is the resolution of duplication errors. We have repeatedly noted that (a) not a single source of information (usually GIS layers or survey reports) provides a complete list of sources or other GDEs within a large landscape; (b) that each source of information contains unique feathers that have not been found elsewhere; and (c) that the same GDEs can be mapped in multiple places with different names. Stevens and Ledbetter (2012) used 10 sources of information to identify 150 sources in the North Kaibab Forest District in northern Arizona, 50% more sources than documented by the administration agency, and field investigations increased the number of known sources in the landscape to more than 200. The development of an appropriate map and database of the sources of large landscapes provides an essential tool for monitoring, modelling and further research into the status of the underlying aquifers. Estimating the groundwater needed to maintain the GDE function is an essential step towards the sustainable management of both GDEs and groundwater resources. However, it represents many methodological obstacles, including:1. Up-scaling of tree scale measurements of tree water use; 2. dividing the total use of vegetation into rain and groundwater sources; 3. understanding seasonal/life cycle fluctuations in groundwater use; 4. Understanding the influence of the climate on seasonal scales on the rates of tree water use and the division of water use into rain and groundwater sources. In addition, what is required for the establishment and persistence of the GDE function is often not well characterized; Therefore, the focus was on measuring the use of water in existing GGD and using this characterisation as a basis for the starting conditions. A number of instruments are available to estimate groundwater use according to Class III GDEs. These will now be briefly discussed. Due to the lack of data on the above points (1)–4, Leaney and colleagues developed a novel, simple but useful first-order method to estimate the groundwater use of vegetation using a simple Excel spreadsheet tool (Leaney et al. 2011). The Excel table contains three methods for estimating groundwater discharge rates by vegetation: (a) a groundwater risk model; (b) an ecological model of optimality; and (c) a groundwater discharge salt function. These are summarized in Table 13.1.Table 13.1, how groundwater discharge rates are estimated by dievegetation in low-data areas, summarized from Leaney et al. (2011) is a simple water balance model that uses historical monthly rainfall and monthly evaporation data for each location. The soil profile is defined by the user and the soil structure is used to estimate the soil moisture properties for each layer. Groundwater drainage through vegetation applies whenever evapotranspiration (ET) exceeds the precipitation plus the soil. Exceeds. Shops. The White Method (White 1932) described in section 13.3.2 for the analysis of underground changes in the deep groundwater situation can not only be used to identify the position/presence of a GDE in a landscape, but can also be used to quantify groundwater utilization rates. The amount of tangleed water shall be calculated from the change in the volume of water in the aquifer, which would explain the observed changes in the depth of the aquifer every hour or daily, provided that the specific yield of the aquifer is known with sufficient accuracy and confidence. Butler et al. (2007) investigated the control of differences in groundwater use rates at several waterfront sites in the High Plains region of the United States. They found that the main drivers of vegetation water use were meteorological properties, vegetation attributes, and the specific yield of the aquifer. Their estimates of groundwater use (3-5 mm d'1) were well in consistent with those derived from sap flow measurements of the use of tree water. For a detailed assessment of the technical problems associated with the application of the White method, the reader is referred to Loheide et al. (2005). Further examples of the estimation of groundwater use rates using the White method can be found in Lautz (2008), Martinet et al. (2009) and Gribovszki et al. (2008). Methods for remotely tested estimates of groundwater discharge are currently being developed. It is important to quantify the water balance of dry and semi-dry groundwater basins in order to define safe yields for these resources. Obtaining accurate and spatially distributed estimates of vegetation discharge is problematic, expensive and time-consuming with field techniques. Therefore, Groeneveld and Baugh (2007) derived a new formulation of the NDVI standard that extends the NDVI distribution for vegetation from zero to one. This new NDVI (NDVI*) can be calibrated to quantify the actual evapotranspiration rates (ET a), and calibration requires only standard weather data from which to calculate (E o) (the grass reference ET, calculated according to the Penman-Monteith equation as described in the FAO-56 method (Allen et al. 1998). The NDVI* is functionally equivalent to the crop coefficient commonly used in micrometeorology (K c). This method is particularly applicable to green edible dry and semi-dry areas with a shallow groundwater level where rainfall is low, often irregular, but the water supply of the roots is relatively constant. As a result, ET ET o , which varies depending on solar radiation, wind speed and vapour pressure deficit, tracks closely. Groeneveld et (2007) applied the NDVI* methodology to three different drying points in the US where annual ET values were available through the use of Bowen ratio or whirlcoscovore devices. A linear correlation (R2 = 0.94) between measured annual ET a and mid-summer NDVI* was measured over the pooled data with three three Composition and structure at the three sites. The deduction of the contribution of the annual rainfall to the annual ET results in the amount of groundwater generated by the vegetation (ET gw). For example, ET gw = (ET o – precipitation)NDVI* Over sites and over years, the average error in ET gw was estimated at about 12%, which is a very valuable estimate of groundwater use due to lack of field assessments. Groeneveld (2008) applied the methodology of Groeneveld et al. (2007) using HOCHsummer NDVI data to estimate the annual total amount et of alkali peeling vegetation in Colorado. An estimate of the annual groundwater use was then estimated as the difference between annual rainfall and annual ET for each year. Estimates of groundwater on the ground were greater than the estimated ndVI data and ET o, as the remote sensing method does not include surface evaporation of groundwater. The annual ET gw * was compared with the measurements of Cooper et al. (2006) at the same site, which were agreed within 20%. Similarly, as noted in the discussion on RS methods for searching for ET in 2008, Scott et al. (2008) developed a numerical relationship for ET a and concluded that the difference between ET a and annual rainfall was groundwater use. Stable isotopes have been extensively used to provide estimates of the proportion of total vegetation water extracted from groundwater (Feikema et al. 2010; Kray et al. 2012; 2011; McLendon et al. 2008; Querejeta et al. 2007). Therefore, in addition to analyses of the stable isotope composition of soil water, groundwater and xylem water, an independent estimate of water use rates is required. Methods for estimating vegetation water use include vortex covariance (Eamus et al. 2013), measurement of juice flow rates (Zeppel et al. 2008) and remotely tested estimates (Nagler et al. 2009). If only a single isotope is analyzed (2H or 18O), a linear mixing model can only distinguish between two potential water sources (groundwater and soil water). When both isotopes are used, the spatial resolution is increased and one can distinguish between three water sources, but only if the two isotope compositions are independent of each other, which is often not the case. Interestingly, early work in 1996 found that the application of stable isotope analyses was the most accurate method available in a comparative analysis of groundwater inflows into wetlands (Springs Stewardship Institute 2012). Two generalities can be identified in the results of stable isotope studies of GDEs. First, as the depth increases, the proportion of total vegetation water use discharged from groundwater (O'Grady et al. 2006), although this may vary between different vegetation communities (McLendon et al. 2008). Secondly, the proportion of groundwater normally used by vegetation (McLendon et al. 2008, but not always et al. 2012) increases with increasing time and soils dry out, so seasonal use of groundwater can occur when rainfall is very seasonal and groundwater availability is maintained throughout the dry season (O'Grady et al. 2006). The stable isotope composition varies depending on the depth (Fig. 13.3) and the use of an average to represent the entire rooting depth of the vegetation leads to errors. Even with two independent isotopes available for analysis, the relative contribution can be determined from only three sources. To overcome this limitation, Cook and O'Grady (2006) developed a simple model of water absorption, in which the relative uptake from different depths is determined by (1) the gradient of the water potential between the ground and the canopy; (2) Root distribution depending on depth; and (3) an incompetent hydraulic conductivity parameter. The isotopic composition of the water by the soil profile and the xylem water is then used to limit root distributions (as opposed to destructive measurement in situ). This model has several advantages over the most commonly used end element analyses (Phillips and Gregg 2003): (1) produces a more quantitative estimate of the proportion of extracted water from different depths (including groundwater); (2) does not require different values of isotope composition for final element analyses and can therefore deal with the more typical classification of the isotope composition observed by the soil profile; and (3) is based on simple ecophysiological principles. Sapflow sensors were used to measure the rates of tree water use for four species growing in a tropical remnant, and this was scaled up using surface basal surfaces. Cook and O'Grady (2006) showed that two species source7-15% of their transpirational water from the groundwater table, a third species had access to 100% of their water from the groundwater table, and a fourth type of access received between 53% and 77% of their water from the groundwater table – further confirmation of the niche separation of water absorption patterns for coexisting species. Human activities threaten GdEs with disruptive habitats, the depletion of groundwater reserves, the change in groundwater regime at a site that goes beyond the natural variations that have been observed at the site so far, and a reduction in groundwater quality. Worldwide, GdEs are and remain threatened by groundwater extraction, as the water demand of the growing population increases and industrial demand increases (Danielopol et al. 2003). Wada et al. (2010) estimated that global groundwater depletion (i.e. groundwater abstraction beyond recharging) in sub-humid to dry areas was about 280 km3 in 2000 and has doubled since 1960. The increasing demand for water is likely to significantly outweigh climate change in the definition of global water resource by 2025 (Vörösmarty et al. 2000). Local, human human GDE habitats have been affected by the clearing, filling or drainage of wetlands and changes in surface water courses. Regionally, the main anthropogenic threats to GdEs are the change in surface water regime and quality through river regulation and land-use changes; Changes in groundwater regimes and quality resulting from agricultural practices, urban and industrial development, mining and forestry planning (Fig. 13.1). For GDEs that rely on both surface and groundwater sources, surface water regime (including floods) and quality are considered to be the most important factor threatening GDEs (Eamus et al. 2006). Elsewhere, evidence of changes in the ecosystem due to changes in currents and a decline in surface water quality has been reviewed elsewhere (Nilsson et al. 2005; DeFries et al. 2004). This section focuses on groundwater regulation and groundwater quality. Groundwater extraction is one of the greatest threats to the groundwater regime. Groundwater has been promoted to support agricultural activities (in particular irrigation) to cover the use of housing and to support urban and industrial development. In these cases, groundwater is often obtained by pumping wells into closed or unlimited aquifers. Excessive aquiferpumping in a closed aquifer will depress the entire water bottom and reduce groundwater discharge into the springs (Weber and Perry 2006) (Fig. 13.6). The effects are regional. In contrast, the effects of aquifer pumping from an unlimited aquifer are more localized. In unlimited aquifers, when extraction is faster than charging, the groundwater depth increases and forms a cone of deepening around the well, which can extend over many hundreds of meters from the well (Fig. 13.6). In addition, the direction of groundwater flow can be changed by the generation of new hydraulic gradients: groundwater may no longer flow into the local stream, and some water can be drawn from the stream into the well, thereby reducing the flow flow. The time delay between extraction and a reduction in discharge into a stream varies from a few hours to many centuries, depending on the extraction sites (relative to the stream), the extraction volume and the groundwater flow (Evans 2007).Fig. 13.6 Chart showing the possible effects of groundwater pumps on GDIt Increased depth into groundwater and the disappearance of sources have been reported worldwide and are associated with excessive groundwater extraction for agricultural and urban development. , Mining and plantation forestry (Fig. 13.1). The groundwater depth has increased by 4-17 m in an irrigation region in northwestern China and forms several deepening cones with a depth of about 1000 km2 (Wang et al. 2003). Similarly, Burri and Petitta (2004) observed the progressive disappearance of numerous sources in the Fucino Plain, Italy, due to the increasing agricultural Horticulture and second harvesting practices. In some areas with extensive urban development, groundwater degradation has occurred at alarming rates. In London, for example, the groundwater level has fallen more than 70 metres below the surface (Elliot et al. 1999); in Bangkok, the groundwater level has fallen by 25 m since 1958; in Tamil Nadu, India, there is a decrease of 30 million. in 15 years (Danielopol et al. 2003). The decline in groundwater levels in the Spanish city of Doana was mainly due to the pumping of the urban water supply of a tourist resort and, secondly, to the phasing out of large pine plantations. The drainage of mines (removal of water by pumping or evaporation) can have a major impact on the groundwater and cave system on site and on springs near mine sites. Clusters of mining operations can have an impact on the depths of groundwater at regional level due to their cumulative impact (Clifton and Evans 2001). In addition to groundwater extraction and drainage of mines, canal gravel or sand mining can cause the incision of a river bed that lowers the floodplain water tables (Kondolf 1994). Scott et al. (1999) reported a decrease in groundwater levels of more than 1 m at sites affected by gravel mining (compared to no significant decrease in control points). A sustained reduction in groundwater levels of more than 1 m has led to a significant decline in Populus growth and a mortality rate of 88% over a three-year period (Scott et al. 1999). Water logging, typically caused by forest clearing and poorly managed irrigation in agricultural land, can lead to an increase in groundwater levels and associated effects due to impaired root function due to the development of anoxic conditions within the root zone (Pimentel et al. 1997). Reports of groundwater contamination caused by human activities are abundant. Nitrate leaching from agricultural land into shallow groundwater has been reported in many regions of the world (Andrade and Stigter 2009). Increased nitrate levels in groundwater can be obtained from nitrogen fertilizers and manure, oxidation of organically bound nitrogen in soils, animal feed sites, septic tanks and sanitation. The severity of contamination is altered by other factors such as lithology, dissolved oxygen content and land use. Andrade and Stigter (2009) reported that rice fields on fine-grained alluvium usually have low dissolved oxygen and minimum nitrate concentrations in groundwater due to denitrification. In contrast, areas with vegetable plants have Combination with coarse grain lithology and high hydraulic conductivity, higher nitrate concentrations in shallow groundwater. The discharge of nitrate-enriched groundwater can alter nitrogen concentrations in the receiving water, thus increasing the risk of eutrophication and algal blooms. Pesticide contamination can be a problem for shallow groundwater. In the U.S., more than half of the wells in agricultural contain one or more pesticide compounds (Gilliom et al. 2006). The use of low-quality pesticides with low degradation rates, incorrect use of pesticides and inappropriate disposal methods can lead to groundwater being contaminated by pesticides, among which herbicides are most commonly detected in groundwater (Andrade and Stigter 2009). Urban development can affect groundwater quality and thus damage urban ecosystems. Examples include leaks from septic tanks, underground fuel tanks, landfills and the use of fertilisers and pesticides for gardens and recreation areas. Animal husbandry, horticulture, waste disposal, pit lour construction and rainwater/sanitation have led to increased local microbial and organic contamination of shallow groundwater (Kulabako et al. 2007; Massone et al. 1998). Foppen (2002) reported increased concentrations of almost all major cations and anions and acidification of groundwater in Sanaa, Yemen, due to the continuous infiltration of wastewater into the aquifers via sinks. More recently, it has been shown that urban groundwater in German cities is polluted with xenobiotics such as medicines, personal care products (collectively known as PPCPs) and endocrine-active substances (Schirmer et al. 2011). However, their potential long-term effects on ecosystems and humans remain largely unknown. Mining can contaminate groundwater during mining operations (e.B. leakage of dams and crushed rock dumps, which can cover hundreds of hectares on a mine site), as well as the recovery phases after the abandonment of mine sites (Younger and Wolkersdorfer 2004; Gao et al. 2011). Drainage disrupts groundwater stratification and thus alters the environment needed by cave or groundwater ecosystems and the associated stygofauna. Cidu et al. (2001) reported that the closure of mines and the associated cessation of aquifers and mine flooding due to the increase in deep, saline groundwater may pose a risk of contamination to shallow aquifers. Progressive mine floods also cause groundwater contamination due to weathering of ore minerals and remobilization of metals in mine waste (Razowska 2001). To sum up, groundwater regulation and quality are threatened by many human activities, including agricultural practices, urban and industrial development, mining and forestry. These threats can have a profound impact on GDEs at local and regional level in the short and long term. The impact of groundwater abstraction on THE GDEs and their will be discussed below on the basis of two case studies. The impact of groundwater abstraction on forests has been documented for gnangara Mound, a shallow, unlimited aquifer of the Swan Coastal Plain in Western Australia (Canham et al. 2009, 2012; Groom et al. 2000; Stock et al. 2012). Increased groundwater depth is the result of a Decrease in annual rainfall throughout the region, increased abstraction for human use and increased discharge (reduced charge) as a result of the development of a plantation industry in the region. A number of changes in plant physiology, Ecophysiology and ecology medium- and long-term changes in water availability (Fig. 13.7).Fig. 13.7 Schematic representation of some changes in plant physiological, ecophysiological and ecological development in connection with short-, medium- and long-term changes in water availability in 1985 increased rates of summer withdrawal in this Mediterranean climate led to increased and widespread mortality (up to 80% mortality near the abstract) of the domestic bank. In order to determine longer-term floristic changes resulting from groundwater abstraction, a series of transect studies was launched in 1988. An increase in depth to groundwater of 2.2 m, coupled with above-average summer temperatures, led to a mortality rate of 20 to 80% in adults and a mortality rate of up to 64% in the underclass species, 2 years after the start of groundwater pumping (Groom et al. 2000). Control points that were not affected by groundwater pumps did not show increased mortality. Due to the large differences between species in mortality rates, another study examined the susceptibility of different species to reduced water availability (Canham et al. 2009; Froend and Drake in 2006). Froend and Drake (2006) compared three Bankisa and one Melaleuca species by using Xylem-Embolie vulnerability curves as an indicator of sensitivity to water stress. They found that the Xylem vulnerability reflected the broad ecohydrological distribution of species over the topographic gradient at the site, and they were able to identify a threshold water potential below the probability. Similarly, Canham et al. (2009) investigated Huber values (ratio of sapwood to leaf area), leaf-specific hydraulic conductivity (k l) and xylem susceptibility of two obligate phreatophytes and two optional phreatophytes. At sites where water availability was high (no increase in depth to groundwater), there were no interspecific differences in the susceptibility to water stress. In comparison with the upper and lower slopes (corresponding to larger or smaller groundwater depths), the two optional phreatophytes (but not the obligatory phreatophytes) were more resistant to xylemembolise on the upper slope than on the lower slope, while one of the obligate phreatophytes did not change its sensitivity (Canham et al. In addition to the differences in the sensitivity of above-ground tissues to changes in water availability, it is likely that differences in the reactions of the root to changes in groundwater deep groundwater contribute to the effects of changes in the depth s. groundwater on the vegetation in the GDEs. In a one-way Study of two Banksia tree species, Canham et al. (2012), observed that root growth at low-depth sites up to groundwater was in sync with above-ground growth patterns. This was in contrast to patterns observed at depth, where root growth took place throughout the year and was independent of the air climate. As the groundwater depth increased in the summer on this winter precipitation area, the roots became deeper and deeper after the capillary edge. When the charge occurred in winter and the depth decreased to the groundwater, the anoxie led to root death in the depths. These authors concluded that the ability to rapidly increase root depth in summer is a critical attribute of phreatophytes occupying sites with seasonally dynamic depthto groundwater. Long-term (>2 years) studies on the impact of changes in groundwater are relatively rare, despite the importance of such studies for the development of ecosystem response pathways for the effects of groundwater abstraction. Froend and Summer (2010) studied a rare, 40-year duration, vegetation survey s-set for the Gnangarra Mound in Western Australia. Although the average long-term rainfall (1976-2008) is 850 mm, it has declined over the last 40 years. At present, the annual average is about 730 mm. Together with the increased groundwater abstraction, this has led to an increase in groundwater of about 1 m in the last 50 years. Seasonally, the depth fluctuates to the groundwater about 0.5-3 m, with a maximum depth occurring at the end of the summer. Two transects were used – a control transsect in which there have been gradual increases in depth into groundwater (9 cm y-1) in recent decades; and an experimental transect in which large rates of increase in groundwater (50 cm y-1) have occurred due to falling rainfall and extensive groundwater abstraction. Three vegetation communities were identified with main coordinate analyses that were clearly connected to slope, middle and upper slope positions corresponding to shallow, medium or deep deep groundwater. Species, which are known to have a high dependence on a consistent water supply (Mesische), dominated the slope, while xeric species dominated the upper slopes. The control transect (slow rates of increase in the depth-to-groundwater) did not support the hypothesis that groundwater abstraction would lead to a replacement of the mesic by the xeric species. Most of the compositional and structural characteristics of the three Communities remained unchanged. The main Community response was to change the abundance of mesian and Xeric species rather than to replace one species completely with another. In on the results of Shafroth et al. (2000) (2000) in areas with shallow groundwater were no more sensitive to the increase in groundwater depth than xeric species. In the experimental transsect, where the increase in depth into groundwater was much faster (50 cm y-1), the changes in composition were much more pronounced and mass mortality was observed in all classes (mesisch to xeric). This result underlines the importance of increasing groundwater depth in determining the response of species and communities. Stacey et al. (2011) carried out a systematic review of the literature on the restoration of dry land sources to determine how successful projects were in the restoration of hydrology, geomorphology and biological assemblage composition and structure in relation to those of natural sources with minimal anthropogenic disturbances. Unfortunately, the great inconsistency in the justification for and in the implementation, monitoring and reporting of efforts to restore feathers worldwide made it impossible to carry out metastatistical analyses of the quality of the restoration worldwide. Stacey et al. (2011) recommended standardized ecosystem conditions and restoration assessment protocols are needed to better understand the success of projects. Due to the inability to report on a global summary of the success of restoration and management, we are providing a case study on specific areas of relief to provide some lessons from restoration and management efforts. Hoxworth Springs is a source of Rheochrene on the Mogollon Rim of the southwestern Colorado Plateau (Godwin 2004). This system is typical of both morphology and the dismantling of many power channels connected to rheochren sources in the southwestern United States. Causes of the system's drainage are attributed to anthropogenic modification of the canal with the installation of a series of dams and the grazing of pets and imported non-native wildlife in the canal and catchment area. In cooperation with Denland administrators, the sewer rehabilitation was completed in order to restore the function and structure of the system. The restoration included the morphological reconstruction of the electricity channel as well as the hydrological and vegetative monitoring. The canal was severely cut and the sinusity decreased, resulting in greater flow speeds, steep canal banks and flood currents that could not dissolve over the floodplain. The restoration of Hoxworth Springs involved redesigning the canal based on morphological patterns observed in abandoned reference sections of the canal on the surface of the flood plain and with similar departures, Rheocren spring channels dominated in the region (Griffiths et al. 2008). Re-vegetation was carried out to stabilize the restored canal banks and large exclosures were built to manage grazing along the canal. A three-dimensional groundwater flow model has been created to assess the interpretation and Impact of recovery efforts on multi-year flow discharge, recovery effectiveness, and system response to climatic extremes. The model showed that the length of the perenning river in the canal depends on recent climatic conditions. The use of a groundwater model to assess recovery efforts allows the user to change the charging conditions based on climatic or hydrological disturbances and to estimate the impact on the length of the multi-year flow and water availability for the adjacent ecosystem. For the first time, we now have a set of tools that cover the complete temporal and spatial scales over which ecology moves (seconds to decades; from leaf to whole catchment area). Measurements of stomatal or soon conductivity, juice flow, treetop temperature, leaf area index and evapotranspiration and productivity rates can be carried out using ecophysiological techniques and remote sensing technologies. This data can be used in simple, moderate and complex ecosystem structure and function models to identify the presence, spatial scale and health of GDEs. What remains to be done? The three biggest knowledge gaps, in our opinion (1), are the definition of ecosystems' response to changes in groundwater availability or quality; (2) setting the threshold for THE GDEs from which unacceptable changes in the GDE structure and function occur; and (3) a mechanistic understanding (and thus predictive capacity) of the interplay of future climate variability on GDEs.Allen RG, Pereira LS, Raes D, Smith M (1998) Harvest-evapotranspiration guidelines for calculating the need for plant water. 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