Chapter 3
Ecohydrology: Processes and Implications for Rangelands

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Abstract This chapter is organized around the concept of ecohydrological processes that are explicitly tied to *ecosystem services*. Ecosystem services are benefits that people receive from ecosystems. We focus on (1) the regulating services of water distribution, water purification, and climate regulation; (2) the supporting services of water and nutrient cycling and soil protection and restoration; and (3) the provisioning services of water supply and biomass production. Regulating services are determined at the first critical juncture of the water cycle—on the soil surface, where water either infiltrates or becomes overland flow. Soil infiltrability is influenced by vegetation, grazing intensity, brush management, fire patterns, condition of biological soil crusts, and activity by fauna. At larger scales, water-regulating services are influenced by other factors, such as the nature and structure of riparian zones and the presence of shallow groundwater aquifers. Provisioning services are those goods or products that are directly produced from ecosystems, such as water, food, and fiber. Work over the last several decades has largely overturned the notion

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that water supply can be substantially increased by removal of shrubs. In riparian areas, surprisingly, removal of invasive, non-native woody plants appears to hold little potential for increasing water supply. Here, the primary factor appears to be that non-native plants use no more water than the native vegetation they displace. Clearly there is a close coupling between biota (both fauna and flora) and water on rangelands—which is why water-related ecosystem services are so strongly dependent on land management strategies.

Keywords Ecosystem Services • Infiltration • Rangeland Hydrology • Riparian • Groundwater • Overland Flow • Soil Water • Climate • Water Supply • Climate Regulation • Erosion • Spatial Variability • Scale • Thresholds • Connectivity

3.1   Introduction

The distribution, quality, and provisioning of water are intimately related to how rangeland landscapes function and are managed, particularly with respect to land-use change. Understanding the linkages between vegetation and the water cycle is a major focus of ecohydrology, an emerging discipline that melds the sciences of hydrology and ecology as a means of addressing complex environmental issues. Its scientific heritage also embraces many other disciplines, including watershed management, plant physiology, soil science, geomorphology (Newman et al. 2006), and of course rangeland hydrology (Branson et al. 1981). In addition, the importance of interactions between fauna and the water cycle is increasingly being recognized.

Ecohydrology is very much an applied science with a focus on problem solving (Nuttle 2002; Jackson et al. 2009b; Wilcox et al. 2011), but at the same time it has a firm theoretical foundation (D'Odorico et al. 2012, 2013a; Turnbull et al. 2012; Saco and Moreno de las Heras 2013). Because of its strong intellectual roots in research conducted on drylands—including semiarid and subhumid rangelands (Rodriguez-Iturbe and Porporato 2004; D'Odorico and Porporato 2006; Newman et al. 2006)—and its "transdisciplinary" nature, ecohydrology has advanced our knowledge of rangelands (Wilcox and Newman 2005; Wilcox et al. 2012a). But much more needs to be done to take full advantage of the scientific strengths of ecology and hydrology (King and Caylor 2011).

In this chapter, we present some of the major ecohydrological advances that have occurred in rangelands in the last quarter century and discuss their importance for management. There has been extraordinary scientific progress on so many fronts that it will be impossible to adequately address all of them; but we aim to provide a comprehensive overview of those most relevant to rangeland systems. We rely extensively on the recent publication of several review papers and books dealing with the ecohydrology of rangelands (D'Odorico and Porporato 2006; Newman et al. 2006; D'Odorico et al. 2010; Asbjornsen et al. 2011; Wang et al. 2012a).

We have organized our chapter around the concept of *ecosystem services*—as elaborated in the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment

2005). Ecosystem services are benefits that people receive from ecosystems. They can be categorized as *regulating services*, *supporting services*, *provisioning services*, and *cultural services*. We focus on (1) the regulating services of water distribution, water purification, and climate regulation; (2) the supporting services of water and nutrient cycling and soil protection and restoration; and (3) the provisioning services of water supply and biomass production. In addition, we review current conceptual, theoretical, and technical developments that will provide a foundation for future advances in rangeland ecohydrology—advances critical to informed management decisions and actions needed to meet the growing environmental challenges of rangeland systems.

3.2 Ecosystem Services

The provisioning of water to ensure that humans obtain the quantity and quality of water needed is the most fundamental service provided by ecosystems (Falkenmark and Rockstrom 2004; Brauman et al. 2007). Paradoxically, this is especially true of rangelands, even though most are considered "drylands," which by definition convert a relatively small percentage of precipitation into streamflow or groundwater (Wilcox et al. 2003b). Water produced on rangelands, whether drawn from aquifers or from surface sources, is vitally important to support the people, livestock, and wildlife that inhabit these regions (Le Maitre et al. 2007; Reynolds et al. 2007). Many dryland population centers are growing at alarming rates, and this growth brings with it numerous environmental stresses (D'Odorico et al. 2013a). The degradation of rangelands diminishes their ability to regulate and provide water (MEA).

Figure 3.1 illustrates an important conceptual advance in understanding water dynamics in rangelands: the explicit partitioning of water resources into "blue water" (liquid water) and "green water" (vapor- or water-produced evapotranspiration [ET]) (Falkenmark and Rockstrom 2004, 2006; Gordon et al. 2005; Falkenmark et al. 2009; Rockstrom et al. 2009; Hoff et al. 2010). To date, the water management community has focused almost exclusively on blue water resources and has failed to recognize the opportunity to effectively allocate green water. Maximizing the amount of green water used for plant production or transpiration and minimizing the amount lost as soil evaporation is an imperative. How rangelands are managed—especially their surface cover—has a tremendous effect on both the relative proportion of blue water to green water and the partitioning of green water between E and T. The ability of rangelands to regulate and provide water is strongly dependent on conditions at three critical junctures in the terrestrial water cycle (Falkenmark and Rockstrom 2004). The first and most critical is whether water infiltrates into the soil or becomes *overland flow*—which is mainly a function of rainfall intensity, slope, and soil infiltrability. The importance of soil infiltrability has long been recognized (Smith and Leopold 1941) and has been the focus of considerable research in the last half century or more. It is influenced by many factors, including management practices; for example, overgrazing that results in a loss of vegetation cover and an increased exposure of bare soil can dramatically reduce soil infiltrability (Blackburn et al. 1982; Snyman and du Preez 2005).

Water that does not infiltrate becomes overland flow on slopes, but the final outcome in terms of net water losses is highly scale dependent. At the hillslope scale, runoff–runon dynamics become important and are strongly influenced by the spatial variability of infiltration. For example, overland flow may be generated from some areas on the hillslope only to infiltrate the soil somewhere downslope (Bergkamp 1998a; Wilcox et al. 2003a), and can contribute to surface and groundwater recharge.

The second critical juncture is at the root zone: soil water may drain out of the root zone, and eventually be stored as groundwater or discharged into a stream as baseflow, or may stay in the root zone and eventually be transpired or evaporated from the soil surface. Although largely a function of climate, soil, and geological characteristics, this process can also be affected by management strategies, especially if the functional type of vegetation—and particularly its rooting depth—is changed. The linkage between vegetation and groundwater is very much influenced by the depth to groundwater. Recent work has highlighted the importance of rangelands where groundwater tables are shallow and strongly influenced by vegetation that are termed *groundwater-coupled rangelands* (Jobbágy and Jackson 2004a).

The third critical juncture is the fate of soil water: whether it is absorbed by plants and transpired or lost through evaporation from the soil surface, which is often described as the partitioning of E and T (Fig. 3.1). This juncture is critical because it dictates the amount of biologically available water on rangelands (Falkenmark and Rockstrom 2004; Newman et al. 2006). The portioning of E and T is central to water cycling and is discussed in more detail in the section *Supporting Services*, below.
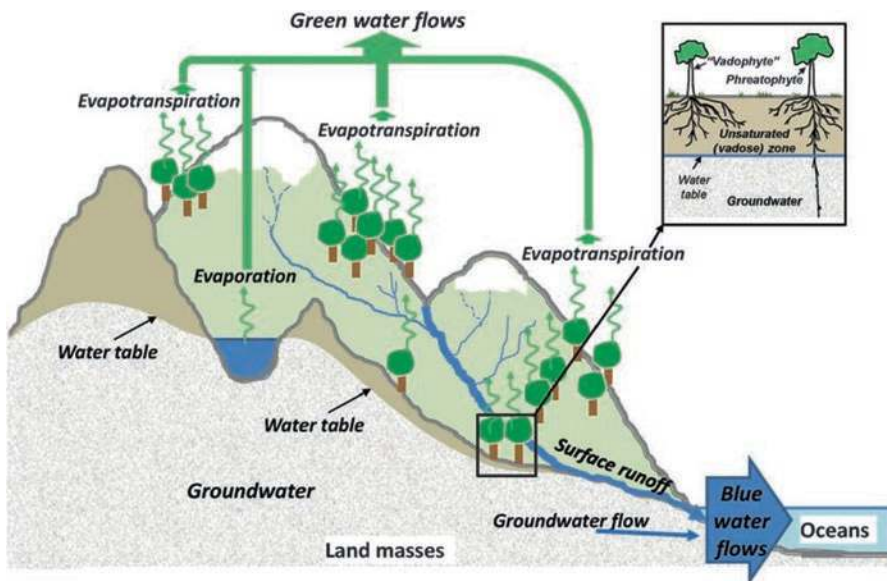

Fig. 3.1 Conceptual diagram of the water cycle, highlighting blue (liquid) and green (vapor) flows. Source: Figure 1 in D'Odorico et al. (2010)

3.2.1 Regulating Services: Water Distribution and Purification

Ecosystem services regulating water on rangelands include those that affect the amount, timing, and quality of blue water flows. These are to a large extent determined at the first critical juncture of the water cycle—on the soil surface, where water either infiltrates or becomes overland flow. For this reason, a great deal of research, most of it conducted at the point or plot scale, has focused on understanding the infiltration process and how it is affected by different management strategies (Pyke et al. 2002; Stavi et al. 2009).

3.2.1.1 Infiltration: Water Regulation at the Soil Surface

Infiltration of water into the soil is enhanced and maintained by the presence of vegetation, both by direct influences (soil protection, root action, etc.) and by modification of the soil through the addition of organic matter. This tight coupling between vegetation and soil infiltrability on rangelands was recognized many years ago (Smith and Leopold 1941; Woodward 1943; Dyksterhuis and Schmutz 1947; Dortignac and Love 1961); but recent research is adding greatly to our understanding by providing specifics concerning how management practices and disturbances (grazing, shrub management, fire) and vegetation cover types (shrubs vs. grasses, biological soil crusts) affect soil infiltrability, but also the contributions of spatial variability and scale. In addition, we now recognize that fauna—large and small— can significantly affect soil infiltrability.

Influence of Grazing. There is an extensive body of work examining the ecohydrological influence of grazing, and specifically its influence on soil infiltration. Much of this work was conducted in the USA in the 1970s and 1980s and has been summarized in several review papers (Gifford 1978; Wood et al. 1978; Wood and Blackburn 1981; Blackburn et al. 1982; Trimble and Mendel 1995). The findings consistently show that, irrespective of grazing systems, light-to-moderate grazing generally has little adverse effect on the ecohydrology of rangelands and may even have a positive effect, whereas heavy grazing generally significantly decreases soil infiltrability. These conclusions have been verified by more recent investigations conducted on rangeland throughout the globe (Hiernaux et al. 1999; Ludwig et al. 1999; Savadogo et al. 2007).

Influence of Shrubs. Over the past several decades, grasslands and savannas worldwide have been undergoing a process of woodland conversion, often described as woody plant encroachment (Archer 1994; Archer et al. 2011). For many rangelands, attempts to reverse this process or even to control it have met with minimal success (Archer et al. 2011). During the past quarter century, considerable research has been focused on understanding the ecohydrological implications of this conversion (Huxman et al. 2005; Wilcox et al. 2006). It has generally been found (though not always—see Moran et al. (2010)) that infiltration rates are higher beneath shrub canopies than in intercanopy areas (Lyford and Qashu 1969; Seyfried 1991;

Bergkamp 1998b; Schlesinger et al. 1999; Wilcox 2002; D'Odorico et al. 2007; Wilcox et al. 2008; Pierson et al. 2010; Daryanto et al. 2013; Eldridge et al. 2013), primarily owing to the accumulation of organic matter under shrubs, root activity (Joffre and Rambal 1993; Martinez-Meza and Whitford 1996; Jackson et al. 2000), and soil disturbance by fauna (see "Influence of Fauna" section). In some situations the chemical composition of the litter may cause water repellency (hydrophobicity), which reduces the infiltration capacity of soils beneath the canopy, at least in the short term (Doerr et al. 2000). In addition, burning can cause or aggravate hydrophobicity (Hester et al. 1997; Cammeraat and Imeson 1999).

Influence of Biological Soil Crusts. Biological soil crusts are the community of living organisms, including fungi, lichens, cyanobacteria, and algae, at the soil surface. The integrity of biological soil crusts, which are common in many drylands, is extremely sensitive to disturbance such as heavy grazing or off-road vehicle traffic (Belnap and Lange 2001). The relationship between biological soil crusts and processes of soil infiltrability is complex: their presence can increase, decrease, or have no effect on this process (Eldridge 2003; Warren 2003; Belnap 2006b). One factor that appears to determine local hydrological response is the successional stage, or status of crust development. As crusts mature, the biomass of cyanobacteria, mosses, and lichens increases—which in turn increases aggregate stability, shear strength, and roughness of the soil surface (Belnap 2003, 2006a). A six-level classification of level of crust development (LOD) was recently developed for biological soil crusts, based on (1) color (light to dark, visual assessment); (2) presence of mosses/lichens; and (3) soil surface roughness (Belnap et al. 2008). Soil crust classification was found to be strongly related to infiltration rates, with infiltration being highest where crusts were the most developed (Belnap et al. 2013).

Influence of Fauna. A recent review of ecohydrological studies revealed a strong emphasis on plant–hydrology interactions, with few studies of fauna–hydrology interactions (Westbrook et al. 2013). Only 17 % of the 339 papers reviewed considered fauna–hydrology interactions, and more than half of those focused on how hydrology affects fauna rather than how fauna function to influence ecohydrology. Fauna are usually seen as passive beneficiaries of ecohydrological changes rather than as playing a key role in the formation of vegetation patterns.

Fauna have both direct and indirect effects on ecohydrology, ranging from micro-perturbations to the macro-perturbation commonly described as ecosystem engineering (Whitford and Kay 1999; Jones et al. 2006; Butler 2007; Hastings et al. 2007; Jones 2012; Raynaud et al. 2013). These processes are critical for producing the organic matter that binds with mineral soil particles to form aggregates (peds), which facilitate the movement of water through soils and thereby increase infiltration and percolation rates and capacities (Weaver 1926; Coleman et al. 1992; Lavelle 1997; Angers and Caron 1998; Roth 2004; Jones et al. 2006). Soil fauna, particularly the mammals and macro-invertebrates (such as earthworms, termites, or cicadas), engineer ecosystems by creating openings at the soil surface and tunnels, also known as macropores, beneath the soil surface (Beven and Germann 1982; Lavelle 1997; Leonard et al. 2004; Roth 2004). These openings increase infiltration and percolation of water through the soil profile (Dean 1992; Angers and Caron 1998;

Whitford and Kay 1999; O'Farrell et al. 2010), in the same way as do the channels left by decayed plant roots (Beven and Germann 1982). Clearly, one cannot separate the roles played by animals from those played by plants; but, in combination, they significantly affect how water moves through the soil (Shafer et al. 2007)—including processes such as groundwater recharge, which in turn affect plant productivity and other ecosystem services.

Influence of Fire. The frequency and intensity of wildfires are increasing on rangelands as a result of several factors, including rising temperatures and the invasion of non-native grasses (Running 2006; Wilcox et al. 2012b). In addition, prescribed fire is now more commonly applied as a management tool for many rangelands (Twidwell et al. 2013). A number of recent reviews summarize the extensive literature on the hydrological consequences of fire on rangelands; in general, study results indicate that the infiltration capacity of soils is significantly reduced immediately following fires, but the extent of this reduction depends on fire severity, degree of hydrophobicity, antecedent soil moisture, and topographic position (Baker and Shinneman 2004; Shakesby and Doerr 2006; Pierson et al. 2011).

3.2.2 Overland Flow: Regulation at the Hillslope Scale

Water that does not infiltrate, of course, becomes overland flow or surface runoff. It is at the hillslope scale that important interactions take place between vegetation patches and runoff. Surface runoff may be captured and stored by vegetation patches or other surface obstructions, a process known as runoff–runon (Ludwig et al. 2005).

An important conceptual advance in describing and clarifying the linkages between surface runoff and vegetation patches is the trigger-transfer-reserve-pulse (TTRP) framework (Fig. 3.2) (Ludwig et al. 1997, 2005). This framework was originally proposed as a way of describing runoff–runon processes observed in areas of banded vegetation (Anderson and Hodgkinson 1997; Dunkerley and Brown 1999; Valentin and d'Herbes 1999; Tongway and Ludwig 2001); it was subsequently verified for other vegetation patch types in semiarid settings (Reid et al. 1999; Wilcox et al. 2003a; Ludwig et al. 2005). The framework assumes that the redistribution of resources from source areas (bare patches) to sink areas is a fundamental process within drylands, and that this process may be disrupted if vegetation patch structure is altered by disturbances such as overgrazing or multiyear drought. These dynamics govern how runoff and runon vary with scale in semiarid settings. In regions where runoff is efficiently captured down slope by vegetation patches, unit-area runoff and erosion diminish rapidly with increasing scale. But where vegetation patch structure has been disturbed and runoff is not efficiently captured, declines in runoff with increasing scale are much smaller (Fig. 3.3). Erosion may even increase as runoff increases with increasing scale, leading to rilling and gully formation (Wilcox et al. 2003a; Moreno de las Heras et al. 2010).
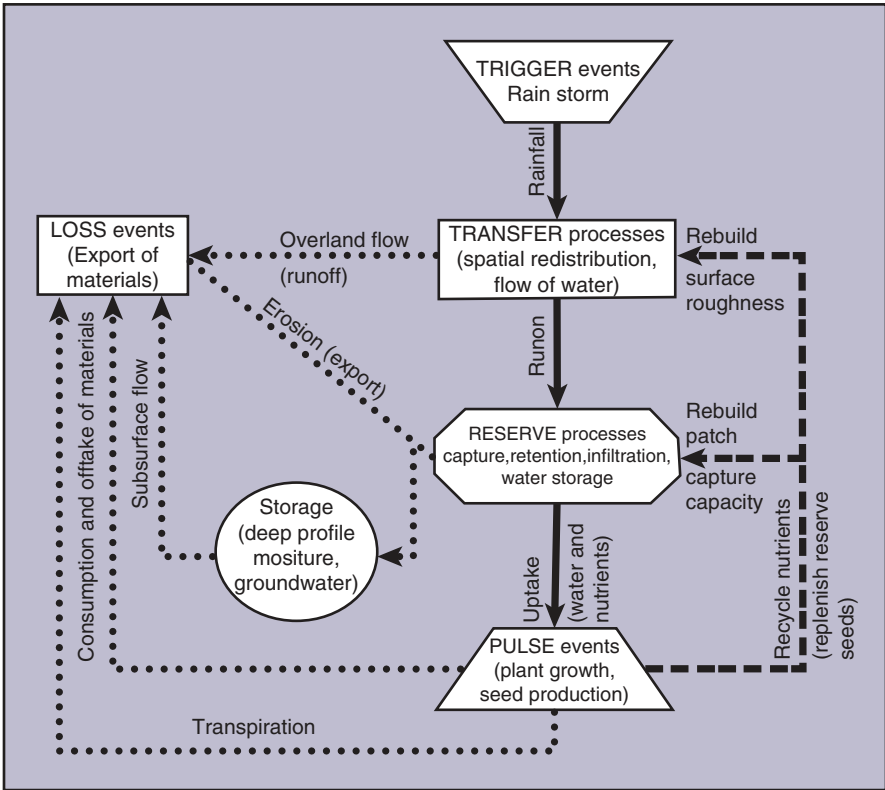
Fig. 3.2 The Trigger-Transfer-Reserve-Pulse framework illustrates how temporal events (e.g., water input from rainfall) initiate a number of other events. Solid arrows indicate direct action of flows of water, dashed arrows indicate feedbacks, and dotted arrows indicate losses (including transpiration, lateral subsurface flow, and groundwater recharge). Source: Figure 4 in Le Maitre et al. (2007)

Fig. 3.3 Hypothetical relationships demonstrating the relative changes in runoff and erosion with changes in scale and how these relationships are altered by disturbance. Source: Figure 9 in Wilcox et al. (2003)
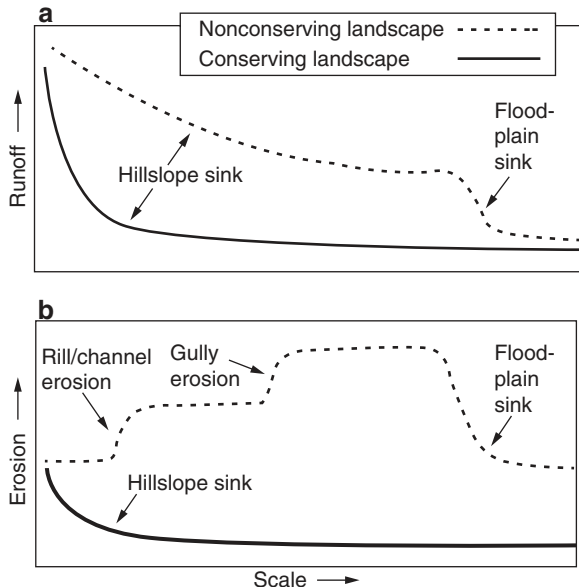
3.2.3 Drainage: Water Regulation Within the Soil

Water that enters the soil may either evaporate, be transpired by plants, or drain out of the root zone and ultimately contribute to groundwater and streamflow. On rangelands, drainage is generally (but not always) a small percentage (<5 %) of the water budget (Wilcox et al. 2003b). Vegetation management that alters the amount of woody plants may affect drainage because woody plants are deeper rooted than grasses or forbs and tend to transpire more water. Therefore, woodlands and forests generally use more water than grasslands (Zhang et al. 2001). The relationship is complex, however, especially for drylands, as it is modified by other factors— including climate, soils, and topographic position (Huxman et al. 2005).

The shrub–streamflow framework (Fig. 3.4) was developed to aid in determining which landscapes are most "hydrologically sensitive" to changes in woody plant cover. A hydrologically sensitive landscape is one in which a shift in functional vegetation type (woody to nonwoody or vice versa) causes an important shift in the water balance. Hydrological sensitivity is dictated or strongly influenced by how vegetation change affects drainage; and it is also influenced by factors such as depth to water table, soil and geological characteristics, and topographic position. The shrub–streamflow framework uses these concepts to predict where hydrologically sensitive shrublands might exist (Wilcox et al. 2006).

The first criterion for hydrologic sensitivity—the presence of shallow groundwater—is likely to be stronger where the groundwater table is within a few meters of the surface, as in riparian zones or groundwater-coupled rangelands. Obviously, this condition affords more opportunity for interaction between deep-rooted vegetation and groundwater.

Seasonality of precipitation is a second criterion in determining hydrologic sensitivity. Those rangelands having the greatest potential for water to move deeply into the soil—beneath the rooting zone of herbaceous plants—will be the most hydrologically sensitive. Such deep drainage occurs in regions where winter precipitation is high. It is no coincidence that the strongest linkage between woody plants and streamflow has been observed in Mediterranean climates where precipitation is often "out of phase" with potential ET. For example, in South Africa (van Wilgen et al. 1998), Spain (Puigdefabregas and Mendizabal 1998), Australia (Walker et al. 1993), and California (Hibbert 1983), dramatic changes in drainage have been observed following vegetation changes in native shrublands. Similarly, shrublands in which soil recharge comes mainly from snowmelt may be hydrologically sensitive; a large pulse of melting snow often produces enough water to saturate or exceed the water storage capacity of the upper soil (Baker 1984; Seyfried and Wilcox 2006).

Finally, soil or geological conditions also determine hydrologic sensitivity, by affecting the potential for deep drainage. We would expect higher hydrologic sensitivity where soils are sandy (Moore et al. 2012; Dzikiti et al. 2013), are deeply cracked (Richardson et al. 1979), or are shallow and overlie fractured bedrock (Huang et al. 2006).
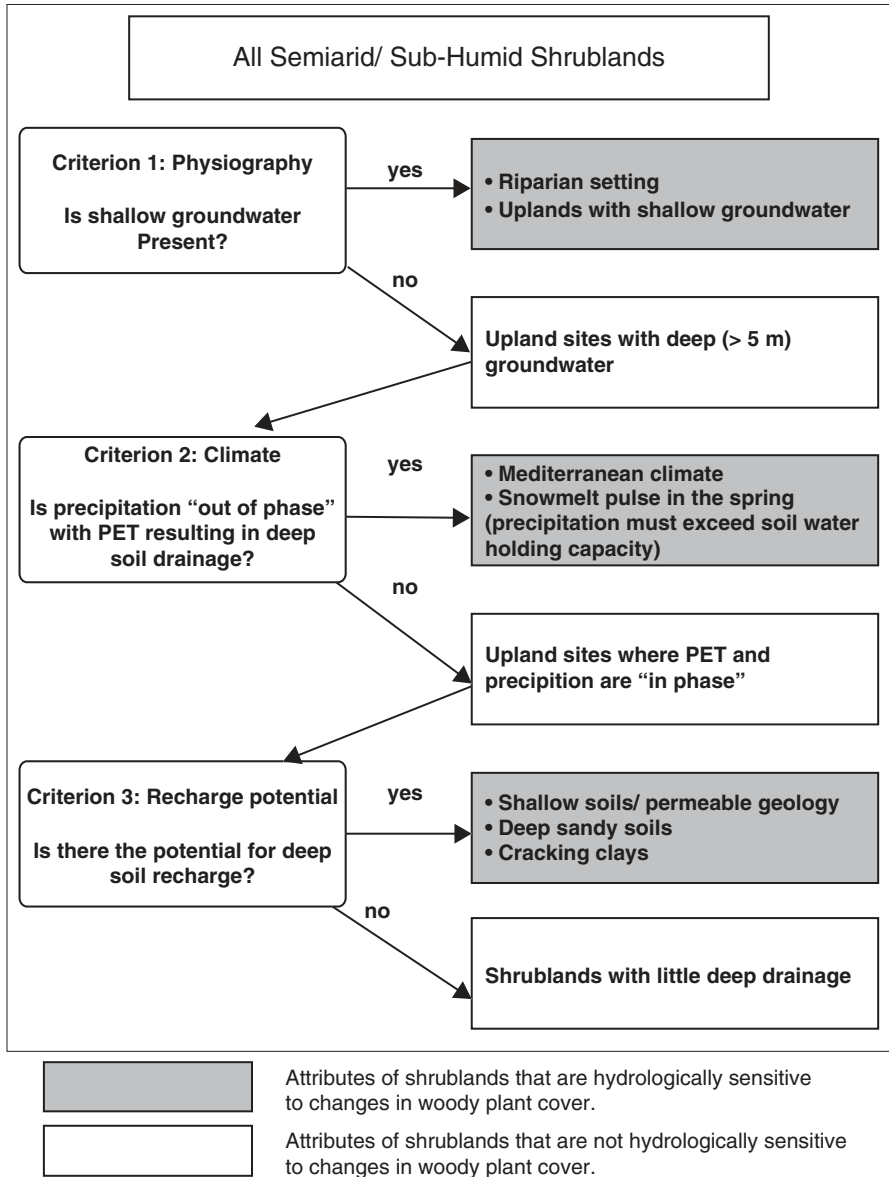
Fig. 3.4 The shrub–streamflow framework: Classification of the potential for increasing stream-flow in various shrublands. Source: Figure 1 in Wilcox et al. (2006)

For many rangelands, the opportunities for deep drainage are quite limited, because of either climate or soils. In these landscapes, shifting from a grassland to a shrubland will have little effect on the overall water balance, but may nevertheless affect drainage in important ways. For example, in areas where even small amounts of drainage can be significant (such as sites where hazardous wastes are buried), the

presence of deep-rooted shrubs may ensure that drainage below the root zone seldom occurs (Scanlon et al. 2005b, c; Seyfried et al. 2005).

3.2.4 Riparian Systems: Regulation at the Watershed Scale

The common perception is that rangelands are exclusively dryland environments. Even when the presence of riparian environments is recognized, these are generally seen as a minor component of the entire landscape system. However, there is a growing body of research showing that riparian environments are not only key habitats for rangeland fauna and flora, but also critical providers of ecosystem services to rangeland inhabitants (Milton 1990; Dean et al. 1999, 2002; Naiman et al. 2002; Sabo et al. 2005; Le Maitre et al. 2007; Soykan and Sabo 2009; Jones et al. 2010; Merritt and Bateman 2012).

Many of the features and key processes in groundwater-coupled systems are likewise found in riparian environments. The principal differences are that (1) riparian zones receive inputs of surface water, often from areas far upstream, that play a major role in their ecology (Boulton and Hancock 2006; Bunn et al. 2006; Nagler et al. 2008) and (2) the dynamics of riparian zones are strongly influenced by flow regimes and fluvial processes (Ward 1998), particularly sediment movement (Naiman et al. 1999; Tabacchi et al. 2000).

Riparian environments are typically located in the lowest parts of a landscape, where surface water (overland flow) and groundwater (subsurface flow) collect; thus they integrate outputs from all watershed-scale processes (Naiman et al. 2002). Their structure is long and narrow with a very large perimeter-to-area ratio—which makes them highly connected to, and thereby highly influenced by, events originating in the adjacent drylands. The headwaters section of a river typically has steep gradients, and the river bed contains rocks or boulders with little accumulation of fine sediments, whereas the middle and lower reaches are characterized by extensive alluvial deposits and wide floodplains (e.g., Nile, Platte, Euphrates, Ganges) (Vannote et al. 1980; Wiens 2002). These deposits are typically heterogeneous, with coarse sediments that can capture, store, and transmit large volumes of water interspersed vertically and horizontally with fine sediments that have a lower storage capacity and low transmissivity (Blasch et al. 2006; Morin et al. 2009). In humid-to-semiarid landscapes, rainfall is sufficient to generate runoff and groundwater that sustain river systems, and the rivers are gaining water, albeit seasonally. But in arid landscapes the rivers are often hydrologically disconnected from the adjacent dryland areas—except for losing water to the floodplain, and gaining water during rainy seasons or after very high rainfall events.

Water use by riparian communities has been intensively studied in the southwestern USA, mainly to estimate transmission losses, but also to quantify the effects of woody species such as the invasive or saltcedar (*Tamarix* spp.) on these losses. Evapotranspiration rates can exceed annual rainfall in these arid environments (Doody et al. 2011). Early research on saltcedar suggested that its water use was very high: up to 200 m^3 ha^{-1} day^{-1} (Sala et al. 1996); but subsequent research

has found that it is much lower and close to that of the native poplars and willows that grow in riparian systems of the southwest. Therefore, removal of these plants would have little effect on water loss if native riparian shrubs remain (Nagler et al. 2009; Shafroth et al. 2010b; Doody et al. 2011). Similar work in Australia found that although invasive *Salix* species in the river channel can use large quantities of water (±2000 mm year^{-1} vs. 1500 mm year^{-1} for open water), overall ET for this invasive species is very similar to that found for native riparian *Eucalyptus* forest (Doody and Benyon 2011; Doody et al. 2011).

In the floodplain of the perennial San Pedro River in Arizona, *Prosopis* woodlands have replaced native grasslands, increasing ET from 407 to 639 mm year^{-1} (Scott et al. 2006). Evapotranspiration from *Prosopis* woodlands in floodplains linked to perennial rivers ranges from about 350 to 750 mm year^{-1} (Scott et al. 2004, 2008), which suggests that other perennial river systems (such as those in South Africa where native tree species such as *Acacia karroo* are sparse or absent) could be similarly affected by invasion of non-native species.

The Working for Water program in South Africa, a national initiative for removal of invasive plants, emphasizes clearing to increase river flows (van Wilgen et al. 1998). Extensive invasions by *Acacia mearnsii*, Eucalyptus species, willows, and poplars have taken place along perennial rivers in the arid grasslands and savannas of the interior, where the native riparian species are mainly shrubs or small trees. If the difference between the annual ET from stands of these species and that from native species is as much as indicated by some studies (Dye and Jarmain 2004), or by data for willows and eucalypts from Doody and Benyon (2011) and Doody et al. (2011), removal could lead to an increase in river flows that would be substantial and very important for downstream water users and ecosystems. However, there may be other cases in which streamflows could be significantly reduced, such as invasions of species that are high water users along ephemeral streams (Doody et al. 2011; Hultine and Bush 2011).

The distinct species composition, structure, and dynamics of riparian environments generate a suite of ecosystem services very different from that of dryland environments. This makes them a key resource area, particularly in developing countries where they are less likely to have undergone extensive transformation by agriculture and other activities (Tockner and Stanford 2002; Kgathi et al. 2005; Brauman et al. 2007).

Recent work has also documented the extent to which large fauna can alter riparian processes. For example, in riparian ecosystems, large-scale earthworks are created mainly by the activities of fauna, particularly large mammals, which shape floodplains at a range of scales, from the microtopographical to that of river channels (Naiman and Rogers 1997; Moore 2006). Ecosystem modifications brought about by beavers, through the construction of dams, have been well studied; but much less is known about the ecological roles played by large mammals. In wetlands like the Okavango, large mammals (elephant, buffalo, hippopotamus) open up flow paths for water through reeds, changing water circulation patterns. Similarly, warthogs carve out feeding patches that form temporary pools during the wet season, creating habitats for many other species to complete their life cycles.

Riparian vegetation provides important feedbacks to the river system: it captures and stabilizes sediments, shapes river channels, and determines and regulates biotic

processes (and, thus, water quality) (Tabacchi et al. 2000; Naiman et al. 2002). By these means, the vegetation creates its own habitat as well as that for animal species, and ensures its replacement through succession. Further, riparian vegetation serves as a buffer, shielding the aquatic ecosystems from the effects of land-use practices in adjacent environments—by filtering sediments, nutrients, and other pollutants (Naiman et al. 1999; Tabacchi et al. 2000; Brauman et al. 2007; Corenblit et al. 2009).

3.2.5 Regulation in Groundwater-Coupled Rangelands

In all rangelands the recharge, transport, and quality of groundwater depend on the nature of deep drainage and solute leaching. Where groundwater tables are shallow, reciprocal interactions between vegetation and groundwater are often observed (Le Maitre et al. 1999). This two-way exchange of water and solutes increases primary and secondary production, particularly under dry climatic conditions; at the same time; however, it renders water, soil, and vegetation resources more vulnerable to land management.

Groundwater-coupled rangelands—those in which shallow water tables are found, and the potential for a strong coupling between vegetation and groundwater exists—are increasingly recognized as important, and yet are poorly understood. These ecosystems have been categorized as "groundwater dependent"; yet the degree to which they are dependent varies greatly in time and space (Boulton and Hancock 2006; Eamus and Froend 2006). For this reason, we prefer the term "groundwater coupled" to describe the broad array of rangelands characterized by shallow water tables. We do know that vegetation has a major role in regulating groundwater resources in these systems, and significant strides have been made recently in understanding these ecohydrological interactions.

Many rangeland landscapes host, at their lowest topographic points, shallow groundwater zones that are sustained by local or distant recharge sources (Tóth 1999). In dry rangelands, where evapotranspiration recycles essentially all precipitation inputs back to the atmosphere, local recharge is negligible (Scanlon et al. 2006) and such shallow aquifers are rare. Regions where they do occur are characterized by sandy or rocky soils (such as sand dunes, fractured rock outcrops), highly seasonal and intense precipitation regimes, and zones of extensive lateral flow and intense runon. In such regions, at least some deep drainage into the saturated zone will eventually take place (Scanlon and Goldsmith 1997; Athavale et al. 1998; Seyfried et al. 2005; Small 2005; Gates et al. 2008). Recharge from more distant sources is particularly significant in arid regions located downstream of water-yielding mountains. For example, shallow water tables, wetlands, and lakes fed by mountain snowmelt are found at topographic lows within sand-dune rangelands such as the Great Sand Dunes of Colorado (Wurster et al. 2003), the Bahrain Jaram and Taklamakan deserts in China (Thomas et al. 2000; Chen et al. 2004; Gates et al. 2008), and the Monte desert in Argentina (Jobbágy et al. 2011).
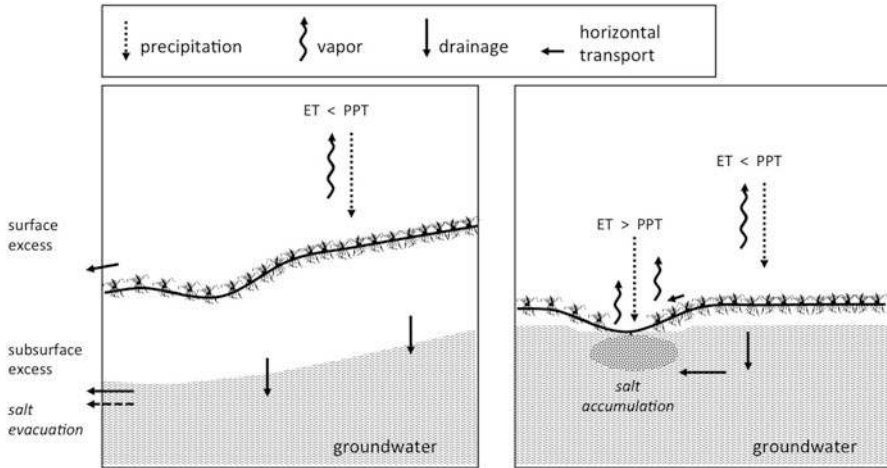
Fig. 3.5 Schematic of water balance for two landscape types in a subhumid climate. In sloped landscapes, vegetation regulates the rate of groundwater recharge. Groundwater gains that are eventually balanced by liquid discharge to streams (taking with it salts and dissolved nutrients). In flat landscapes, groundwater gains can be balanced only through higher evaporative discharge. When water leaves the landscape as vapor, its solute load is left behind. Transpiration results in salt build-up in the root zone, and direct evaporation results in salt build-up on the surface

In more humid rangelands, local groundwater recharge is more widespread and frequent, and shallow water tables are commonly found at topographic lows and along riparian corridors (Jobbágy and Jackson 2007). Finally, shallow water tables are a widespread feature in very flat, sedimentary rangelands (Fan et al. 2013) (Fig. 3.5); some conspicuous examples are the steppes of Western Siberia, the Northern Caspian plains in Asia, the grasslands of the Llanos and Pampas in South America, and the Miombo systems of sub-Saharan Africa (von der Heyden 2004).

3.2.5.1   Vegetation Dynamics Affect Groundwater Consumption

Groundwater consumption by rangeland plants is dictated by the interplay of water demand and accessibility. Most plants use groundwater in a dynamic and facultative manner, according to rainfall variability, preferring surface/shallow soil moisture when available (Engel et al. 2005). When soil moisture is low, the ability of plants to access groundwater depends, first, on the depth to the water table. Most rangeland plants access groundwater from the capillary fringe, where upflowing water and air coexist in the pore spaces of soils. Special adaptations include root aerenchyma tissue that enables species to survive and grow in saturated soils by providing air spaces to supply oxygen and facilitate groundwater consumption where water tables are very close to the surface (Visser et al. 2000). The optimum condition for plants is one in which the water table is deep enough to prevent waterlogging, but still high enough for easy access to groundwater (Jackson et al. 2009a). Groundwater use declines as water table levels drop, both in space—along topographic gradients

(Zencich et al. 2002; Gries et al. 2003; Nosetto et al. 2009)—and through time, e.g., as depth shifts seasonally (Stromberg et al. 1992; Naumburg et al. 2005; Cooper et al. 2006). Certain shrub and tree species can have maximum rooting depths of many meters (Schenk and Jackson 2002), and some observations suggest ground-water uptake from as deep as 20 m below the surface (Haase et al. 1996; Gries et al. 2003); but in dry ecosystems where water tables exceed depths of 10 m, it is rare to find significant groundwater use by plants (Zencich et al. 2002).

Other variables governing groundwater consumption by plants include water salinity and the fluid transport properties of soils and sediments. As the salinity of groundwater increases, the number of plant species able to tolerate the salinity declines, as does the rate at which the water is used. In groundwater-coupled range-lands, this relationship is commonly reflected by a series of drops in the diversity and productivity of vegetation along gradients of increasing salinity (Perelman et al. 2001). With regard to fluid transport properties, coarse-textured materials with high hydraulic conductivity favor groundwater recharge, whereas clay-dominated mate-rials limit it to negligible rates (Jobbágy and Jackson 2004b).

3.2.5.2   Land Use/Management Affects Groundwater Consumption

The way in which the vegetation and soils of groundwater-coupled rangelands are managed can have a strong influence on the exchange of water and solutes, and thereby the availability and quality of groundwater for human and livestock con-sumption, among other uses. Reciprocally, intense extraction of groundwater can significantly alter the structure and functioning of groundwater-coupled rangelands. Groundwater consumption often rises when deep-rooted woody species become abundant (Huxman et al. 2005) or when salt-tolerant species proliferate in areas where high salinity levels previously limited consumption of groundwater (Pataki et al. 2005). Conversely, where rangeland use contributes to a reduction in the den-sity of deep-rooted species the overall reduction in leaf area and transpiration decrease groundwater consumption (Meglioli et al. 2013).

This trade-off can lead to actions having different potential outcomes in different situa-tions. In the very dry, sand-dune landscapes of central Argentina, groundwater-coupled woodlands occupy less than 15 % of the land area, but represent the major source of forage for local herders. At the same time, these woodlands may consume up to 17 % of the mountain-source recharge that sustains the aquifer—which is the only local source of water for humans and livestock (Jobbágy et al. 2011). If groundwater consumption by these woodlands were to be reduced, for example through clearing of the vegetation, the actual effect on groundwater availability would be very localized and minor, but the nega-tive effect on forage availability and the herding economy would be huge.

A contrasting example comes from groundwater-coupled rangelands along river banks in the southwestern USA; here, mesquite encroachment has resulted in a dou-bling of groundwater consumption—producing more biomass, but with little benefit to livestock production (Scott et al. 2006). Finally, there are situations in which groundwater consumption can be a desirable factor in hydrological regulation. In many rangelands in Australia, the removal of native vegetation for cultivation led

to massive waterlogging and salinization of the soils (Turner and Ward 2002). The only means of reversing this process has been reforestation of large areas of the watershed (Barrett-Lennard 2002; Asseng et al. 2010)—the biomass gains and consequent water losses to lower the water table in this case both bringing benefits.

Rangeland use can affect not only the amount of available groundwater, but also its quality. When plants consume groundwater, they typically filter out salts at the root surface, which then accumulate in the absorption zone (Heuperman 1999), raising groundwater salinity (Jobbágy and Jackson 2007). Salinity levels tend to stabilize once the maximum tolerance of the consuming species is reached (Nosetto et al. 2008). It should be noted that where water tables are shallow enough to connect the capillary fringe with the surface, substantial amounts of groundwater can be lost through direct evaporation. If salinity is high, evaporation can seriously damage surface soils (Lavado and Taboada 1987). To reduce direct evaporation and restore transpiration, management methods such as halting grazing and creating means for retaining surface runoff appear to be effective (Alconada et al. 1993; Chaneton and Lavado 1996).

Groundwater availability and quality can also be compromised by rangeland uses involving animals, such as livestock. For example, continual livestock trampling has worn channels in groundwater-fed meadows. If the overall slope of the ground is somewhat steep, such channeling can rapidly lower the water table, leading to shifts in rangeland composition and productivity (Loheide and Booth 2011). The quality of groundwater is often affected as well, as has been documented in corrals and homestead areas in the groundwater-coupled woodlands of central Argentina. The combined effects of denudation from overgrazing and nutrient concentration from feces and urine have switched the net groundwater flux from discharge (losing water) to recharge (gaining water), at the same time placing soluble nitrogen contaminants into the groundwater (Meglioli et al. 2013).

Groundwater-coupled rangelands in many regions have been severely affected by direct human interventions—such as intensive pumping of groundwater—greatly drawing down the water table. Some of the most dramatic examples have been documented in the Owens Lake basin in California (Elmore et al. 2006; Pritchett and Manning 2012).

3.3   Regulating Services: Climate Regulation

The water cycle in rangelands is strongly influenced by vegetation dynamics, owing in part to the tight coupling between the water, energy, and biogeochemical cycles in these systems (Noy-Meir 1973; Austin et al. 2004; Wang et al. 2009b). In rangelands where water availability is typically low, the dominant factor controlling vegetation cover and interannual variability in vegetation productivity is mean annual precipitation. The effects of rainfall on vegetation productivity have been investigated in many parts of the world, such as the western USA (Nippert et al. 2006) and northern Africa (Le Houérou and Hoste 1977). For example, shrub encroachment has been shown to change the spatial patterns of water infiltration into soils (Daryanto et al. 2013), thus affecting local water balance. In the Mojave desert in

the southwestern USA, paired lysimeter data showed that when vegetation productivity increased significantly following elevated winter precipitation, soil water storage was reduced by half, precluding drainage below the root zone (Scanlon et al. 2005a). Such vegetation-controlled soil water flow has been occurring for 10,000–15,000 years in this region (Scanlon et al. 2005a), as it most likely has in many other rangeland ecosystems across the globe. A contrasting example comes from southwestern Australia, where replacement of perennial vegetation with annual crops led to much higher groundwater recharge, which resulted in soil salinity problems (Turner and Ward 2002).

Vegetation dynamics not only influence local hydrological conditions, but they also affect local and regional climate. Recent studies have shown that invasive shrubs in rangelands modify surface energy fluxes, causing greater nighttime air temperatures near the soil surface—particularly during the winter—thus producing a positive feedback for further shrub encroachment (D'Odorico et al. 2013b). At the regional scale, the effect of vegetation changes on climate has been observed in the Sahel (West Africa); although rainfall variability in this region is mainly influenced by variations in the surface temperature of the oceans, it is also accompanied by variations in vegetation, as seen during the multi-decadal drying trend from the 1950s to the 1980s (Zeng et al. 1999; Hein and de Ridder 2006; Prince et al. 2007). Another modeling exercise showed, in addition, that vegetation dynamics in the late 1960s in the Sahel played a critical role in maintaining the drought through the following decades. The course of the drought has been marked by a forced shift from a self-sustaining wet climate equilibrium to a similarly self-sustaining, but dry climate equilibrium (Wang and Eltahir 2000). Other research has indicated the role vegetation plays in the dynamics of the West African monsoon (Zheng and Eltahir 1998; McAlpine et al. 2009).

3.4 Supporting Services: Water Cycling and Protection Against Erosion

Supporting services are those required for the production of other ecosystem services. Their effects on people are either indirect or manifest over a very long time. Examples of supporting services include soil formation, nutrient cycling, water cycling, and protection against erosion. Of these, water cycling and protection against erosion are most germane to ecohydrology.

3.4.1 Water Cycling: With a Focus on E vs. T

The cycling of water on rangelands is obviously driven by many factors, some of which have been discussed in the previous section. A fundamental factor is the process of evapotranspiration (ET), which on most rangelands accounts for more than

95 % of the water budget (Wilcox et al. 2003b). Evapotranspiration is the sum total of interception—water captured by vegetation or litter and subsequently evaporated, transpiration, and evaporation from the soil or surface of water bodies. Recently, ecohydrologists have recognized the importance of better understanding the dynamics of ET, and in particular have placed more emphasis on accurately partitioning ET into soil evaporation and transpiration (Newman et al. 2006). Soil evaporation, from an ecohydrological perspective, is not a productive use of water because it does not contribute to plant productivity and carbon sequestration, food, fiber, or fuel production (D'Odorico et al. 2013a). This insight indicates that the main focus of ecohydrology should be to develop methods for better partitioning of the green water resources (i.e., decrease soil evaporation and increase transpiration) in semiarid and subhumid landscapes (Falkenmark and Rockstrom 2004). The same insight is motivating ecohydrologists to better understand and quantify ET.

Evapotranspiration can be partitioned into three components: (1) water that is intercepted by foliage and then evaporates back to the air; (2) water that is intercepted by litter on the soil surface, infiltrates into that litter and into the soil, and then evaporates; and (3) water that infiltrates into soil, is absorbed by plants, and later transpired back to the atmosphere. An additional process, previously not taken into account, is the potential for plants to absorb foliar-intercepted rainfall (Breshears et al. 2008); this process can be important during protracted periods of water stress, allowing plants to take advantage of rainfall events that are just large enough to be intercepted, but not large enough to infiltrate soil (Loik et al. 2004; Owens et al. 2006). This process has not been fully investigated, and the degree to which it may affect multiple species of plants is not yet known.

The rate at which soil evaporation takes place depends on several variables, including soil texture, soil temperature, and near-surface wind; these in turn are affected by basic properties of rangeland structure, such as the amount and type of woody-plant canopy cover. Recently, considerable work has focused on ways to identify the linkages between vegetation characteristics, soil evaporation, and microclimates for a diverse set of rangeland vegetation types—including mesquite, piñon-juniper, ponderosa pine, eucalypt, and saguaro cactus (Breshears and Ludwig 2010; Royer et al. 2010; Villegas et al. 2010a, b; Zou et al. 2010; Royer et al. 2012). Other recent work has focused on understanding how changes in woody plant cover may affect the ratio of transpiration to ET (Wang et al. 2010b, 2012a).

3.4.2   Protection of Soils Against Erosion and Degradation

3.4.2.1   Understanding the Importance of Vegetation Patch Structure

Another important supporting service of healthy rangelands is that of soil protection from erosion—in other words, on healthy rangelands, soils are not eroding. The obvious reason for this is that vegetation cover is adequate. But what is adequate cover? Many rangelands, particularly in drier climates, have significant areas of

bare ground and yet are not eroding. According to Ludwig et al. (1997), Vegetation patch structure is the key: vegetation patches must be numerous enough and large enough to be able to recapture soil eroded from bare areas. In fact, the transfer of water, soil, and nutrients from bare areas (sources) to vegetated areas (sinks) is a fundamental process within drylands that may be disrupted if the vegetation patch structure is disturbed. "Resource-conserving" drylands are organized such that run-off is quickly captured by, and concentrated in, vegetation patches—minimizing the loss of resources from the landscape. Resource concentration of resources increases the efficiency of their use, which translates to higher net primary productivity and the maintenance of rangeland functionality (Stavi et al. 2009).

If a disturbance, such as overgrazing, reduces the density and size of vegetation patches, the system will become "leaky" or "nonconserving"—less efficient at trapping runoff, leading to a loss of valuable water and nutrient resources (Ludwig and Tongway 2000). A positive-feedback loop may then reinforce the degradation process: the higher runoff rates will mean less water available to plants and higher erosion rates (Davenport et al. 1998; D'Odorico et al. 2013a). This degradation cycle may proceed to the point that overland-flow runoff increases in both amount and energy, erosion increases, and plant density and production declines, and the microclimate becomes more extreme (Fig. 3.6). Recognition of these processes is important not only for understanding how rangelands retain function, but also for how to devise more effective remediation strategies (Tongway and Ludwig 1997).

3.4.2.2 Wind and Water Erosion

Erosion research on rangelands has traditionally focused on water erosion and associated fluvial processes. One key advance in recent decades is recognition of the importance of wind-driven transport (aeolian) and its linkage with water erosion (Breshears et al. 2003; Belnap et al. 2011). Aeolian processes are much better understood now, thanks to improvements in measurement methods (Zobeck et al. 2003)—including relative humidity near the soil surface (Ravi et al. 2007a), the effects of vegetation patterns, and predictions of how vegetation structure influences horizontal sediment transport (Okin and Gillette 2001). Like water erosion, aeolian sediment transport is strongly influenced by the structure and arrangement of vegetation patches (Field et al. 2012). But when a grass patch is denuded (as can be caused by overgrazing) and the soil is exposed to wind action, there is a "double-whammy" effect: not only is the potential for recapturing the sediment lost, but also the wind causes the bare patch to generate additional sediment (Field et al. 2012). In the absence of disturbance, shrublands may inherently generate more wind-derived sediment than grasslands, as they have greater surface roughness as well as less intercanopy ground cover (Breshears et al. 2009). Aeolian erosional processes may also be interrelated with fire dynamics (Ravi et al. 2007b, 2009; Field et al. 2011a).

Under future climatic conditions, in regions where precipitation may become more intense while simultaneously drought frequency and intensity increase,
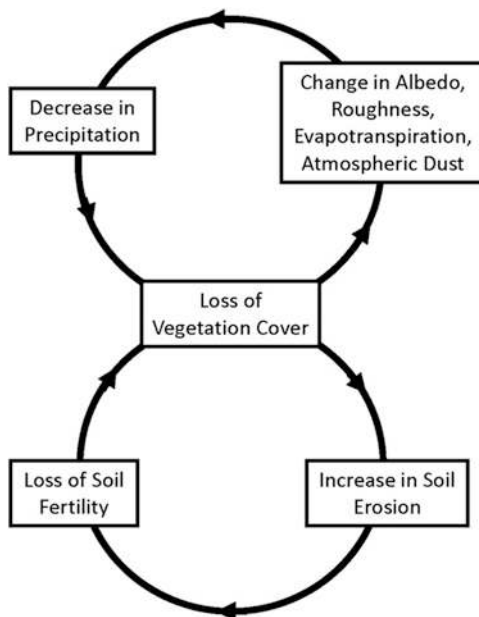
fluvial and aeolian erosion processes will be affected in different ways. A simple
but important point is that fluvial erosion occurs in rangelands only during the
infrequent precipitation events that have sufficient magnitude and intensity to
generate runoff, whereas small wind gusts occurring frequently can result in
regular, ongoing aeolian sediment transport even during less windy periods of
the year. Consequently, aeolian transport is a relatively constant and ongoing
process (Field et al. 2011b) and may even be highly interactive with fluvial pro-
cesses (Belnap et al. 2011).

3.5 Provisioning Services: Water Supply

Provisioning services are considered as those goods or products—food, fiber, and
water—that are directly produced from ecosystems. Water supply, including its
magnitude, timing, and quality, is a fundamental service provided by rangelands,
even those having relatively dry climates. Given that most rangelands are in semiarid
settings, the amount of "blue water" is generally quite low—often less than 5 %
(Wilcox et al. 2003b). Nevertheless, given the extent of rangelands, even a relatively
small fraction of blue water can translate to a considerable amount of freshwater,
which is of particularly high value in regions where the quantity is very limited. In
rangelands having more humid climates, cold and snowy climates, or rocky or very
sandy soils, "blue water" outputs can be much higher (Wilcox et al. 2006).

Water supply as a provisioning service is essentially a product of the array of regulating services discussed above. The amount of "blue water" supplied by a given landscape, i.e., water flow to streams and aquifers, is fundamentally a function of climate, vegetation, soils, and geology. Of these factors, vegetation and—to a lesser extent—soils are the most affected by rangeland management. The concept of managing vegetation for the purpose of augmenting water supply has a long, complicated, and rich history. In fact, one could argue that it is a fundamental tenet of the science and art of watershed management (Wilcox 2010). The last decade in particular has seen a considerable refinement of our understanding of the linkage between vegetation and water supply on rangelands—especially concerning the effects of removing woody plants and invasive riparian species (Huxman et al. 2005; Edwards and Roberts 2006; Shafroth et al. 2010a; Doody et al. 2011; Hultine and Bush 2011; van Wilgen et al. 2012). With respect to the USA, the issue has been reviewed in detail in Archer et al. (2011). In general, large-scale woody plant removal has not resulted in measurable increases in streamflows or groundwater recharge, although increases would have been anticipated given (1) the long experience of similar manipulations (various levels of clear-cutting) carried out in forest watersheds (Bosch and Hewlett 1982) and (2) experience with the reductions in water yield brought about by the reverse type of intervention, i.e., the establishment of tree plantations in areas that were originally treeless (Farley et al. 2005; Jobbagy et al. 2013). The only areas in which there may be a true potential for enhancing water supply through woody plant removal appears to be those having annual precipitation above 500 mm (Zhang et al. 2001) along with at least one of the following conditions: (1) predominantly winter precipitation or significant snow accumulation; (2) permeable (sandy) and deep soils; and (3) karst geology (Huxman et al. 2005).

Surprisingly, the conventional wisdom has even been overturned in the case of riparian areas invaded by alien shrubs. Until recently it was widely accepted that removal or control of invasive riparian shrubs such as Russian olive (*Elaeagnus angustifolia)* and saltcedar (*Tamarix ramosissima)* would result in substantial water savings. A recent comprehensive review on the subject, however, concluded that there is in fact little evidence for large-scale water supply augmentation through these interventions (Shafroth et al. 2010a). The primary finding was that the invasive species do not appear to use more water than the native vegetation they displace (Doody et al. 2011; Hultine and Bush 2011).

Finally, studies of groundwater recharge in the sandy deserts of Central Argentina have yielded some paradoxical results. Certain regions that are highly degraded from constant wind erosion and dune formation, with severe loss of both forage and sediment, have nevertheless seen an improvement in freshwater supply as recharge gives rise to high-quality groundwater lenses (Jobbágy et al. 2011). Except for this peculiar example of vegetation denudation proceeding in concert with gains in groundwater, the region is characterized by low-quality groundwater. In undisturbed areas it exhibits high total salt and/or arsenic content, while in disturbed areas with high animal concentrations it is less salty but polluted with nitrogen (Aranibar et al. 2011; Meglioli et al. 2014).

3.6 Observational and Conceptual Advances

As noted in previous sections, the availability and distribution of water in the landscape are of paramount importance for rangelands. Over the last few decades, a number of exciting developments have taken shape, both observational and conceptual. The former category includes *in situ* and remote-sensing monitoring tools—such as field-deployable, laser-based spectroscopy instruments that determine the ratios of hydrogen and oxygen isotopes (Lee et al. 2005; Wang et al. 2009a); portable 3D LIDAR systems for plant canopy analysis; electromagnetic imaging (EMI) devices for *in situ* soil water moisture monitoring; and distributed temperature-sensing (DTS) and remote-sensing tools, including drones and radio-controlled helicopters with lightweight digital cameras, that gather data for estimating key hydrological variables (Alsdorf et al. 2000). These and other recent developments are revolutionizing data gathering, in terms of both the scale and the precision of information used to inform ecohydrological measurement and investigation. It would be impractical to try to cover all the advances here; we have therefore selected a few that are closely related to topics already discussed in this chapter: the observational technologies of remote sensing and stable isotopes, and the conceptual advances in understanding nonlinear ecosystem behavior, scale and spatial variability, and hydrological connectivity. Discussions of other geophysical advances (e.g., EMI) can be found in the following sources (e.g., Robinson et al. 2008; Zreda et al. 2012).

3.6.1 Observational Advances

3.6.1.1 Remote Sensing for Investigating Components of the Water Budget

Remote-sensing technology has a long history in rangeland management (e.g., Prince and Tucker 1986). One of its key advantages is that it enables extrapolation not just in space, but also temporally, offering insight into change of vegetation pattern and development through time. Rapid developments in remote-sensing-based hydrological monitoring are providing unprecedented temporal and spatial coverage in estimates of hydrological variables such as rainfall, soil moisture, ET (Kustas et al. 1994; Garcia et al. 2008), surface water level (Alsdorf et al. 2000), and groundwater storage (Yeh et al. 2006).

In rangelands, the irregular spatial and temporal distribution of rainfall imposes key constraints on ecosystem function and development. Remote measurement of precipitation has an extensive history, with numerous hydrological investigations being informed by the two-decade-long Tropical Rainfall Measuring Mission (TRMM) satellite system (Kummerow et al. 2000) and related sensors. Over the coming years, the next generation of satellite rainfall-measuring systems, referred to as the Global Precipitation Measurement (GPM) mission, will provide a much-needed update to the space-based rainfall monitoring capacity. The GPM Core Observatory is in the final stages of testing at the NASA Goddard Space Flight

Center; launch is scheduled for early 2014. The resolution of spatial and temporal rainfall data derived with CPM will exceed that possible with previous designs and it will enable a much greater range of ecohydrological investigations in rangelands.

Like other water-limited systems, rangelands are characterized by a strong coupling between the dynamics of soil moisture and vegetation productivity. Soil moisture can be estimated remotely, through either active or passive microwave-based systems—each of which involves a compromise between spatial and temporal resolution. Although passive microwave sensing can be used for routine, daily global-scale estimates of soil moisture (Njoku and Entekhabi 1996), which makes it an ideal technique for large-scale studies, it also has a clear limitation: the spatial resolution of retrievals is quite coarse (approximately 25 km) (McCabe et al. 2005). Active microwave sensing provides a higher spatial resolution (up to a few kilometers), but the repeat time is generally on the order of a few days. It is possible that improved data sets for large-scale ecohydrological investigations can be obtained by merging the best features from multiple systems and sensors (e.g., Liu et al. 2011).

3.6.1.2 In Situ Methods for Measuring Components of the Water Budget

Partitioning of Evapotranspiration

Evapotranspiration is a major component of the water budget and accounts for up to 95 % of the total water input (e.g., precipitation) in rangelands (Huxman et al. 2005). It has two distinct constituents (E and T), which are controlled by different mechanisms. Partitioning of ET is important not only for better understanding the water budget but also for predicting the biogeochemical fluxes driven by hydrological variations (Wang et al. 2010a). Efficient use of the limited water resources in rangelands requires maximizing the productive water loss (T) and minimizing the unproductive water loss (E) (Wang and D'Odorico 2008). Separating E from T, however, has always been a difficult task—especially from the observational point of view at larger scales.

A useful tool for separating E from T is stable isotopes of water, because E and T carry distinct isotopic signatures. Traditionally, the stable isotopic compositions of water samples are measured by isotope ratio mass spectrometry (IRMS), while the vapor-phase measurements are based on cryogenic water vapor collection coupled with IRMS. Such methods are labor intensive and time consuming. Over the past decade, a revolutionary change has taken place in water isotope measurement: the appearance of spectroscopy-based instruments capable of continuously measuring water vapor isotopic compositions (Fig. 3.7) (Lee et al. 2005; Wen et al. 2008; Wang et al. 2009a; Griffis et al. 2010).

Monitoring of Soil Moisture

Perhaps the most important recent innovation for measuring soil moisture *in situ* is the COSMOS monitoring system (Zreda et al. 2012). Based on both the release of fast and slow neutrons from interactions between water in the soil column and a
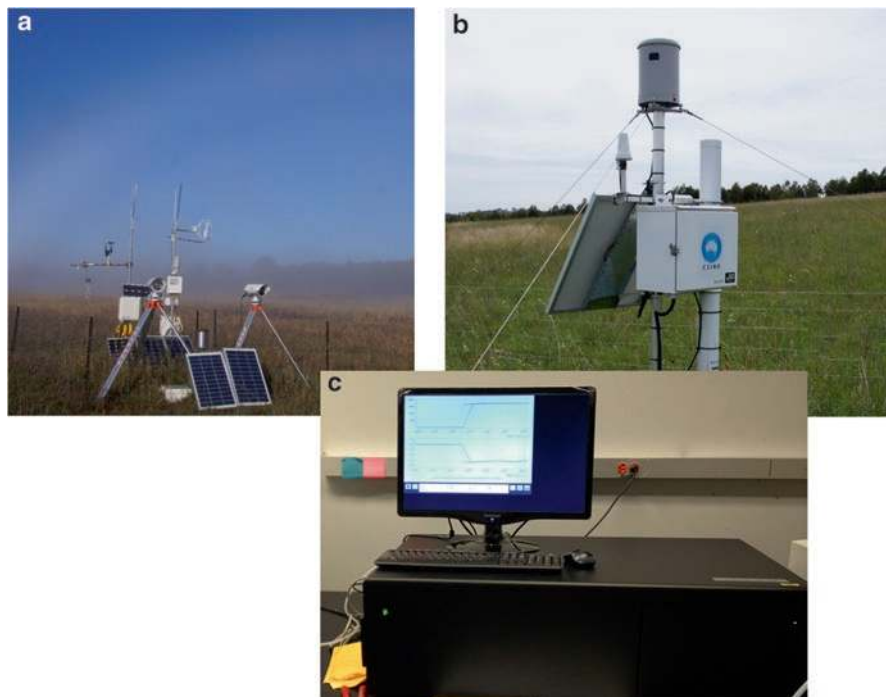
Fig. 3.7 Examples of recent advances in hydrological monitoring technology. (A) Eddy covariance system and scintillometer for ecosystem-scale measurements of sensible heat flux (evapotranspiration); (B) COSMOS system for monitoring ecosystem-scale soil moisture; (C) spectroscopy-based instrument for measuring the isotopic composition of water vapor in situ, which can be used in applications such as partitioning of evapotranspiration

regular flux of cosmic rays from space, the COSMOS system provides, for the first time, a reliable estimate of the soil wetness in a system. In addition, because the hydrogen in the top layer will have more sensitivity to the neutron counts, COSMOS, combined with modeling to separate the various hydrogen pools in the average measurement, has the potential to discriminate between moisture in the topsoil and that in the subsoil. The COSMOS installations are revolutionary in terms of bridging the spatial divide that often exists between remote-sensing and *in situ* measurement approaches. If a network of these systems can be distributed globally, our ability to monitor ecosystem change and development will be markedly improved.

3.7   Conceptual Advances

The last quarter century has seen considerable advances in our conceptual understanding of ecohydrological processes and interactions, particularly in regard to (1) spatial variability and scale, (2) ecosystem thresholds and feedbacks, and (3) hydrological connectivity of landscapes.

3.7.1   Spatial Variability and Scale

Understanding spatial dynamics and scale relationships has been a formidable challenge in both ecology and hydrology and, by extension, ecohydrology (Wood et al. 1990; Sivapalan and Kalma 1995; Sposito 1998; Grayson and Bloschl 2000; Western et al. 2001). Nevertheless, important advances have been made (Newman et al. 2006; Asbjornsen et al. 2011). For example, comparative studies across spatial scales have revealed the nonlinear nature of runoff and erosion with changing scales and how disturbance alters these relationships (Fig. 3.3) (Bergkamp 1998a; Puigdefabregas et al. 1999; Wilcox et al. 2003a; Favreau et al. 2009; Moreno de las Heras et al. 2010). Nonlinear responses in runoff and erosion are the result of redistribution across the landscape as well as alterations in runoff generation mechanisms with changing scale (Seyfried and Wilcox 1995).

Similarly significant strides have been made in quantifying the spatial variability of infiltration at the hillslope scale (Berndtsson and Larson 1987; Seyfried 1991; Pierson et al. 1994, 2001; Bhark and Small 2003; Daryanto et al. 2013). Infiltration capacities are generally higher under shrub canopies than in intercanopy areas, and these differences markedly influence patterns of soil moisture (Breshears and Barnes 1994, 1999). In addition, runon from intercanopy patches often contributes additional water to the shrub patches (Ludwig et al. 2005). Correspondingly, a number of conceptual advances have enhanced our understanding of the spatial variability of vegetation patterns on rangelands and how these are regulated by rainfall and runoff (HilleRisLambers et al. 2001; Rietkerk et al. 2002, 2004; Thompson et al. 2011).

Faunal activities also play an important role in structuring dryland landscapes as well. One feature of many arid landscapes is the formation of mounds, generally regularly dispersed, that range in diameter from a few meters to tens of meters. Known as *mima mounds* in the western USA, they are called *heuweltjies* in South Africa, where they cover from 14 to 25 % of the landscape (Lovegrove and Siegfried 1986, 1989; Whitford and Kay 1999). Their regular distribution is probably the result of competition among fauna for resources (Lovegrove and Siegfried 1986; Laurie 2002). Most authors agree that these enigmatic features are initiated by animals, whether mammals or invertebrates. One theory regarding the heuweltjies is that they developed over buried termite nests (Milton and Dean 1990; Moore and Picker 1991), but a recent paper argues that they are relics of shrub-clump-controlled erosion processes (Cramer et al. 2012). Whatever their origin, the accumulation of transported organic matter, softer soil, and food remains they contain increases their fertility (Midgley and Musil 1990) and infiltration rates (Dean 1992), supports a distinctive suite of plant species (Knight et al. 1989), and attracts faunal activity—digging by termite-eating mammals, burrowing by rodents and/or nesting ostrich (Lovegrove and Siegfried 1986, 1989; Milton and Dean 1990), and foraging by game and domestic livestock (Armstrong and Siegfried 1990; Kunz et al. 2012). The movement of water across and between the vegetation mosaic and the heuweltjies has not been studied to determine whether these mounds contribute

to groundwater recharge; but their higher infiltration rates (Dean 1992) suggest that their ecohydrological function may be analogous to that of the vegetation patches; that is, they may capture and filter runoff and act as foci for deep infiltration and recharge of groundwater.

The origins of the mima mounds in North America are no less controversial, but in this case small mammals (gophers) appear to be the primary drivers for the accumulation of materials (Whitford and Kay 1999; Jackson et al. 2003; Horwath and Johnson 2006; Johnson and Horwath-Burnham 2012). Whether or not that proves to be the sole explanation, these features also accumulate materials and alter the ecohydrology of the landscape. These important soil modifications justify the need for further research into the ecohydrological consequences of soil (Westbrook et al. 2013).

As noted by Vivoni (2012), our understanding of the role of scale and spatial variability in ecohydrological processes on rangelands will certainly increase in the future as remote-sensing and computational capabilities continue to progress.

3.7.2   Ecological Threshold and Feedback Mechanisms

Ecological thresholds and feedback loops are intimately related (Runyan et al. 2012; D'Odorico et al. 2013a). Threshold behavior occurs when a relatively small change in external drivers causes a disproportionally large response. A classic example of an ecological threshold is the transition between two stable states—such as the transition from a grassland or savanna to woodland or highly eroded state (D'Odorico et al. 2013a). The shift or change in state is induced and maintained by positive feedbacks that destabilize the system (Chapter 6, this volume). Examples of positive feedbacks are those between vegetation cover and (1) erosion, (2) soil moisture, and (3) climate (Runyan et al. 2012; D'Odorico et al. 2013a). The desertification feedback loop presented in D'Odorico et al. (2013a) (Fig. 3.6) illustrates these: A decrease in vegetation cover triggers the loss of water, nutrients, and soil that may as changes in albedo and evapotranspiration. All of these changes in turn create an environment that is less conducive to vegetation growth. In the last decade in particular, a considerable amount of work has been done that helps us better understand feedback loops and their important role in ecohydrological interactions (D'Odorico et al. 2007, 2012, 2013a, b; Stavi et al. 2009; Runyan et al. 2012; Turnbull et al. 2012).

3.7.3   Hydrological Connectivity

Hydrological connectivity refers to the water-mediated transfer of matter, energy, and organisms within or between elements of the hydrologic cycle (Pringle 2001). We now recognize that hydrological connectivity is essential for ecological integrity—and, more important, that activities by humans that disrupt this connectivity

(dams, interbasin water transfers, etc.) can have dramatic negative consequences (Pringle 2003). "Connectivity" can be more broadly understood as the transfer of energy, matter, and organisms by not only water but also other vectors—such as wind and animals (Peters et al. 2006; Okin et al. 2009). One of the major benefits of studying connectivity in physical processes is that it identifies cross-scale interactions. For example, how do various different stomata in individual grass leaves, when under stress (e.g., from grazing or drought), function to modify water fluxes at the landscape scale? Answering such questions, on the basis of information from smaller scales, will significantly improve our ability to make predictions at larger scales (Peters et al. 2004). Hydrological connectivity has proved useful in explaining ecohydrological patterns on at the landscape scale as previously indicated (Wainwright et al. 2011). However, quantifying connectivity among different scales is still a major challenge, owing largely to a lack of a conceptual framework and modeling approaches applicable at multiple scales (Miller et al. 2012). Analogical models, which simulate the behaviors of complex physical systems using laws and theorems known to control components of those systems, may be able to fill some of these gaps. Recently, Wang et al. (2012b) developed a conceptual framework that uses electrical circuit analogies and Thévenin's theorem to upscale ecohydrological and biogeochemical processes from point scales to watershed scales. This conceptual work, by providing a means of representing concomitant processes at both small and large spatial scales, may prove useful for multi-scale rangeland management efforts.

A number of important conceptual advances have improved our understanding of hydrological connectivity and flows—longitudinal, lateral, and vertical—within river systems as well as between river systems and landscapes, and the importance of this connectivity for river ecosystem structure, functioning, and maintenance of ecosystem services (Naiman et al. 1999; Ward et al. 2001; Wiens 2002; Caylor et al. 2004; Boulton and Hancock 2006). Combined with hydrogeomorphology, connectivity processes play a vital role in the structuring of river systems and the ecosystem services they provide (Thorp et al. 2006, 2010) (Fig. 3.8). The implication, for those involved in land management and in water resource management— two traditionally separate policy and legislative domains—is important: the two are actually inseparable (Postel and Thompson 2005). In fact, rivers are complex social–ecological systems, and if we are to ensure continued delivery of the numerous essential ecosystem services they provide, including their traditional use as water conduits, we must advance our knowledge of not only the scientific but also the social and economic aspects of managing them (Chapter 8, this volume).

3.8 Future Perspectives

The past quarter century has seen impressive advances in our understanding of ecohydrological processes on rangelands, and new research is providing a much clearer picture of water dynamics (amounts and timing of both green and blue water and

how these fluxes are affected by biota). These advances are attributable not only to the sheer number of new studies but also to the development of new observational methodologies, such as remote sensing and the use of stable isotopes. We anticipate that these advances will continue.

In addition, new conceptual and theoretical approaches, coupled with increases in computational power, have significantly improved our ability to predict and model ecohydrological processes. These approaches have and will continue to prove particularly useful for elucidating (1) spatial variability and scale, (2) ecosystem thresholds and feedbacks, and (3) hydrological connectivity of landscapes. We expect that the near future will bring further developments in all these areas, paving the way for more new and exciting insights into the ecohydrology of rangelands.

3.9   Summary

Our discussion of recent advances in the ecohydrology of rangelands has been organized around the concept of ecosystem services, especially those related to water. The fate of water in rangeland environments and, by extension, that of the flora and fauna that depend on this water are determined by conditions at three critical junctures: (1) The soil surface—will water infiltrate or run off? (2) The vadose zone—will water remain in the root zone or move beyond it? (3) The root zone—will water be transpired or evaporate?

Rangeland ecosystem services are categorized as regulating, supporting, and provisioning. Water-regulating services include those that affect the amount, timing, and quality of blue water flows. These are to a large extent determined at the first critical juncture of the water cycle—on the soil surface, where water either infiltrates or becomes overland flow, depending on the infiltrability of the soil. Soil infiltrability in turn depends on myriad factors, including vegetation, grazing intensity, brush management, fire patterns, condition of biological soil crusts, and activity by fauna. At larger scales, water-regulating services are influenced by other factors, such as the nature and structure of riparian zones and the presence of shallow groundwater aquifers. Finally, an important ecohydrological interaction that occurs at large scales is that between the land surface and the atmosphere. Climate regulation may result from feedbacks between rangeland vegetation and rainfall patterns.

Supporting services are those required for the production of other ecosystem services. Examples include the process of ET, which supports water cycling, and the

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Fig. 3.8 (Continued) Thoms and Parsons 2003); and (2) the ecological measures of food chain length (FCL), nutrient spiraling (NS), and species diversity (SpD), the first two scaled from long to short and the third from low to high. The light bar within each box is the expected median, with the shading estimating the range of conditions. The size of each arrow reflects the magnitude of lateral, longitudinal, and vertical connectivity. Source: Figure 1.1 and color plate 1 (revised) in Thorp et al. (2008)
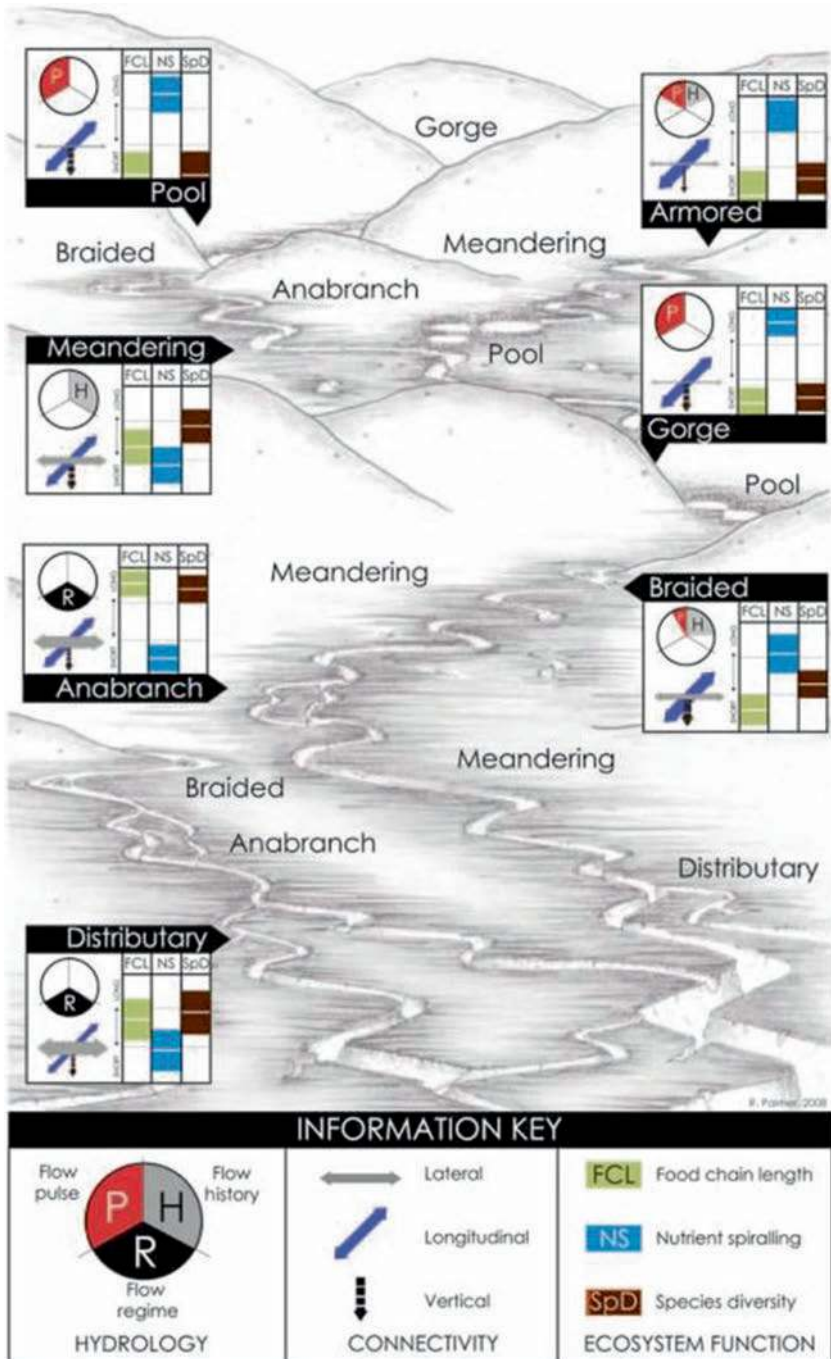
Fig. 3.8 A conceptual riverine landscape, depicting various functional process zones (FPZs) and their possible arrangement in the longitudinal dimension. Information contained in the boxes show the hydrological and ecological conditions predicted for that FPZ, including (1) the hydrological scale of greatest importance (scales being flow pulse, flow history, and flow regime, as defined by

processes by which soils are protected against erosion or degradation. The process of ET has become a subject of active inquiry in ecohydrological research—in particular, the partitioning of ET into soil evaporation and transpiration. From an ecohydrological perspective, soil evaporation is not a productive use of water because it does not contribute to plant productivity. An improved understanding of ET partitioning may lead to new management insights concerning methods for shifting more green water to productive uses. The processes that act to protect soils from erosion and/or degradation are also important ecohydrological support services. We now recognize that vegetation patch structure has a very large influence on soil erosion. Fundamentally, vegetation patches must be numerous enough and large enough to effectively capture water and sediment coming off of the bare patches. If such a patch structure is lost, rangelands begin eroding at rates that render the ecohydrological balance of the land unsustainable. Another factor affecting soil erosion, and which has been the focus of much recent research, is wind—and how it is related to water erosion. New measurement methods are yielding fresh insights into aeolian processes.

Provisioning services are those goods or products that are directly produced from ecosystems, such as water, food, and fiber. With respect to ecohydrology, the production of water from rangelands and how that production is affected by different management strategies are issues of paramount importance—but concerning which there has also been considerable misunderstanding. Work over the last several decades has largely overturned the notion that water supply can be substantially increased by removal of shrubs. Evidence of a true potential for enhancing water supply through woody plant removal has so far been found only in upland regions, and appears to be limited to those having annual precipitation above 500 mm, along with at least one of the following conditions: (1) predominantly winter precipitation or significant snow accumulation and (2) deep and permeable (sandy) soils (Huxman et al. 2005). But even where these conditions are met, in many cases the additional amount of water gained through manipulation of vegetation may be marginal. In riparian areas, surprisingly, removal of invasive, non-native woody plants appears to hold little potential for increasing water supply. Here, the primary factor appears to be that non-native plants use no more water than the native vegetation they displace.

We hope that by making an explicit linkage between ecohydrological processes and the ecosystem services concept, we have made it easier to grasp the multifaceted and complex nature of these processes on rangelands. Clearly there is a close coupling between biota (both fauna and flora) and water on rangelands—which is why water-related ecosystem services are so strongly dependent on land management strategies.

References

Alconada, M., O.E. Ansin, R.S. Lavado, V.A. Deregibus, G. Rubio, and F.H.G. Boem. 1993. Effect of retention of run-off water and grazing on soil and on vegetation of a temperate humid grassland. *Agricultural Water Management* 23: 233–246.

Alsdorf, D.E., J.M. Melack, T. Dunne, L.A.K. Mertes, L.L. Hess, and L.C. Smith. 2000. Interferometric radar measurements of water level changes on the Amazon flood plain. *Nature* 404: 174–177.

Anderson, V.J., and K.C. Hodgkinson. 1997. Grass-mediated capture of resource flows and the maintenance of banded mulga in a semi-arid woodland. *Australian Journal of Botany* 45: 331–342.

Angers, D.A., and J. Caron. 1998. Plant-induced changes in soil structure: Processes and feedbacks. *Biogeochemistry* 42: 55–72.

Aranibar, J.N., P.E. Villagra, M.L. Gomez, E. Jobbágy, M. Quiroga, R.G. Wuilloud, R.P. Monasterio, and A. Guevara. 2011. Nitrate dynamics in the soil and unconfined aquifer in arid groundwater coupled ecosystems of the Monte desert, Argentina. *Journal of Geophysical Research, Biogeosciences* 116, G04015. doi:10.1029/2010JG001618.

Archer, S. 1994. Woody plant encroachment into southwestern grasslands and savannas: Rates, patterns and proximate causes. In *Ecological implications of livestock herbivory in the west*, ed. M. Vavra, W.A. Laycock, and R.D. Pieper, 13–68. Denver, CO: Society for Range Management.

Archer, S., K.W. Davies, T.E. Fulbright, K.C. McDaniel, B.P. Wilcox, and K.I. Predick. 2011. Brush management as a rangeland conservation tool: A critical evaluation. In *Conservation effects assessment project*, ed. D.D. Briske, 105–170. Washington, DC: USDA-National Soil Conservation Service.

Armstrong, A.J., and W.R. Siegfried. 1990. Selective use of heuweltjies earth mounds by sheep in the Karoo. *South African Journal of Ecology* 1: 77–80.

Asbjornsen, H., G.R. Goldsmith, M.S. Alvarado-Barrientos, K. Rebel, F.P. Van Osch, M. Rietkerk, J.Q. Chen, S. Gotsch, C. Tobon, D.R. Geissert, A. Gomez-Tagle, K. Vache, and T.E. Dawson. 2011. Ecohydrological advances and applications in plant-water relations research: A review. *Journal of Plant Ecology* 4: 3–22.

Asseng, S., A. Dray, P. Perez, and X. Su. 2010. Rainfall–human–spatial interactions in a salinity-prone agricultural region of the Western Australian wheat-belt. *Ecological Modelling* 221: 812–824.

Athavale, R.N., R. Rangarajan, and D. Muralidharan. 1998. Influx and efflux of moisture in a desert soil during a 1 year period. *Water Resources Research* 34: 2871–2877.

Austin, A.T., L. Yahdjian, J.M. Stark, J. Belnap, A. Porporato, U. Norton, D.A. Ravetta, and S.M. Schaeffer. 2004. Water pulses and biogeochemical cycles in arid and semiarid ecosystems. *Oecologia* 141: 221–235.

Baker, M.B. 1984. Changes in streamflow in an herbicide-treated Pinyon-juniper watershed in Arizona. *Water Resources Research* 20: 1639–1642.

Baker, W.L., and D.J. Shinneman. 2004. Fire and restoration of pinon-juniper woodlands in the western United States: A review. *Forest Ecology and Management* 189: 1–21.

Barrett-Lennard, E.G. 2002. Restoration of saline land through revegetation. *Agricultural Water Management* 53: 213–226.

Belnap, J. 2003. Comparative structure of physical and biological soil crusts. In *Biological soil crusts: Structure, function, and management*, 177–192. Berlin: Springer.

———. 2006a. The potential role of biological soil crusts in dryland hydrologic cycles. *Hydrological Processes* 20(15): 3159–3178.

———. 2006b. The potential roles of biological soil crusts in dryland hydrologic cycles. *Hydrological Processes* 20: 3159–3178.

Belnap, J., and O.L. Lange (eds.). 2001. *Biological soil crusts: Structure, function, and management*. Berlin: Springer.

Belnap, J., S.L. Phillips, D.L. Witwicki, and M.E. Miller. 2008. Visually assessing the level of development and soil surface stability of cyanobacterially dominated biological soil crusts. *Journal of Arid Environments* 72: 1257–1264.

Belnap, J., S.M. Munson, and J.P. Field. 2011. Aeolian and fluvial processes in dryland regions: The need for integrated studies. *Ecohydrology* 4: 615–622.

Belnap, J., B.P. Wilcox, M.W. Van Scoyoc, and S.L. Phillips. 2013. Successional stage of biological soil crusts: An accurate indicator of ecohydrological condition. *Ecohydrology* 6: 474–482.

Bergkamp, G. 1998. A hierarchical view of the interactions of runoff and infiltration with vegetation and microtopography in semiarid shrublands. *Catena* 33: 201–220.

———. 1998b. Hydrological influences on the resilience of Quercus Spp. dominated geoecosystems in Central Spain. *Geomorphology* 23: 101–126.

Berndtsson, R., and M. Larson. 1987. Spatial variability of infiltration in a semiarid environment. *Journal of Hydrology* 90: 117–133.

Beven, K., and P. Germann. 1982. Macropores and water flow in soils. *Water Resources Research* 18: 1311–1315.

Bhark, E.W., and E.E. Small. 2003. Association between plant canopies and the spatial patterns of infiltration in shrubland and grassland of the Chihuahuan Desert, New Mexico. *Ecosystems* 6: 185–196.

Blackburn, W.H., R.W. Knight, and M.K. Wood. 1982. *Impact of grazing on watersheds: A state of knowledge*. MP 1496, Texas Agricultural Experiment Station, College Station, TX.

Blasch, K.W., T.P.A. Ferre, J.P. Hoffmann, and J.B. Fleming. 2006. Relative contributions of transient and steady state infiltration during ephemeral streamflow. *Water Resources Research* 42(8), W08405. doi:10.1029/2005WR004049.

Bosch, J.H., and J.D. Hewlett. 1982. A review of catchment experiments to determine the effect of vegetation changes and water yield and evapotranspiration. *Journal of Hydrology* 55: 3–23.

Boulton, A.J., and P.J. Hancock. 2006. Rivers as groundwater-dependent ecosystems: A review of degrees of dependency, riverine processes and management implications. *Australian Journal of Botany* 54: 133–144.

Branson, F.A., G.F. Gifford, K.G. Rennard, and R.F. Hadley. 1981. *Rangeland hydrology*. Dubuque, IA: Kendall/Hunt.

Brauman, K.A., G.C. Daily, T.K. Duarte, and H.A. Mooney. 2007. The nature and value of ecosystem services: An overview highlighting hydrologic services. *Annual Review of Environment and Resources* 32: 67–98.

Breshears, D.D., and F.J. Barnes. 1999. Interrelationships between plant functional types and soil moisture heterogeneity for semiarid landscapes within the grassland/forest continuum: A unified conceptual model. *Landscape Ecology* 14: 465–478.

———. 1994. *Spatial partitioning of water use by herbaceous and woody plants in semiarid woodlands*. Report submitted to Ecology for publication LA-UR-94-1806, Los Alamos National Laboratory, Los Alamos.

Breshears, D.D., and J.A. Ludwig. 2010. Near-ground solar radiation along the grassland-forest continuum: Tall-tree canopy architecture imposes only muted trends and heterogeneity. *Austral Ecology* 35: 31–40.

Breshears, D.D., J.J. Whicker, M.P. Johansen, and J.E. Pinder. 2003. Wind and water erosion and transport in semi-arid shrubland, grassland and forest ecosystems: Quantifying dominance of horizontal wind-driven transport. *Earth Surface Processes and Landforms* 28: 1189–1209.

Breshears, D.D., N.G. McDowell, K.L. Goddard, K.E. Dayem, S.N. Martens, C.W. Meyer, and K.M. Brown. 2008. Foliar absorption of intercepted rainfall improves woody plant water status most during drought. *Ecology* 89: 41–47.

Breshears, D.D., J.J. Whicker, C.B. Zou, J.P. Field, and C.D. Allen. 2009. A conceptual framework for dryland aeolian sediment transport along the grassland-forest continuum: Effects of woody plant canopy cover and disturbance. *Geomorphology* 105: 28–38.

Bunn, S.E., M.C. Thoms, S.K. Hamilton, and S.J. Capon. 2006. Flow variability in dryland rivers: Boom, bust and the bits in between. *River Research and Applications* 22: 179–186.

Butler, D.R. 2007. *Zoogeomorphology: Animals as geomorphic agents*. Cambridge, UK: Cambridge University Press.

Cammeraat, L.H., and A.C. Imeson. 1999. The evolution and significance of soil-vegetation patterns following land abandonment and fire in Spain. *Catena* 37: 107–127.

Caylor, K.K., T.M. Scanlon, and I. Rodriguez-Iturbe. 2004. Feasible optimality of vegetation patterns in river basins. *Geophysical Research Letters* 31(13), L13502. doi:10.1029/2004GL020260.

Chaneton, E.J., and R.S. Lavado. 1996. Soil nutrients and salinity after long-term grazing exclusion in a Flooding Pampa grassland. *Journal of Range Management* 49: 182–187.

Chen, J.S., L. Li, J.Y. Wang, D.A. Barry, X.F. Sheng, W.Z. Gu, X. Zhao, and L. Chen. 2004. Groundwater maintains dune landscape. *Nature* 432: 459–460.

Coleman, D.C., E.P. Odum, and D.A. Crossley. 1992. Soil biology, soil ecology, and global change. *Biology and Fertility of Soils* 14: 104–111.

Cooper, D.J., J.S. Sanderson, D.I. Stannard, and D.P. Groeneveld. 2006. Effects of long-term water table drawdown on evapotranspiration and vegetation in an arid region phreatophyte community. *Journal of Hydrology* 325: 21–34.

Cramer, M.D., S.N. Lnnes, and J.J. Midgley. 2012. Hard evidence that heuweltjies earth mounds are relictual features produced by differential erosion. *Palaeogeography, Palaeoclimatology, Palaeoecology* 350: 189–197.

D'Odorico, P., and A. Porporato. 2006. *Dryland ecohydrology*. Netherlands: Springer.

D'Odorico, P., K. Caylor, G.S. Okin, and T.M. Scanlon. 2007. On soil moisture-vegetation feedbacks and their possible effects on the dynamics of dryland ecosystems. *Journal of Geophysical Research, Biogeosciences* 112: G04010.

D'Odorico, P., F. Laio, A. Porporato, L. Ridolfi, A. Rinaldo, and I.R. Iturbe. 2010. Ecohydrology of terrestrial ecosystems. *BioScience* 60: 898–907.

D'Odorico, P., G.S. Okin, and B.T. Bestelmeyer. 2012. A synthetic review of feedbacks and drivers of shrub encroachment in arid grasslands. *Ecohydrology* 5: 520–530.

D'Odorico, P., A. Bhattachan, K.F. Davis, S. Ravi, and C.W. Runyan. 2013a. Global desertification: Drivers and feedbacks. *Advances in Water Resources* 51: 326–344.

D'Odorico, P., Y.F. He, S. Collins, S.F.J. De Wekker, V. Engel, and J.D. Fuentes. 2013b. Vegetation-microclimate feedbacks in woodland-grassland ecotones. *Global Ecology and Biogeography* 22: 364–379.

Daryanto, S., D.J. Eldridge, and L. Wang. 2013. Spatial patterns of infiltration vary with disturbance in a shrub-encroached woodland. *Geomorphology* 194: 57–64.

Davenport, D.W., D.D. Breshears, B.P. Wilcox, and C.D. Allen. 1998. Viewpoint—sustainability of piñon-juniper ecosystems—a unifying perspective of soil erosion thresholds. *Journal of Range Management* 51: 231–240.

Dean, W.R.J. 1992. Effects of animal activity on the absorption rate of soils in the southern Karoo, South Africa. *Journal of the Grasslands Society of South Africa* 9(4): 178–180.

Dean, W.R.J., S.J. Milton, and F. Jeltsch. 1999. Large trees, fertile islands, and birds in arid savanna. *Journal of Arid Environments* 41: 61–78.

Dean, W.R.J., M.D. Anderson, S.J. Milton, and T.A. Anderson. 2002. Avian assemblages in native Acacia and alien Prosopis drainage line woodland in the Kalahari, South Africa. *Journal of Arid Environments* 51: 1–19.

Doerr, S.H., R.A. Shakesby, and R.P.D. Walsh. 2000. Soil water repellency: Its causes, characteristics and hydro-geomorphological significance. *Earth Science Reviews* 51: 33–65.

Doody, T., and R. Benyon. 2011. Quantifying water savings from willow removal in Australian streams. *Journal of Environmental Management* 92: 926–935.

Doody, T.M., P.L. Nagler, E.P. Glenn, G.W. Moore, K. Morino, K.R. Hultine, and R.G. Benyon. 2011. Potential for water salvage by removal of non-native woody vegetation from dryland river systems. *Hydrological Processes* 25: 4117–4131.

Dortignac, E.J., and L.D. Love. 1961. *Infiltration studies on Ponderosa pine ranges of Colorado*. Station paper 59, USDA Forest Service, Fort Collins.

Dunkerley, D.L., and K.J. Brown. 1999. Banded vegetation near Broken Hill, Australia: Significance of surface roughness and soil physical properties. *Catena* 37: 75–88.

Dye, P., and C. Jarmain. 2004. Water use by black wattle (Acacia mearnsii): Implications for the link between removal of invading trees and catchment streamflow response. *South African Journal of Science* 100: 40–44.

Dyksterhuis, E.J., and E.M. Schmutz. 1947. Natural mulches or litter of grasslands: With kinds and amounts on a southern prairie. *Ecology* 28(2): 163–179.

Dzikiti, S., K. Schachtschneider, V. Naiken, M. Gush, G. Moses, and D.C. Le Maitre. 2013. Water relations and the effects of clearing invasive Prosopis trees on groundwater in an arid environment in the Northern Cape, South Africa. *Journal of Arid Environments* 90: 103–113.

Eamus, D., and R. Froend. 2006. Groundwater-dependent ecosystems: The where, what and why of GDEs. *Australian Journal of Botany* 54: 91–96.

Edwards, M.B.P., and P.J.T. Roberts. 2006. Managing forests for water: The South African experience. *International Forestry Review* 8: 65–71.

Eldridge, D.J. 2003. Biological soil crusts and water relations in Australian deserts. In *Biological soil crusts: Structure, function, and management*, ed. J. Belnap and O.L. Lange, 315–326. Berlin: Springer.

Eldridge, D.J., S. Soliveres, M.A. Bowker, and J. Val. 2013. Grazing dampens the positive effects of shrub encroachment on ecosystem functions in a semi-arid woodland. *Journal of Applied Ecology* 50: 1028–1038.

Elmore, A.J., S.J. Manning, J.F. Mustard, and J.M. Craine. 2006. Decline in alkali meadow vegetation cover in California: The effects of groundwater extraction and drought. *Journal of Applied Ecology* 43: 770–779.

Engel, V., E.G. Jobbágy, M. Stieglitz, M. Williams, and R.B. Jackson. 2005. Hydrological consequences of eucalyptus afforestation in the Argentine Pampas. *Water Resources Research* 41: W10409.

Falkenmark, M., and J. Rockstrom. 2004. *Balancing water for humans and nature: The new approach in ecohydrology*. London: Earthscan Publications.

———. 2006. The new blue and green water paradigm: Breaking new ground for water resources planning and management. *Journal of Water Resources Planning and Management: ASCE* 132: 129–132.

Falkenmark, M., J. Rockstrom, and L. Karlberg. 2009. Present and future water requirements for feeding humanity. *Food Security* 1: 59–69.

Fan, Y., H. Li, and G. Miguez-Macho. 2013. Global patterns of groundwater table depth. *Science* 339: 940–943.

Farley, K.A., E.G. Jobbágy, and R.B. Jackson. 2005. Effects of afforestation on water yield: A global synthesis with implications for policy. *Global Change Biology* 11: 1565–1576.

Favreau, G., B. Cappelaere, S. Massuel, M. Leblanc, M. Boucher, N. Boulain, and C. Leduc. 2009. Land clearing, climate variability, and water resources increase in semiarid southwest Niger: A review. *Water Resources Research* 45: W00A16.

Field, J.P., D.D. Breshears, J.J. Whicker, and C.B. Zou. 2011. Interactive effects of grazing and burning on wind- and water-driven sediment fluxes: Rangeland management implications. *Ecological Applications* 21: 22–32.

———. 2011b. On the ratio of wind- to water-driven sediment transport: Conserving soil under global-change-type extreme events. *Journal of Soil and Water Conservation* 66: 51A–56A.

———. 2012. Sediment capture by vegetation patches: Implications for desertification and increased resource redistribution. *Journal of Geophysical Research: Biogeosciences* 117: G01033.

Garcia, M., C. Oyonarte, L. Villagarcia, S. Contreras, F. Domingo, and J. Puigdefabregas. 2008. Monitoring land degradation risk using ASTER data: The non-evaporative fraction as an indicator of ecosystem function. *Remote Sensing of Environment* 112: 3720–3736.

Gates, J.B., W.M. Edmunds, J. Ma, and B.R. Scanlon. 2008. Estimating groundwater recharge in a cold desert environment in northern China using chloride. *Hydrogeology Journal* 16: 893–910.

Gifford, G.F. 1978. Hydrologic impact of grazing on infiltration: A critical review. *Water Resources Research* 14: 305–313.

Gordon, L.J., W. Steffen, B.F. Jonsson, C. Folke, M. Falkenmark, and A. Johannessen. 2005. Human modification of global water vapor flows from the land surface. *Proceedings of the National Academy of Sciences of the United States of America* 102: 7612–7617.

Grayson, R., and G. Bloschl. 2000. *Spatial patterns in catchment hydrology*. New York: Cambridge University Press.

Gries, D., F. Zeng, A. Foetzki, S.K. Arndt, H. Bruelheide, F.M. Thomas, X. Zhang, and M. Runge. 2003. Growth and water relations of Tamarix ramosissima and Populus euphratica on Taklamakan desert dunes in relation to depth to a permanent water table. *Plant, Cell and Environment* 26: 725–736.

Griffis, T.J., S.D. Sargent, X. Lee, J.M. Baker, J. Greene, M. Erickson, X. Zhang, K. Billmark, N. Schultz, W. Xiao, and N. Hu. 2010. Determining the oxygen isotope composition of evapo-transpiration using eddy covariance. *Boundary-Layer Meteorology* 137: 307–326.

Haase, P., F.I. Pugnaire, E.M. Fernández, J. Puigdefábregas, S.C. Clark, and L.D. Incoll. 1996. An investigation of rooting depth of the semiarid shrub Retama sphaerocarpa (L.) Boiss. by labelling of ground water with a chemical tracer. *Journal of Hydrology* 177: 23–31.

Hastings, A., J.E. Byers, J.A. Crooks, K. Cuddington, C.G. Jones, J.G. Lambrinos, T.S. Talley, and W.G. Wilson. 2007. Ecosystem engineering in space and time. *Ecology Letters* 10: 153–164.

Hein, L., and N. de Ridder. 2006. Desertification in the Sahel: A reinterpretation. *Global Change Biology* 12: 751–758.

Hester, J.W., T.L. Thurow, and C.A. Taylor. 1997. Hydrologic characteristics of vegetation types as affected by prescribed burning. *Journal of Range Management* 50: 199–204.

Heuperman, A. 1999. Hydraulic gradient reversal by trees in shallow water table areas and repercussions for the sustainability of tree-growing systems. *Agricultural Water Management* 39: 153–167.

Hibbert, A.R. 1983. Water yield improvement potential by vegetation management on western rangelands. *Water Resources Bulletin* 19: 375–381.

Hiernaux, P., C.L. Bielders, C. Valentin, A. Bationo, and S. Fernandez-Rivera. 1999. Effects of livestock grazing on physical and chemical properties of sandy soils in Sahelian rangelands. *Journal of Arid Environments* 41: 231–245.

HilleRisLambers, R., M. Rietkerk, F. van den Bosch, H.H.T. Prins, and H. de Kroon. 2001. Vegetation pattern formation in semi-arid grazing systems. *Ecology* 82: 50–61.

Hoff, H., M. Falkenmark, D. Gerten, L. Gordon, L. Karlberg, and J. Rockstrom. 2010. Greening the global water system. *Journal of Hydrology* 384: 177–186.

Horwath, J.L., and D.L. Johnson. 2006. Mima-type mounds in southwest Missouri: Expressions of point-centered and locally thickened biomantles. *Geomorphology* 77: 308–319.

Huang, Y., B.P. Wilcox, L. Stern, and H. Perotto-Baldivieso. 2006. Springs on rangelands: Runoff dynamics and influence of woody plant cover. *Hydrological Processes* 20: 3277–3288.

Hultine, K.R., and S.E. Bush. 2011. Ecohydrological consequences of non-native riparian vegetation in the southwestern United States: A review from an ecophysiological perspective. *Water Resources Research* 47: WO7542.

Huxman, T., B. Wilcox, D.D. Breshears, R.L. Scott, K.A. Snyder, E.E. Small, K. Hultine, J. Pittermann, and R. Jackson. 2005. Ecohydrological implications of woody plant encroachment. *Ecology* 86: 308–319.

Jackson, R.B., H.J. Schenk, E.G. Jobbágy, J. Canadell, G.D. Colello, R.E. Dickinson, C.B. Field, P. Friedlingstein, M. Heimann, K. Hibbard, D.W. Kicklighter, A. Kleidon, R.P. Neilson, W.J. Parton, O.E. Sala, and M.T. Sykes. 2000. Belowground consequences of vegetation change and their treatment in models. *Ecological Applications* 10: 470–483.

Jackson, E.C., S.N. Krogh, and W.G. Whitford. 2003. Desertification and biopedturbation in the northern Chihuahuan Desert. *Journal of Arid Environments* 53: 1–14.

———. 2009a. Ecohydrology bearings-invited commentary. Ecohydrology in a human-dominated landscape. *Ecohydrology* 2: 383–389.

———. 2009b. Ecohydrology in a human-dominated landscape. *Ecohydrology* 2: 383–389.

Jobbágy, E.G., and R.B. Jackson. 2004. Groundwater use and salinization with grassland afforestation. *Global Change Biology* 10: 1299–1312.

———. 2004b. Groundwater use and salinization with grassland afforestation. *Global Change Biology* 10: 1299–1312.

———. 2007. Groundwater and soil chemical changes under phreatophytic tree plantations. *Journal of Geophysical Research G: Biogeosciences* 112: G02013.

Jobbágy, E.G., M.D. Nosetto, P.E. Villagra, and R.B. Jackson. 2011. Water subsidies from mountains to deserts: Their role in sustaining groundwater-fed oases in a sandy landscape. *Ecological Applications* 21: 678–694.

Jobbagy, E.G., A.M. Acosta, and M.D. Nosetto. 2013. Rendimiento hídrico en cuencas primarias bajo pastizales y plantaciones de pino de las sierras de Córdoba (Argentina). *Ecología Austral* 23: 87–96.

Joffre, R., and S. Rambal. 1993. How tree cover influences the water balance of Mediterranean rangelands. *Ecology* 74: 570–582.

Johnson, D.L., and J.L. Horwath-Burnham. 2012. Introduction: Overview of concepts, definitions and principles of soil mound studies. *Geological Society of America Special Papers* 490: 1–19.

Jones, C.G. 2012. Ecosystem engineers and geomorphological signatures in landscapes. *Geomorphology* 157: 75–87.

Jones, C.G., J.L. Gutierrez, P.M. Groffman, and M. Shachak. 2006. Linking ecosystem engineers to soil processes: A framework using the Jenny State Factor Equation. *European Journal of Soil Biology* 42: S39–S53.

Jones, K.B., E.T. Slonecker, M.S. Nash, A.C. Neale, T.G. Wade, and S. Hamann. 2010. Riparian habitat changes across the continental United States (1972–2003) and potential implications for sustaining ecosystem services. *Landscape Ecology* 25: 1261–1275.

Kgathi, D.L., G. Mmopelwa, and K. Mosepele. 2005. Natural resources assessment in the Okavango Delta, Botswana: Case studies of some key resources. *Natural Resources Forum* 29: 70–81.

King, E.G., and K.K. Caylor. 2011. Ecohydrology in practice: Strengths, conveniences, and opportunities. *Ecohydrology* 4: 608–612.

Knight, R.S., A.G. Rebelo, and W.R. Siegfried. 1989. Plant assemblages on mima-like earth mounds in the Clanwilliam District. *South Africa South African Journal of Botany* 55: 465–472.

Kummerow, C., J. Simpson, O. Thiele, J. Barnes, A.T.C. Chang, E. Stocker, R.F. Adler, and A. Hou. 2000. The status of the Tropical Rainfall Measuring Mission (TRMM) after two years in orbit. *Journal of Applied Meteorology* 39: 1965–1982.

Kunz, N.S., M.T. Hoffman, and B. Weber. 2012. Effects of heuweltjies and utilization on vegetation patterns in the Succulent Karoo, South Africa. *Journal of Arid Environments* 87: 198–205.

Kustas, W., E. Perry, P. Doraiswamy, and M. Moran. 1994. Using satellite remote sensing to extrapolate evapotranspiration estimates in time and space over a semiarid rangeland basin. *Remote Sensing of Environment* 49: 275–286.

Laurie, H. 2002. Optimal transport in central place foraging, with an application to the overdispersion of heuweltjies. *South African Journal of Science* 98: 141–146.

Lavado, R.S., and M.A. Taboada. 1987. Soil salinization as an effect of grazing in a native grassland soil in the Flooding Pampa of Argentina. *Soil Use & Management* 3: 143–148.

Lavelle, P. 1997. Faunal activities and soil processes: Adaptive strategies that determine ecosystem function. In *Advances in ecological research*, eds. M. Begon, and A.H. Fitter, vol. 27, 93–132.

Le Houérou, H., and C. Hoste. 1977. Rangeland production and annual rainfall relations in the Mediterranean Basin and in the African Sahelo-Sudanian zone. *Journal of Range Management* 30: 181–189.

Le Maitre, D.C., D.F. Scott, and C. Colvin. 1999. A review of information on interactions between vegetation and groundwater. *Water SA* 25: 137–152.

Le Maitre, D.C., S.J. Milton, C. Jarmain, C.A. Colvin, I. Saayman, and J.H.J. Vlok. 2007. Linking ecosystem services and water resources: Landscape-scale hydrology of the Little Karoo. *Frontiers in Ecology and the Environment* 5: 261–270.

Lee, X., S. Sargent, R. Smith, and B. Tanner. 2005. In situ measurement of the water vapor $^{18}O/^{16}O$ isotope ratio for atmospheric and ecological applications. *Journal of Atmospheric and Oceanic Technology* 22: 555–565.

Leonard, J., E. Perrier, and J.L. Rajot. 2004. Biological macropores effect on runoff and infiltration: a combined experimental and modelling approach. *Agriculture, Ecosystems & Environment* 104: 277–285.

Liu, Y.Y., R.M. Parinussa, W.A. Dorigo, R.A.M. De Jeu, W. Wagner, A.I.J.M. Van Dijk, M.F. McCabe, and J.P. Evans. 2011. Developing an improved soil moisture dataset by blending passive and active microwave satellite-based retrievals. *Hydrology and Earth System Sciences* 15: 425–436.

Loheide, S.P., and E.G. Booth. 2011. Effects of changing channel morphology on vegetation, groundwater, and soil moisture regimes in groundwater-dependent ecosystems. *Geomorphology* 126: 364–376.

Loik, M.E., D.D. Breshears, W.K. Lauenroth, and J. Belnap. 2004. A multi-scale perspective of water pulses in dryland ecosystems: Climatology and ecohydrology of the western USA. *Oecologia* 141: 269–281.

Lovegrove, B.G., and W.R. Siegfried. 1986. Distribution and formation of Mimalike earth mounds in the western Cape province of South Africa. *South African Journal of Science* 82: 432–436.

———. 1989. Spacing and origin of mima-like mounds in the Cape province of South Africa. *South African Journal of Science* 85: 108–112.

Ludwig, J.A., and D.J. Tongway. 2000. Viewing rangelands as landscape systems. In *Rangeland desertification*, ed. O. Arnalds and S. Archer, 39–52. Dordrecht, The Netherlands: Kluwer Academic Publishers.

Ludwig, J.A., D.J. Tongway, D. Freudenberger, J. Noble, and K. Hodgkinson. 1997. *Landscape ecology function and management: Principles from Australia's rangelands*. Collingwood, Australia: CSIRO Publications.

Ludwig, J.A., R.W. Eager, R.J. Williams, and L.M. Lowe. 1999. Declines in vegetation patches, plant diversity, and grasshopper diversity near cattle watering-points in Victoria River District, northern Australia. *Rangeland Journal* 21: 135–149.

Ludwig, J.A., B.P. Wilcox, D.D. Breshears, D.J. Tongway, and A.C. Imeson. 2005. Vegetation patches and runoff-erosion as interacting ecohydrological processes in semiarid landscapes. *Ecology* 86: 288–297.

Lyford, F., and H.K. Qashu. 1969. Infiltration rates as affected by desert vegetation. *Water Resources Research* 5: 1373–1377.

Martinez-Meza, E., and W.G. Whitford. 1996. Stemflow, throughfall and channelization of stemflow by roots in three Chihuahuan Desert shrubs. *Journal of Arid Environments* 32: 271–287.

McAlpine, C.A., J. Syktus, J.G. Ryan, R.C. Deo, G.M. McKeon, H.A. McGowan, and S.R. Phinn. 2009. A continent under stress: interactions, feedbacks and risks associated with impact of modified land cover on Australia's climate. *Global Change Biology* 15: 2206–2223.

McCabe, M.F., H. Gao, and E.F. Wood. 2005. Evaluation of AMSR-E-derived soil moisture retrievals using ground-based and PSR airborne data during SMEX02. *Journal of Hydrometeorology* 6: 864–877.

Meglioli, P.A., J.N. Aranibar, P.E. Villagra, J.A. Alvarez, and E.G. Jobbágy. 2013. Livestock stations as foci of groundwater recharge and nitrate leaching in a sandy desert of the Central Monte, Argentina. *Ecohydrology*. doi: 10.1002/eco.1381.

Meglioli, P. A., J. N. Aranibar, P. E. Villagra, J. A. Alvarez, and E. G. Jobbagy. 2014. Livestock stations as foci of groundwater recharge and nitrate leaching in a sandy desert of the Central Monte, Argentina. Ecohydrology 7:600–611

Merritt, D.M., and H.L. Bateman. 2012. Linking stream flow and groundwater to avian habitat in a desert riparian system. *Ecological Applications* 22: 1973–1988.

Midgley, G.F., and C.F. Musil. 1990. Substrate effects of zoogenic soil mounds on vegetation composition in the Worchester-Robertson Valley, Cape Province. *South African Journal of Botany* 56: 158–166.

Millennium Ecosystem Assessment. 2005. *Ecosystems and human well-being: Syntheses*. Washington, DC: Island Press.

Miller, G., J. Cable, A. McDonald, B. Bond, T. Franz, L. Wang, S. Gou, A. Tyler, C. Zou, and R. Scott. 2012. Understanding ecohydrological connectivity in savannas: A system dynamics modeling approach. *Ecohydrology* 5: 200–220.

Milton, S.J. 1990. Aboveground biomass and plant cover in a succulent shrubland in the southern Karoo. *South Africa South African Journal of Botany* 56: 587–589.

Milton, S.J., and W.R.J. Dean. 1990. Mima-like mounds in the southern and western cape-are origins so mysterious. *South African Journal of Science* 86: 207–208.

Moore, J.W. 2006. Animal ecosystem engineers in streams. *BioScience* 56: 237–246.

Moore, J.M., and M.D. Picker. 1991. Heuweltjies (earth mounds) in the Clanwilliam District, Cape province, South Africa--4000 year old termite nests. *Oecologia* 86: 424–432.

Moore, G.W., D.A. Barre, and M.K. Owens. 2012. Does shrub removal increase groundwater recharge in Southwestern Texas semiarid rangelands? *Rangeland Ecology & Management* 65: 1–10.

Moran, M.S., E.P. Hamerlynck, R.L. Scott, J.J. Stone, C.D.H. Collins, T.O. Keefer, R. Bryant, L. DeYoung, G.S. Nearing, Z. Sugg, and D.C. Hymer. 2010. Hydrologic response to precipitation pulses under and between shrubs in the Chihuahuan Desert, Arizona. *Water Resources Research* 46: W10509.

Moreno de las Heras, M., J.M. Nicolau, L. Merino-Martin, and B.P. Wilcox. 2010. Plot-scale effects on runoff and erosion along a slope degradation gradient. *Water Resources Research* 46, W04503. doi:10.1029/2009WR00787.

Morin, E., T. Grodek, O. Dahan, C. Benito, C. Kulls, Y. Jacoby, G. Van Langenhove, M. Seely, and Y. Enzel. 2009. Flood routing and alluvial aquifer recharge along the ephemeral arid Kuiseb River, Namibia. *Journal of Hydrology* 368: 262–275.

Nagler, P.L., E.P. Glenn, O. Hinojosa-Huerta, F. Zamora, and K. Howard. 2008. Riparian vegetation dynamics and evapotranspiration in the riparian corridor in the delta of the Colorado River, Mexico. *Journal of Environmental Management* 88: 864–874.

Nagler, P.L., K. Morino, K. Didan, J. Erker, J. Osterberg, K.R. Hultine, and E.P. Glenn. 2009. Wide-area estimates of saltcedar (Tamarix spp.) evapotranspiration on the lower Colorado River measured by heat balance and remote sensing methods. *Ecohydrology* 2: 18–33.

Naiman, R.J., and K.H. Rogers. 1997. Large animals and system level characteristics in river corridors. *BioScience* 47: 521–529.

Naiman, R.J., S.R. Elliott, J.M. Helfield, and T.C. O'Keefe. 1999. Biophysical interactions and the structure and dynamics of riverine ecosystems: The importance of biotic feedbacks. *Hydrobiologia* 410: 79–86.

Naiman, R.J., S.E. Bunn, C. Nilsson, G.E. Petts, G. Pinay, and L.C. Thompson. 2002. Legitimizing fluvial ecosystems as users of water: An overview. *Environmental Management* 30: 455–467.

Naumburg, E., R. Mata-Gonzalez, R.G. Hunter, T. McLendon, and D.W. Martin. 2005. Phreatophytic vegetation and groundwater fluctuations: A review of current research and application of ecosystem response modeling with an emphasis on great basin vegetation. *Environmental Management* 35: 726–740.

Newman, B.D., B.P. Wilcox, S.R. Archer, D.D. Breshears, C.N. Dahm, C.J. Duffy, N.G. McDowell, F.M. Phillips, B.R. Scanlon, and E.R. Vivoni. 2006. Ecohydrology of water-limited environments: A scientific vision. *Water Resources Research* 42, W06302. doi:10.1029/2005WR004141.

Nippert, J.B., A.K. Knapp, and J.M. Briggs. 2006. Intra-annual rainfall variability and grassland productivity: Can the past predict the future? *Plant Ecology* 184: 65–74.

Njoku, E.G., and D. Entekhabi. 1996. Passive microwave remote sensing of soil moisture. *Journal of Hydrology* 184: 101–129.

Nosetto, M.D., E.G. Jobbágy, T. Toth, and R.B. Jackson. 2008. Regional patterns and controls of ecosystem salinization with grassland afforestation along a rainfall gradient. *Global Biogeochemical Cycles* 22(2), GB2015. doi:10.1029/2007GB003000.

Nosetto, M.D., E.G. Jobbágy, R.B. Jackson, and G.A. Sznaider. 2009. Reciprocal influence of crops and shallow ground water in sandy landscapes of the Inland Pampas. *Field Crops Research* 113: 138–148.

Noy-Meir, I. 1973. Desert ecosystems: Environment and producers. *Annual Review of Ecology and Systematics* 4: 25–51.

Nuttle, W.K. 2002. *Ecohydrology's past and future focus*. EOS, pp. 205–216.

O'Farrell, P.J., J.S. Donaldson, and M.T. Hoffman. 2010. Vegetation transformation, functional compensation, and soil health in a semi-arid environment. *Arid Land Research and Management* 24: 12–30.

Okin, G.S., and D.A. Gillette. 2001. Distribution of vegetation in wind-dominated landscapes: Implications for wind erosion modeling and landscape processes. *Journal of Geophysical Research-Atmospheres* 106: 9673–9683.

Okin, G.S., A.J. Parsons, J. Wainwright, J.E. Herrick, B.T. Bestelmeyer, D.C. Peters, and E.L. Fredrickson. 2009. Do changes in connectivity explain desertification? *BioScience* 59: 237–244.

Owens, M.K., R.K. Lyons, and C.L. Alejandro. 2006. Rainfall partitioning within semiarid juniper communities: Effects of event size and canopy cover. *Hydrological Processes* 20: 3179–3189.

Pataki, D.E., S.E. Bush, P. Gardner, D.K. Solomon, and J.R. Ehleringer. 2005. Ecohydrology in a Colorado River riparian forest: Implications for the decline of Populus fremontii. *Ecological Applications* 15: 1009–1018.

Perelman, S.B., R.J.C. León, and M. Oesterheld. 2001. Cross-scale vegetation patterns of Flooding Pampa grasslands. *Journal of Ecology* 89: 562–577.

Peters, D., S.R. Pielke, B. Bestelmeyer, C. Allen, S. Munson-McGee, and K. Havstad. 2004. Cross-scale interactions, nonlinearitites, and forecasting catastrophic events. *Proceedings National Academy of Sciences, U S A* 101: 15130–15135.

Peters, D.P.C., B.T. Bestelmeyer, J.E. Herrick, E.L. Fredrickson, H.C. Monger, and K.M. Havstad. 2006. Disentangling complex landscapes: New insights into arid and semiarid system dynamics. *BioScience* 56: 491–502.

Pierson, F.B., W.H. Blackburn, S.S. Vanvactor, and J.C. Wood. 1994. Partitioning small scale spatial variability of runoff and erosion on sagebrush rangeland. *Water Resources Bulletin* 30: 1081–1089.

Pierson, F.B., P.R. Robichaud, and K.E. Spaeth. 2001. Spatial and temporal effects of wildfire on the hydrology of a steep rangeland watershed. *Hydrological Processes* 15: 2905–2916.

Pierson, F.B., C.J. Williams, P.R. Kormos, S.P. Hardegree, P.E. Clark, and B.M. Rau. 2010. Hydrologic vulnerability of sagebrush steppe following pinyon and juniper encroachment. *Rangeland Ecology & Management* 63: 614–629.

Pierson, F.B., C.J. Williams, S.P. Hardegree, M.A. Weltz, J.J. Stone, and P.E. Clark. 2011. Fire, plant invasions, and erosion events on western rangelands. *Rangeland Ecology & Management* 64: 439–449.

Postel, S.L., and B.H. Thompson. 2005. Watershed protection: Capturing the benefits of nature's water supply services. *Natural Resources Forum* 29: 98–108.

Prince, S., and C. Tucker. 1986. Satellite remote sensing of rangelands in Botswana II. NOAA AVHRR and herbaceous vegetation. *International Journal of Remote Sensing* 7: 1555–1570.

Prince, S.D., K.J. Wessels, C.J. Tucker, and S.E. Nicholson. 2007. Desertification in the Sahel: A reinterpretation of a reinterpretation. *Global Change Biology* 13: 1308–1313.

Pringle, C.M. 2001. Hydrologic connectivity and the management of biological reserves: A global perspective. *Ecological Applications* 11: 981–998.

Pringle, C. 2003. What is hydrologic connectivity and why is it ecologically important? *Hydrological Processes* 17: 2685–2689.

Pritchett, D., and S.J. Manning. 2012. Response of an intermountain groundwater-dependent ecosystem to water table drawdown. *Western North American Naturalist* 72: 48–59.

Puigdefabregas, J., and T. Mendizabal. 1998. Perspectives on desertification—Western Mediterranean. *Journal of Arid Environments* 39: 209–224.

Puigdefabregas, J., A. Sole, L. Gutierrez, G. del Barrio, and M. Boer. 1999. Scales and processes of water and sediment redistribution in drylands: Results from the Rambla Honda field site in Southeast Spain. *Earth-Science Reviews* 48: 39–70.

Pyke, D.A., J.E. Herrick, P. Shaver, and M. Pellant. 2002. Rangeland health attributes and indicators for qualitative assessment. *Journal of Range Management* 55: 584–597.

Ravi, S., P. D'Odorico, and G.S. Okin. 2007a. Hydrologic and aeolian controls on vegetation patterns in arid landscapes. *Geophysical Research Letters* 34: 5. doi:10.1029/2007GL031023.

Ravi, S., P. D'Odorico, T.M. Zobeck, T.M. Over, and S.L. Collins. 2007b. Feedbacks between fires and wind erosion in heterogeneous arid lands—Art. no. G04007. *Journal of Geophysical Research: Biogeosciences* 112: 4007.

Ravi, S., P. D'Odorico, T.M. Zobeck, and T.M. Over. 2009. The effect of fire-induced soil hydrophobicity on wind erosion in a semiarid grassland: Experimental observations and theoretical framework. *Geomorphology* 105: 80–86.

Raynaud, X., C.G. Jones, and S. Barot. 2013. Ecosystem engineering, environmental decay and environmental states of landscapes. *Oikos* 122: 591–600.

Reid, K.D., B.P. Wilcox, D.D. Breshears, and L. MacDonald. 1999. Runoff and erosion in a pinon-juniper woodland: Influence of vegetation patches. *Soil Science Society of America Journal* 63: 1869–1879.

Reynolds, J.F., D.M. Stafford Smith, E.F. Lambin, B.L. Turner, M. Mortimore, S.P.J. Batterbury, T.E. Downing, H. Dowlatabadi, R.J. Fernandez, J.E. Herrick, E. Huber-Sannwald, H. Jiang, R. Leemans, T. Lynam, F.T. Maestre, M. Ayarza, and B. Walker. 2007. Global desertification: Building a science for dryland development. *Science* 316: 847–851.

Richardson, C.W., E. Burnett, and R.W. Bovey. 1979. Hydrologic effects of brush control on Texas rangelands. *Transactions of the ASAE* 22(2): 315–319.

Rietkerk, M., M.C. Boerlijst, F. van Langevelde, R. HilleRisLambers, J. van de Koppel, L. Kumar, H.H.T. Prins, and A.M. de Roos. 2002. Self-organization of vegetation in arid ecosystems. *American Naturalist* 160: 524–530.

Rietkerk, M., S.C. Dekker, P.C. de Ruiter, and J. van de Koppel. 2004. Self-organized patchiness and catastrophic shifts in ecosystems. *Science* 305: 1926–1929.

Robinson, D.A., C.S. Campbell, J.W. Hopmans, B.K. Hornbuckle, S.B. Jones, R. Knight, F. Ogden, J. Selker, and O. Wendroth. 2008. Soil moisture measurement for ecological and hydrological watershed-scale observatories: A review. *Vadose Zone Journal* 7: 358–389.

Rockstrom, J., M. Falkenmark, L. Karlberg, H. Hoff, S. Rost, and D. Gerten. 2009. Future water availability for global food production: The potential of green water for increasing resilience to global change. *Water Resources Research* 45(7), W00A12. doi:10.1029/2007WR006767.

Rodriguez-Iturbe, I., and A. Porporato. 2004. *Ecohydrology of water-controlled ecosystems: Soil moisture and plant dynamics*. Cambridge: Cambridge University Press.

Roth, C.H. 2004. A framework relating soil surface condition to infiltration and sediment and nutrient mobilization in grazed rangelands of northeastern Queensland, Australia. *Earth Surface Processes and Landforms* 29: 1093–1104.

Royer, P.D., D.D. Breshears, C.B. Zou, N.S. Cobb, and S.A. Kurc. 2010. Ecohydrological energy inputs in semiarid coniferous gradients: Responses to management- and drought-induced tree reductions. *Forest Ecology and Management* 260: 1646–1655.

Royer, P.D., D.D. Breshears, C.B. Zou, J.C. Villegas, N.S. Cobb, and S.A. Kurc. 2012. Density-dependent ecohydrological effects of pinon–juniper woody canopy cover on soil microclimate and potential soil evaporation. *Rangeland Ecology & Management* 65: 11–20.

Running, S.W. 2006. Is global warming causing more, larger wildfires? *Science* 313: 927–928.

Runyan, C.W., P. D'Odorico, and D. Lawrence. 2012. Physical and biological feedbacks of deforestation. *Reviews of Geophysics* 50, RG4006.

Sabo, J.L., R. Sponseller, M. Dixon, K. Gade, T. Harms, J. Heffernan, A. Jani, G. Katz, C. Soykan, J. Watts, and A. Welter. 2005. Riparian zones increase regional species richness by harboring different, not more, species. *Ecology* 86: 56–62.

Saco, P.M., and M. Moreno de las Heras. 2013. Ecogeomorphic coevolution of semiarid hillslopes: Emergence of banded and striped vegetation patterns through interaction of biotic and abiotic processes. *Water Resources Research* 49: 115–126.

Sala, A., S.D. Smith, and D.A. Devitt. 1996. Water use by Tamarix ramosissima and associated phreatophytes in a Mojave Desert floodplain. *Ecological Applications* 6: 888–898.

Savadogo, P., L. Sawadogo, and D. Tiveau. 2007. Effects of grazing intensity and prescribed fire on soil physical and hydrological properties and pasture yield in the savanna woodlands of Burkina Faso. *Agriculture, Ecosystems & Environment* 118: 80–92.

Scanlon, B.R., and R.S. Goldsmith. 1997. Field study of spatial variability in unsaturated flow beneath and adjacent to playas. *Water Resources Research* 33: 2239–2252.

Scanlon, B.R., D.G. Levitt, R.C. Reedy, K.E. Keese, and M.J. Sully. 2005a. Ecological controls on water-cycle response to climate variability in deserts. *Proceedings of the National Academy of Sciences* 102: 6033–6038.

Scanlon, B.R., D.G. Levitt, R.C. Reedy, K.E. Keese, and M.J. Sully. 2005b. Ecological controls on water-cycle response to climate variability in deserts. *Proceedings of the National Academy of Sciences of the United States of America* 102: 6033–6038.

Scanlon, B.R., R.C. Reedy, K.E. Keese, and S.F. Dwyer. 2005c. Evaluation of evapotranspirative covers for waste containment in arid and semiarid regions in the southwestern USA. *Vadose Zone Journal* 4: 55–71.

Scanlon, B.R., K.E. Keese, A.L. Flint, L.E. Flint, C.B. Gaye, W.M. Edmunds, and I. Simmers. 2006. Global synthesis of groundwater recharge in semiarid and arid regions. *Hydrological Processes* 20: 3335–3370.

Schenk, H.J., and R.B. Jackson. 2002. Rooting depths, lateral root spreads and below-ground/above-ground allometries of plants in water-limited ecosystems. *Journal of Ecology* 90: 480–494.

Schlesinger, W.H., A.D. Abrahams, A.J. Parsons, and J. Wainwright. 1999. Nutrient losses in runoff from grassland and shrubland habitats in Southern New Mexico: I. Rainfall simulation experiments. *Biogeochemistry* 45: 21–34.

Scott, R.L., E.A. Edwards, W.J. Shuttleworth, T.E. Huxman, C. Watts, and D.C. Goodrich. 2004. Interannual and seasonal variation in fluxes of water and carbon dioxide from a riparian woodland ecosystem. *Agricultural and Forest Meteorology* 122: 65–84.

Scott, R.L., T.E. Huxman, D.G. Williams, and D.C. Goodrich. 2006. Ecohydrological impacts of woody-plant encroachment: seasonal patterns of water and carbon dioxide exchange within a semiarid riparian environment. *Global Change Biology* 12: 311–324.

Scott, R.L., W.L. Cable, T.E. Huxman, P.L. Nagler, M. Hernandez, and D.C. Goodrich. 2008. Multiyear riparian evapotranspiration and groundwater use for a semiarid watershed. *Journal of Arid Environments* 72: 1232–1246.

Seyfried, M.S. 1991. Infiltration patterns from simulated rainfall on a semiarid rangeland soil. *Soil Science Society of America Journal* 55: 1726–1734.

Seyfried, M.S., and B.P. Wilcox. 1995. Scale and the nature of spatial variability: Field examples having implications for hydrologic modeling. *Water Resources Research* 1: 173–184.

———. 2006. Soil water storage and rooting depth: key factors controlling recharge on rangelands. *Hydrological Processes* 20: 3261–3275.

Seyfried, M.S., S. Schwinning, M.A. Walvoord, W.T. Pockman, B.D. Newman, R.B. Jackson, and E.M. Phillips. 2005. Ecohydrological control of deep drainage in arid and semiarid regions. *Ecology* 86: 277–287.

Shafer, D.S., M.H. Young, S.F. Zitzer, T.G. Caldwell, and E.V. McDonald. 2007. Impacts of inter-related biotic and abiotic processes during the past 125,000 years of landscape evolution in the Northern Mojave Desert, Nevada, USA. *Journal of Arid Environments* 69: 633–657.

Shafroth, P.B., C.A. Brown, and D.M. Merrit, eds. 2010a. *Saltcedar and Russian olive control demonstration act science assessment*. U.G. Geological Survey.

Shafroth, P.B., A.C. Wilcox, D.A. Lytle, J.T. Hickey, D.C. Andersen, V.B. Beauchamp, A. Hautzinger, L.E. McMullen, and A. Warner. 2010a. Ecosystem effects of environmental flows: Modelling and experimental floods in a dryland river. *Freshwater Biology* 55: 68–85.

Shakesby, R.A., and S.H. Doerr. 2006. Wildfire as a hydrological and geomorphological agent. *Earth-Science Reviews* 74: 269–307.

Sivapalan, M., and J.D. Kalma. 1995. Scale problems in hydrology—Contributions of the Robertson workshop. *Hydrological Processes* 9: 243–250.

Small, E.E. 2005. Climatic controls on diffuse groundwater recharge in semiarid environments of the southwestern United States. *Water Resources Research* 41: 1–17.

Smith, H.L., and L.B. Leopold. 1941. Infiltration studies in the Pecos River Watershed, New Mexico and Texas. *Soil Science* 53: 195–204.

Snyman, H.A., and C.C. du Preez. 2005. Rangeland degradation in a semi-arid South Africa— II: Influence on soil quality. *Journal of Arid Environments* 60: 483–507.

Soykan, C.U., and J.L. Sabo. 2009. Spatiotemporal food web dynamics along a desert riparian-upland transition. *Ecography* 32: 354–368.

Sposito, G. 1998. *Scale dependence and scale invariance in hydrology*. New York: Cambridge University Press.

Stavi, I., H. Lavee, E.D. Ungar, and P. Sarah. 2009. Ecogeomorphic feedbacks in semiarid rangelands: A review. *Pedosphere* 19: 217–229.

Stromberg, J.C., J.A. Tress, S.D. Wilkins, and S.D. Clark. 1992. Response of velvet mesquite to groundwater decline. *Journal of Arid Environments* 23: 45–58.

Tabacchi, E., L. Lambs, H. Guilloy, A.M. Planty-Tabacchi, E. Muller, and H. Decamps. 2000. Impacts of riparian vegetation on hydrological processes. *Hydrological Processes* 14: 2959–2976.

Thomas, F.M., S.K. Arndt, H. Bruelheide, A. Foetzki, D. Gries, J. Huang, M. Popp, G. Wang, X. Zhang, and M. Runge. 2000. Ecological basis for a sustainable management of the indigenous vegetation in a Central-Asian desert: Presentation and first results. *Journal of Applied Botany* 74: 212–219.

Thompson, S.E., C.J. Harman, P.A. Troch, P.D. Brooks, and M. Sivapalan. 2011. Spatial scale dependence of ecohydrologically mediated water balance partitioning: A synthesis framework for catchment ecohydrology. *Water Resources Research* 47, W00J03. doi:10.1029/201 0WR009998.

Thorp, J.H., M.C. Thoms, and M.D. Delong. 2006. The riverine ecosystem synthesis: Biocomplexity in river networks across space and time. *River Research and Applications* 22: 123–147.

Thorp, J.H., J.E. Flotemersch, M.D. Delong, A.F. Casper, M.C. Thoms, F. Ballantyne, B.S. Williams, B.J. O'Neill, and C.S. Haase. 2010. Linking ecosystem services, rehabilitation, and river hydrogeomorphology. *BioScience* 60: 67–74.

Tockner, K., and J.A. Stanford. 2002. Riverine flood plains: Present state and future trends. *Environmental Conservation* 29: 308–330.

Tongway, D.J., and J.A. Ludwig. 1997. Chapter 5. The nature of landscape dysfunction in rangelands. In *Landscape ecology, function and management: Principles from Australia's rangelands*, eds. J. Ludwig, D. Tongway, D. Freudenberger, J. Noble, and K. Hodgkinson, 49–61. Collingwood, VIC: CSIRO Publishing.

———. 2001. Theories on the origins, maintenance, dynamics and functioning of banded landscapes. In *Banded vegetation patterning in arid and semiarid environments: Ecological processes and consequences of management*, eds. D.J. Tongway, C. Valentin, and J. Segheri, 20–31. New York, NY: Springer.

Tóth, J. 1999. Groundwater as a geologic agent: An overview of the causes, processes, and manifestations. *Hydrogeology Journal* 7: 1–14.

Trimble, S.W., and A.C. Mendel. 1995. The cow as a geomorphic agent—A critical-review. *Geomorphology* 13: 233–253.

Turnbull, L., B.P. Wilcox, J. Belnap, S. Ravi, P. D'Odorico, D. Childers, W. Gwenzi, G. Okin, J. Wainwright, K.K. Caylor, and T. Sankey. 2012. Understanding the role of ecohydrological feedbacks in ecosystem state change in drylands. *Ecohydrology* 5: 174–183.

Turner, N.C., and P.R. Ward. 2002. The role of agroforestry and perennial pasture in mitigating water logging and secondary salinity: Summary. *Agricultural Water Management* 53: 271–275.

Twidwell, D., W.E. Rogers, S.D. Fuhlendorf, C.L. Wonkka, D.M. Engle, J.R. Weir, U.P. Kreuter, and C.A. Taylor. 2013. The rising Great Plains fire campaign: Citizens response to woody plant encroachment. *Frontiers in Ecology and the Environment* 11: 64–71.

Valentin, C., and J.M. d'Herbes. 1999. Niger tiger bush as a natural water harvesting system. *Catena* 37: 231–256.

van Wilgen, B.W., D.C. Le Maitre, and R.M. Cowling. 1998. Ecosystem services, efficiency, sustainability and equity: South Africa's Working for Water programme. *Trends in Ecology & Evolution* 13: 378–378.

van Wilgen, B.W., G.G. Forsyth, D.C. Le Maitre, A. Wannenburgh, J.D.F. Kotze, E. van den Berg, and L. Henderson. 2012. An assessment of the effectiveness of a large, national-scale invasive alien plant control strategy in South Africa. *Biological Conservation* 148: 28–38.

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. River continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences* 37: 130–137.

Villegas, J.C., D.D. Breshears, C.B. Zou, and D.J. Law. 2010a. Ecohydrological controls of soil evaporation in deciduous drylands: How the hierarchical effects of litter, patch and vegetation mosaic cover interact with phenology and season. *Journal of Arid Environments* 74: 595–602.

Villegas, J.C., D.D. Breshears, C.B. Zou, and P.D. Royer. 2010b. Seasonally pulsed heterogeneity in microclimate: Phenology and cover effects along deciduous grassland-forest continuum. *Vadose Zone Journal* 9: 537–547.

Visser, E.J.W., G.M. Bogemann, H.M. Van de Steeg, R. Pierik, and C.W.P.M. Blom. 2000. Flooding tolerance of Carex species in relation to field distribution and aerenchyma formation. *New Phytologist* 148: 93–103.

Vivoni, E.R. 2012. Spatial patterns, processes and predictions in ecohydrology: Integrating technologies to meet the challenge. *Ecohydrology* 5: 235–241.

von der Heyden, C.J. 2004. The hydrology and hydrogeology of dambos: A review. *Progress in Physical Geography* 28: 544–564.

Wainwright, J., L. Turnbull, T.G. Ibrahim, I. Lexartza-Artza, S.F. Thornton, and R.E. Brazier. 2011. Linking environmental regimes, space and time: Interpretations of structural and functional connectivity. *Geomorphology* 126: 387–404.

Walker, J., F. Bullen, and B.G. Williams. 1993. Ecohydrological changes in the Murray-Darling Basin. I. The number of trees cleared over two centuries. *Journal of Applied Ecology* 30: 265–273.

Wang, L., and P. D'Odorico. 2008. The limits of water pumps. *Science* 321: 36–37.

Wang, G., and E.A. Eltahir. 2000. Ecosystem dynamics and the Sahel drought. *Geophysical Research Letters* 27: 795–798.

Wang, L., K. Caylor, and D. Dragoni. 2009a. On the calibration of continuous, high-precision δ18O and δ2H measurements using an off-axis integrated cavity output spectrometer. *Rapid Communications in Mass Spectrometry* 23: 530–536.

Wang, L., P. D'Odorico, S. Manzoni, A. Porporato, and S. Macko. 2009b. Carbon and nitrogen dynamics in southern African savannas: The effect of vegetation-induced patch-scale heterogeneities and large scale rainfall gradients. *Climatic Change* 94: 63–76.

Wang, L., K.K. Caylor, J.C. Villegas, G.A. Barron-Gafford, D.D. Breshears, and T.E. Huxman. 2010a. Evapotranspiration partitioning with woody plant cover: Assessment of a stable isotope technique. *Geophysical Research Letters* 37, L09401.

Wang, L.X., K.K. Caylor, J.C. Villegas, G.A. Barron-Gafford, D.D. Breshears, and T.E. Huxman. 2010b. Partitioning evapotranspiration across gradients of woody plant cover: Assessment of a stable isotope technique. *Geophysical Research Letters* 37, L09401. doi:10.1029/2010GL043228.

Wang, L., P. D'Odorico, J.P. Evans, D.J. Eldridge, M.F. McCabe, K.K. Caylor, and E.G. King. 2012a. Dryland ecohydrology and climate change: Critical issues and technical advances. *Hydrology and Earth System Sciences* 16: 2585–2603.

Wang, L., C. Zou, F. O'Donnell, S. Good, T. Franz, G. Milller, K. Caylor, J. Cable, and B. Bond. 2012b. Characterizing ecohydrological and biogeochemical connectivity across multiple scales: A new conceptual framework. *Ecohydrology* 5: 221–233. doi:10.1002/eco.1187.

Ward, J.V. 1998. Riverine landscapes: Biodiversity patterns, disturbance regimes, and aquatic conservation. *Biological Conservation* 83: 269–278.

Ward, J.V., F. Malard, and K. Tockner. 2001. Landscape ecology: A framework for integrating pattern and process in river corridors. *Landscape Ecology* 17: 35–45.

Warren, S.D. 2003. Synopsis: Influence of biological soil crusts on arid land hydrology and soil stability. In *Biological soil crusts: Structure, function, and management*, ed. J. Belnap and O.L. Lange, 349–362. Berlin: Springer.

Weaver, J. 1926. *Root development of field crops*. New York: McGraw-Hill Book Company.

Wen, X., X. Sun, S. Zhang, G. Yu, S. Sargent, and X. Lee. 2008. Continuous measurement of water vapor D/H and ^{18}O/^{16}O isotope ratios in the atmosphere. *Journal of Hydrology* 349: 489–500.

Westbrook, C.J., W. Veatch, and A. Morrison. 2013. Ecohydrology bearings—Invited commentary is ecohydrology missing much of the zoo? *Ecohydrology* 6: 1–7.

Western, A.W., G. Bloschl, and R.B. Grayson. 2001. Toward capturing hydrologically significant connectivity in spatial patterns. *Water Resources Research* 37: 83–97.

Whitford, W.G., and F.R. Kay. 1999. Biopedturbation by mammals in deserts: A review. *Journal of Arid Environments* 41: 203–230.

Wiens, J.A. 2002. Riverine landscapes: Taking landscape ecology into the water. *Freshwater Biology* 47: 501–515.

Wilcox, B.P. 2002. Shrub control and streamflow on rangelands: A process based viewpoint. *Journal of Range Management* 55: 318–326.

———. 2010. Transformative ecosystem change and ecohydrology: Ushering in a new era for watershed management. *Ecohydrology* 3: 126–130.

Wilcox, B.P., and B.D. Newman. 2005. Ecohydrology of semiarid landscapes. *Ecology* 86: 275–276.

Wilcox, B.P., D.D. Breshears, and C.D. Allen. 2003a. Ecohydrology of a resource-conserving semiarid woodland: Effects of scale and disturbance. *Ecological Monographs* 73: 223–239.

Wilcox, B.P., M.S. Seyfried, and D.D. Breshears. 2003b. The water balance on rangelands. *Encyclopedia of Water Science* 791–795.

Wilcox, B.P., M.K. Owens, W.A. Dugas, D.N. Ueckert, and C.R. Hart. 2006. Shrubs, streamflow, and the paradox of scale. *Hydrological Processes* 20: 3245–3259.

Wilcox, B.P., P.I. Taucer, C.L. Munster, M.K. Owens, B.P. Mohanty, J.R. Sorenson, and R. Bazan. 2008. Subsurface stormflow is important in semiarid karst shrublands. *Geophysical Research Letters* 35, L10403.

Wilcox, B.P., M.G. Sorice, and M.H. Young. 2011. Drylands in the anthropocene: Taking stock of human-ecological interactions. *Geography Compass* 5(3): 112–127.

Wilcox, B.P., M.S. Seyfried, D.D. Breshears, and J.J. McDonnell. 2012a. Ecohydrologic connections and complexities in drylands: New perspectives for understanding transformative landscape change preface. *Ecohydrology* 5: 143–144.

Wilcox, B.P., L. Turnbull, M.H. Young, C.J. Williams, S. Ravi, M.S. Seyfried, D.R. Bowling, R.L. Scott, M.J. Germino, T.G. Caldwell, and J. Wainwright. 2012b. Invasion of shrublands by exotic grasses: Ecohydrological consequences in cold versus warm deserts. *Ecohydrology* 5: 160–173.

Wood, M.K., and W.H. Blackburn. 1981. Grazing systems: Their influence on infiltration rates in the Rolling Plains of Texas. *Journal of Range Management* 34: 331–335.

Wood, M.K., W.H. Blackburn, F.E. Smeins, and W.A. McGinty. 1978. Hydrologic impacts of grazing systems. In *First international rangeland congress*, 288–291.

Wood, E.F., M. Sivapalan, and K. Beven. 1990. Similarity and scale in catchment storm response. *Reviews of Geophysics* 28: 1–18.

Woodward, L. 1943. Infiltration capacities of some plant-soil complexes on Utah range watershedlands. *Transactions of American Geophysical Union* 24: 468–473.

Wurster, F.C., D.J. Cooper, and W.E. Sanford. 2003. Stream/aquifer interactions at Great Sand Dunes National Monument, Colorado: Influences on interdunal wetland disappearance. *Journal of Hydrology* 271: 77–100.

Yeh, P.J.Ä., S. Swenson, J. Famiglietti, and M. Rodell. 2006. Remote sensing of groundwater storage changes in Illinois using the Gravity Recovery and Climate Experiment (GRACE). *Water Resources Research* 42, W12203. doi:10.1029/2006WR005374.

Zencich, S.J., R.H. Froend, J.V. Turner, and V. Gailitis. 2002. Influence of groundwater depth on the seasonal sources of water accessed by Banksia tree species on a shallow, sandy coastal aquifer. *Oecologia* 131: 8–19.

Zeng, N., J.D. Neelin, K.-M. Lau, and C.J. Tucker. 1999. Enhancement of interdecadal climate variability in the Sahel by vegetation interaction. *Science* 286: 1537–1540.

Zhang, L., W.R. Dawes, and G.R. Walker. 2001. Response of mean annual evapotranspiration to vegetation changes at catchment scale. *Water Resources Research* 37: 701–708.

Zheng, X., and E.A. Eltahir. 1998. The role of vegetation in the dynamics of West African monsoons. *Journal of Climate* 11: 2078–2096.

Zobeck, T.M., G. Sterk, R. Funk, J.L. Rajot, J.E. Stout, and R.S. Van Pelt. 2003. Measurement and data analysis methods for field-scale wind erosion studies and model validation. *Earth Surface Processes and Landforms* 28: 1163–1188.

Zou, C.B., P.D. Royer, and D.D. Breshears. 2010. Density-dependent shading patterns by Sonoran saguaros. *Journal of Arid Environments* 74: 156–158.

Zreda, M., W.J. Shuttleworth, X. Zeng, C. Zweck, D. Desilets, T. Franz, R. Rosolem, and T.P.A. Ferre. 2012. COSMOS: The cosmic-ray soil moisture observing system. *Hydrology and Earth System Sciences Discussions* 9: 4505–4551.