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**Dryland
ecohydrology and
climate change:
critical issues**

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Dryland ecohydrology and climate change: critical issues and technical advances

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Drylands cover about 40 % of the terrestrial land surface and account for approximately 40 % of global net primary productivity. Water is fundamental to the biophysical processes that sustain ecosystem function and food production, particularly in drylands, where a tight coupling exists between water resource availability and ecosystem productivity, surface energy balance, and biogeochemical cycles. Currently, drylands support at least 2 billion people and comprise both natural and managed ecosystems. In this synthesis, we identify some current critical issues in the understanding of dryland systems and discuss how arid and semiarid environments are responding to the changes in climate and land use. Specifically, we focus on dryland agriculture and food security, dryland population growth, desertification, shrub encroachment and dryland development issues as factors of change requiring increased understanding and management. We also review recent technical advances in the quantitative assessment of human versus climate change related drivers of desertification, evapotranspiration partitioning using field deployable stable water isotope systems and the remote sensing of key ecohydrological processes. These technological advances provide new tools that assist in addressing major critical issues in dryland ecohydrology under climate change

1 Introduction

Drylands are regions with relatively low precipitation, long dry spells (e.g. dry seasons), and frequent occurrence of water scarce conditions. They are typically located in areas of prevalent divergence in the patterns of atmospheric circulation, on the leeward (“rain shadow”) of mountain chains, in arid continental regions, or in the proximity of cold ocean surfaces. The drylands definition is often based on total annual precipitation being low relative to potential evapotranspiration (ET). To this end an aridity index (AI), defined as the ratio between precipitation and potential ET, is used to classify drylands

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as regions where the AI is smaller than 0.65 (e.g. sub humid dryland, semi-arid dryland).

Drylands collectively cover about 40 % of the terrestrial land surface (Table 1) and contribute approximately 40 % of global net primary productivity (Grace et al., 2006).

Vegetation dynamics exert a strong control on the water cycle in drylands, due in part to the tight coupling that exists between the water, energy, and biogeochemical budgets in these systems (Noy-Meir, 1973; Austin et al., 2004; Wang et al., 2009a; Tietjen et al., 2010). For example, in the Mojave desert of the southwest US, elevated winter precipitation stimulated a rapid increase in vegetation productivity, which in turn reduced soil water storage by half – compared to a paired unvegetated site – and precluded deep drainage below the root zone (Scanlon et al., 2005). A converse example would be the conversion of perennial vegetation to annual crops, which is typically associated with an increase in groundwater recharge, and – in some cases – the rise of shallow water tables and salt accumulation at the ground surface, as observed in many drylands around the world, including the case of south western Australia.

Besides the strong linkage between water, energy and biogeochemical fluxes, across-scale hydrological connectivity is another important feature of arid and semi-arid landscapes. Hydrological connectivity is a system-level property that results from the linkages in the networks of water transport through ecosystems, by which feedbacks and other emergent system behavior may be generated (Miller et al., 2012). Because of the low hydraulic conductivity of dry soils, the subsurface connectivity of arid and semiarid landscapes is generally low when compared with their wet and sub-humid counterparts (Grayson et al., 1997). The connectivity provided by surface waters is often intermittent or ephemeral and limited to wet periods or seasons when surface overland flow occurs and the stream network is active. Hydrological connectivity is not well characterized in most systems and the challenge of modeling hydrological connectivity lies in the poor understanding of cross-scale interdependencies of the processes controlling water fluxes from the soil to the plant and the atmosphere (e.g. Loik et al., 2004). Representing and synthesizing hydrological connectivity, from the point to

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the landscape scale, will require enhanced knowledge of connections among hydro-
logic conditions, climate, vegetation, soil processes, and landscape morphology. Re-
cent efforts have been focusing on better characterizing hydrological connectivity. For
example, Wang et al. (2012a) developed a conceptual framework for upscaling eco-
hydrological and biogeochemical processes using electrical circuit analogies and the
Thévenin's theorem, highlighting its utility to represent concomitant processes at both
small and large spatial scales.

Water is fundamental for biological processes responsible for ecosystem function
and food production, and for abiotic processes controlling the land-atmosphere interac-
tions. Dryland ecohydrology describes the hydrologic mechanisms that underlie eco-
logical patterns and processes in water-limited ecosystems (Rodríguez-Iturbe, 2000;
D'Odorico et al., 2010b). Currently, drylands support more than 2 billion people and
comprise both natural and managed ecosystems (MEA, 2005; Gilbert, 2011). Growing
global populations are expected to increase the pressure on these ecosystems, thereby
further exacerbating the already tight limitations imposed by water availability and food
security. Thus, there is an urgent need for better management strategies to avoid the
emergence of potential conflicts resulting from poor understanding of the underlying
ecohydrological processes. With increasing anthropogenic influences on hydrological
cycles, ecohydrology research is moving towards more human-dominated landscapes
(Jackson et al., 2009). Future environmental and socio-economic changes such as ris-
ing CO₂ and temperature, changing rainfall patterns and even dietary shifts are likely to
have profound impacts on dryland ecosystem dynamics. Many dryland savannas and
mixed cropping systems have a combination of different plant physiognomies, including
both C₃ and C₄ plants (Wang et al., 2009b, 2010a). Since C₃ and C₄ plants respond to
CO₂ enrichment and temperature increase differently (Morgan et al., 2011), the com-
bination of plant physiognomy increases the complexity of managing and predicting
dryland responses to future environmental changes.

In this synthesis, we aim to use ecohydrological principles and published literature to
identify current critical issues in dryland research. Specifically, we will focus on some

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emerging issues in dryland research, including the relation between agriculture and water use, dryland population growth, shrub encroachment, desertification and dryland development. Some of these points have been widely discussed elsewhere, while others are yet to draw the attention they demand. We will also discuss some current technical advances and future challenges in the developments of new research tools, including remote sensing and stable isotope monitoring tools, which will assist in addressing these critical issues in dryland ecohydrology.

2 Critical issues in drylands

2.1 Dryland population growth and water demands

Global water resources are inherently related to and affected by population growth (Vörösmarty et al., 2000). Developing nations account for 90 % of dryland populations. Figure 1 shows that most dryland dominated countries (defined as where dryland areas are larger than 50% of the total areas, base on the definitions provided in the Introduction) exhibit a much higher population density growth compared with the global average.

Water footprint is an indicator of water consumption that includes both direct and indirect water use, and is defined as the total volume of freshwater used to produce the goods and services consumed by an individual or a community (Chapagain and Hoekstra, 2004). Most of the human appropriation of freshwater resources is used for food production (e.g. Falkenmark and Rockström, 2004). Without accounting for any change in the per capita water footprint, the ongoing demographic growth is expected to increase the pressure exerted by the humanity on the global water resources. At the same time, however, it has been reported that economic growth is allowing some populations to have access to more water intensive food commodities; the shift to more meat based diets will substantially increase the per capita water use, further increasing the water footprint of human societies (Strzepek and Boehlert, 2010). Changes in

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water consumption, induced by shifts to more water intense diets, could be dramatic. For example, in China and India the per capita water footprint is currently 1071 and 1089 m³/person/year, respectively, while in the US it is 2842 m³/person/year (Hoekstra and Mekonnen, 2012). By 2050, the populations of China and India are predicted to reach 1.4 and 1.5 billion, respectively (the United Nation's medium scenario, the population data is from the Department of Economic and Social Affairs of the United Nations). If in these countries the per-capita water footprint reaches the levels of the US, their freshwater consumption would become at least three times larger than the current rates. Since water resources are already under severe pressure in both countries, meeting this future demand will be a daunting challenge for the next generation.

Population growth in conjunction with an increase in per capita water use is affecting a number of dryland countries around the world. Food security is at risk when in these countries the available freshwater is not sufficient to produce the food needed by their populations. Severe water stress conditions are expected to cause malnourishment, famine, and social unrest. However, it has been noted that, despite water scarcity some societies are meeting their food demand through the importation of food commodities. International trade of food products has been associated with a virtual transfer of the water required for their production (Allan, 1998). Virtual water trade is, however, only a short term remedy: it does not correct the unbalance existing between the growing global population and the limited available water resources; it does not promote equality in the access to freshwater (Seekell et al., 2011); it reduces societal resilience (D'Odorico et al., 2010c), and makes some societies (e.g. China) increasingly dependent on more water resources than they do not control (Carr et al., 2012).

2.2 Dryland agriculture and climate change

The effect of climate change on crop production is of considerable concern. Predictions for the US central Great Plains indicate that the negative effects of rising temperature on crop production will offset the positive impacts of CO₂ increase (Ko et al., 2012). The effects of climate change are already being felt in the global food markets, and

are becoming particularly strong in some dryland areas, where crops fail and yields decline (Food and Agriculture Organization, FAO, 2006b). This is, for instance, the case of the semi-arid tropics, which are home to about 22 % of the world's population, and where conditions of chronic poverty and inadequate food consumption are often found (Falkenmark and Rockstroem, 2008).

The United Nations FAO predicts that by 2050 agriculture will have to support additional 2.7 billion people (FAO, 2006a). To feed the increasing global population while eradicating malnourishment, the human appropriation of freshwater resources needs to double within the next 40–50 yr (Falkenmark and Rockstrom, 2004). Our ability to meet such demand is constrained by the limited availability of accessible freshwater resources on Earth. An increase in food production can – in principle – be achieved by increasing irrigated land, expanding croplands at the expenses of natural ecosystems, and developing new technologies that enhance the water use efficiency of agricultural production (“more crop per drop”) through genetically modified crops or water saving agricultural techniques (Falkenmark and Rockstrom, 1996). Currently, 19 % of the global agricultural land is irrigated and produces 40 % of the world's food supply (Hanjra and Qureshi, 2010). Meeting the projected food demand solely by expanding irrigated areas is unfeasible (Falkenmark and Rockstrom, 2004). Irrigation requires water withdrawal from lakes, streams or the groundwater. Globally, about 2600 km³ of water are already withdrawn every year for irrigation (Strzepek and Boehlert, 2010). Agriculture accounts for more than 66 % of the total human withdrawals. Most of the rivers flowing through dryland regions are already intensively exploited. Many of them (e.g. the Rio Grande or the Colorado River) barely make it to the ocean. An expansion of irrigation can contribute only in part to the projected increase in food demand.

There are only limited opportunities to increase agricultural production by sustainably increasing the arable land. Thus, the global agricultural area is not expected to substantially increase (Fedoroff et al., 2010). However, drylands are often considered as possible candidates for cropland expansion (both for food and biofuel production). For example, in 2009, the World Bank identified 600 million hectares of African savannas

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and woodlands as the primary expansion opportunity to increase food production in Africa (World Bank, 2009). However, the conversion to agriculture of these drylands would not occur at no cost if we account for the environmental services they currently provide, including rangeland, firewood production, carbon sequestration, maintenance of biodiversity, and provision of habitat for wildlife. Moreover, the conversion to cropland has the potential to reduce runoff dramatically with important impact on stream and fluvial habitats.

Regardless of a possible increase in arable area, dryland agriculture is expected to be particularly vulnerable to the effect of climate and land use change. In the semiarid tropics smallholder farmers rely on extremely variable and uncertain rainfall regimes. The vulnerability is also contributed by soil salinization and other forms of human- or climate-induced land and water degradation. Future climate projections are uncertain but indicate a possible increase in climate variability, a decrease in mean precipitation (Sheffield and Wood, 2007), the occurrence of more frequent droughts, and increased temperature extremes (Schlenker and Lobell, 2010). All of these factors will further exacerbate the vulnerability of agricultural production in the dry tropics. For instance, in the near future (20–30 yr) climate change is predicted to threaten food security in southern Africa (Lobell et al., 2008) and the Sahel (Patricola and Cook, 2010), while in other areas (e.g. central Africa), the uncertainty of these estimates is too large to make informed decisions. In addition to changes in mean climate conditions, changes in climate variability pose further challenges on farmers who may be able to adapt to long-term changes but not to abrupt interannual variability.

Furthermore, the response to climate change in drylands not only depends on climate itself, but also on the social and economic aspect of the society. For example, in pastoral society, changes in rainfall amount and arrival time will change vegetation productivity. However, to translate productivity consequences into livelihood and development impacts, it requires additional levels of cross-disciplinary synthesis to incorporate geographical, social, economic, and technological dimensions of the linked human-ecohydrology system. In agricultural system, we can also see the

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amplification of meteorological impact by anthropogenic activities. For example, recent work (e.g. Mwale, 2003) has begun to show that in many cases agricultural drought can be quite substantial (i.e. complete crop failure) even when meteorological drought (i.e. rainfall deficit) is mild. Therefore, the frequency and severity of a “drought year” depends heavily on both social and agricultural factors, which are themselves strongly coupled to spatial expressions of hydrological dynamics, land cover patterns, and local coping behaviors. Although much progress has been made in understanding changes in food security under the threat of climate change (Hanjra and Qureshi, 2010), it is still unclear how an intensification of climate fluctuations might affect food production and what policies could mediate those impacts.

2.3 Dryland development challenges

By taking a broader view of drylands not just as ecosystems, but as coupled human-environment systems, we can see the pivotal importance of ecohydrology in the pursuit of human development. In developing countries, the livelihoods and well-being of rural dryland populations tend to be tightly and directly linked to ecological processes, as societies in these regions typically engage in household-scale, low-technology livestock and/or crop production (Reynolds et al., 2007). Pastoral systems use natural dryland ecosystems extensively for livestock production, with mobility, flexibility, and common pool resource management institutions to track and access shifting resource availability (Robinson et al., 2011). Dryland smallholder agriculture is typically rain-fed, or may utilize localized sources or on-site catchment for limited irrigation, making crop yields highly dependent on seasonal rainfall and farming practices that affect the partitioning of precipitation in the water balance equation (Falkenmark and Rockstrom, 2008; Notenbaert et al., 2009).

Today, both systems, and hybrids thereof, are struggling to support human livelihoods and maintain resilient ecological processes in the face of growing populations, land use pressures, and climate change. African drylands, which cover 40% of the continent’s land area, epitomize these challenges for poverty reduction, economic

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development and environmental sustainability. Poverty itself limits choices for coping strategies, such as technological investments or adoption of industrial or other livelihoods (Thornton et al., 2006). A human-ecohydrology lens can be applied to help understand how ecohydrologic conditions govern rural productivity, and can point to appropriate, creative approaches to forge more beneficial feedbacks between landscapes and livelihoods.

In pastoral systems, land degradation is part of a “diminishing resource syndrome”, in which increased livestock densities and limited mobility feed back to degrade the capacity of landscapes to capture and convert incoming rainfall into primary production, while competing land uses and conversion of higher productivity rangelands to agriculture reduce the area available for livestock production. Rangeland rehabilitation research has shown that in many cases physical manipulations such as microcatchments can reinstate more productive water-soil-plant relations. But additional, more holistic assessments are needed to determine: (a) the extent to which livelihood benefits from land rehabilitation can mitigate other stressors due to demographic and land use pressures, and (b) the optimal location of rehabilitation efforts in heterogeneous landscapes to generate the greatest impacts on productivity and livelihoods.

Many smallholder dryland cropping systems in Africa have also undergone suites of social and ecohydrological changes. Demographic pressures have led to increased farming intensity, reduced fallow periods, and driven cropping into increasingly marginal areas. Crop genetics, fertilizers, and pest management have spurred the Green Revolution in agricultural intensification in Asia and Latin America, but not in rural Africa. Lack of water availability and irrigation infrastructure are key barriers to such development (Rockstrom et al., 2007). While major aid organizations are currently redoubling their efforts to create more efficient, drought resistant crops for Africa, knowledge of dryland ecohydrology has inspired other approaches: Conservation Agriculture and the Green Water approach (FAO, 2006a; Falkenmark and Rockstrom, 2008). With the mantra, “more crop per drop”, these approaches seek to maximize the fraction of precipitation (or irrigation) that is routed through productive plant growth (transpiration), by

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reducing losses to runoff, evaporation, and deep drainage. With technologically simple tactics like microcatchments, mulching, and strategic timing of watering, crop yields per water input can increase several-fold (FAO, 2006a; Falkenmark and Rockstrom, 2008). At present, we lack coupled evaluations of plant-level (crop genetics) and farm-level (conservation agriculture) approaches, which assess the climatic, ecological, and social conditions under which the approaches offer higher or more sustainable productivity gains.

2.4 Desertification and human vs. climate induced desertification

Many drylands around the world are affected by rapid change in vegetation cover and composition, hydrologic conditions, and soil properties, which result in an overall loss of ecosystem services and poses serious threats to sustainable livelihoods. The process underlying these changes is often termed “desertification” (D’Odorico et al., 2012). The United Nations Convention to Combat Desertification (UNCCD, 1994) defines desertification as land degradation in arid, semi-arid or sub-humid areas resulting from various factors that include climate variations and human activities. About 10 % to 20 % of global drylands suffer from desertification and are prone to a decline in land productivity (Reynolds et al., 2007; D’Odorico et al., 2012). A number of processes can contribute to this decline, including soil erosion (Li et al., 2007; Ravi et al., 2009), salinization, loss of soil fertility or depletion of seed banks. These factors have important impacts on vegetation density and species composition. Desertification is commonly associated with changes that persist for several decades and are presumably irreversible, at least within the time scales of a few human generations.

As recognized by the UNCCD (1994), dryland desertification may be broadly associated with two underlying drivers, namely changes in climate or human activities. These drivers may cause an ecosystem shift to a “desertified” state, while positive feedbacks stabilize the system in this new state (D’Odorico et al., 2012). To most effectively combat degradation, we need to quantitatively assess the extent to which a region’s degradation is caused by climate variations or human activities. Recent methods utilizing

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remote sensing and modeling techniques to distinguish between human versus climate induced desertification are presented in details in the “technical advances” section.

2.5 Ecohydrological consequences of shrub encroachment

Woody thickening or encroachment, is the increase in the density and cover of woody plants into open grasslands and woodlands, and is a global phenomenon (Archer et al., 1995). Encroachment is associated with a number of ecosystem changes ranging from a change in the spatial distribution of soil nutrients, altered habitat value for wildlife, and changes in the ability of the soil to redistribute water vertically and horizontally (Schlesinger et al., 1990, 1996; Archer et al., 2001; Bhark and Small, 2003; Zarovalli et al., 2007). Changes in shrubland communities that alter the balance between precipitation, run-off, interception and infiltration are likely to have marked effects on the structure and function of shrubland ecosystems. The likely long-term effect is to reinforce the persistence of shrublands at the expense of grasslands (Reynolds et al., 2007).

Although water is a substantial driver of ecosystem processes in semi-arid shrublands, relatively little is known about run-off and infiltration processes and the hydrological responses to encroachment. The dichotomy between shrub canopy and interspace is a major determinant of ecosystem productivity and diversity. The heterogeneous nature of the vegetation in drylands is thought to be controlled by processes of upslope water erosion and sedimentation, and complex interactions among individual plants and the surrounding soil matrix (Puigdeféregas and Sanchez, 1996; Bochet et al., 1999; Reid et al., 1999; Wang et al., 2007). Both the movement and storage of water within shrublands is highly variable (e.g. Breshears et al., 2009). These issues make it extremely difficult to model or predict likely hydrological responses to changes in management or climate.

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2.5.1 The effects at plot scale

A number of knowledge gaps relate to the effect of woody encroachment on soil hydrologic conditions in semi-arid systems, which hinder the prediction of climate change effects on soil-vegetation interactions. Below we identify the main knowledge gaps that relate to shrub encroachment, with an emphasis on eastern Australia and the western US:

1. Enhanced levels of infiltration surrounding the canopy are a defining feature of arid zone shrub communities (e.g. Bhark and Small, 2003). Soil porosity is greater and macropores are present adjacent to the roots and stem of woody plants, which also have well-developed tap roots allowing the plants to access water from greater depths (Archer et al., 2002). While the extent of infiltration is known to decline with increasing distance from the canopy, the exact nature of this decline is largely unknown for most woody species, and has been studied in only a few arid zone shrubs (e.g. *Atriplex* spp., Dunkerley, 2000).
2. The relative interception value of woody plants is poorly known. For example, we are aware of only a few studies of interception of shrubs (e.g. Wood and Wood, 1986) and there are few data on interception and stemflow for woody plants in arid and semi-arid eastern Australia. Limited data suggest that stemflow and interception for box eucalypt communities are low (<3% of total rainfall) (Johns, 1981; Tunstall and Connor, 1981), suggesting that the majority of precipitation passes directly through the canopies. Thus while this suggests that surface hydrologic conditions are more influenced by soil characteristics than plant architecture, there are limited studies to confirm this.
3. The degree to which woody and encroached communities enhance water erosion (either through physical effects or by reducing infiltration) is poorly understood (Eldridge et al., 2003). Hydrological models have been used to study how the amount of rainfall reaching the ground, and hence the risk of erosion, varies according to

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changing cover of woody plant canopies (Wu et al., 2001). As woody vegetation tends to intercept more rainfall than groundstory vegetation (Thurow, 1991; Wu et al., 2001), it could be argued that woody vegetation reduces the risk of erosion to a greater degree. However, the situation is far more complex. As raindrops falling from taller (>2 m tall) canopies tend to be more erosive (i.e. with higher kinetic energy) than those falling from shorter heights (Moss and Green, 1978), canopy height is likely to be an important driver of erosion risk. There is an urgent need to test empirically some of the relationships between canopy cover and height for different vegetation communities.

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4. The relationship between woody plants and understory herbaceous cover is complex, and likely mediated by grazing intensity (Barger et al., 2011). While overgrazing is likely to lead to reductions in ground cover, this may be compensated for by increases in the cover of other components such as litter. The interception capacity of grasses is biomass- and cover-dependent (Crouse et al., 1966), ranging from 0.3 to 2.5 mm of water (Branson et al., 1972). However, litter also has the capacity to intercept rainfall, depending on the type and depth (Branson et al., 1972). Litter cover, origin and degree of incorporation are known to be correlated with the capacity of the soil to resist erosion and infiltrate water (Tongway, 1995). However, the relationships among litter depth, type and interception are not known for semi-arid Australian landscapes.
5. Runoff coefficients are thought to be much less in woody communities than in the herbaceous communities that they replace. For example, run-off coefficients reported by Harrington et al. (1981) for the semiarid Australian woodlands were seven-times lower for thickets of trees and shrubs than for the inter-thickets, similar to results from the piñon-juniper woodlands in the western US (Reid et al., 1999). Data on the differences in runoff coefficients across soil types and vegetation communities are needed in order to parameterise runoff and erosion models for wooded communities.



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6. There is a pressing need to separate out the direct effects of woody plants on sub-surface flow, through enhancement of macro-porosity, from the indirect effect of woody plants via their mediation of soil surface condition. Increased shading under woody plants is known to alter the richness and cover of understory plants (Smit et al., 2007). Sub-canopies microsites are also highly preferred by biocrusts; complex communities of mosses, lichens and cyanobacteria (Eldridge et al., 2010). Biocrusts are known to have substantial effects on hydrology in the near-surface layers (Eldridge et al., 2010), but the extent to which this is moderated by shrubs, or by the herbivores that tend to graze under the canopy, is largely unknown. New models of water flow through soils, using different supply potentials, are currently being evaluated using systems-based approaches (e.g. Eldridge et al., 2010).

2.5.2 The effects on regional hydrological processes

Shrub encroachment may also have dramatic effects on regional hydrological processes. Encroachment can lead to land-to-atmosphere feedbacks with possible impacts on rainfall and temperature regimes. Small and Kurc (2003) found only limited potential feedbacks to precipitation in Creosote bush (*Larrea tridentata*) shrublands in North America. However, it has been argued that reduced woody cover may reduce rainfall by altering surface roughness, ET and cloud formation (McAlpine et al., 2009). Much of this is largely unknown, however, and the exact magnitude of any regional hydrological changes resulting from encroachment can only be speculated upon. Shrub encroachment can affect the land surface albedo, emissivity, and roughness with important impacts on the near surface climate (Beltran-Przekurat et al., 2008). Even though in some cases changes in albedo are negligible, the increase in soil energy storage encroached area can modify the microclimate with a positive feedback on vegetation (D'Odorico et al., 2010a; He et al., 2010). Recent regional climate modelling activities that seek to change the boundary conditions of the surface state, may

provide some insight into the influence and strength of land-atmosphere couplings as a response to changing surface condition.

These effects are likely to change markedly with increases in global temperatures, increases in the severity of high intensity rainfall events, and greater spatial variability in groundcover and therefore the capacity of the soil to resist erosion. Predictions of more frequent drought for the south-western US are likely to result in an increased frequency and severity of wildfires, increasing the proliferation of shrubs by removing competition from grasses and reducing biological soil crusts (Eldridge et al., 2011). However, fire frequency or intensity above a critical threshold will have the reverse effect of reducing woody shrub density (Hodgkinson et al., 1984). For example, in the dryland catchments around major centres of population in southern Australia, more frequent wildfire is likely to stimulate the germination of woody plants by removing the dominant fire-tolerant Eucalypt overstory. These dense thickets of regenerating Eucalyptus will support relatively high rates of infiltration thereby reducing run-off capture to water storages. The long-term result of this is a decrease in water storage for major capital cities. We lack the long-term data sets to be able to assess the likely impacts of regrowth, a point echoed by Asbjornsen et al. (2011).

A similar scenario, though with different results is apparent in arid south-western US. Here the replacement of grassland and shrubland exposes more of the surface to the action of raindrop impact resulting in accelerated erosion and potential sedimentation of drainage (Abrahams et al., 1994). These regional studies reinforce the notion that more specific catchment-specific data are needed for both the ecological (e.g. tree rooting depth, canopy architecture and structure, depth of water intake) and hydrological (soil texture and hydraulic conductivity, soil moisture availability, hydrological connectivity) components of these systems, in order to improve our catchment wide modeling of the likely effects of vegetation change on catchment hydrology.

The positive effects of woody plants on soils and groundstory vegetation needs to be balanced with their negative effects of competing directly for water and intercepting rainfall. Whilst the relationship between woody plants, groundstory plants and soil water

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the climate variability is removed, and any remaining trend in these residuals is then ascribed to human activities (Fig. 2). This method is hereafter referred to as RESTREND. Using this methodology Evans and Geerken (2004) identified the regions undergoing significant human caused degradation in the Syrian drylands. With a focus on the hot drylands, changes in temperature were not found to be important, however it does have the potential to be important in cold drylands.

The RESTREND method has subsequently been used in many studies to identify the contribution of climate variability and human activities to land degradation (Herrmann et al., 2005; Wessels et al., 2007; Propastin and Kappas, 2008; Paudel and Andersen, 2010; Brinkmann et al., 2011). Some studies have further shown the connection between the identified degraded areas and a particular human activity such as over-grazing (Geerken and Ilaiwi, 2004; Paudel and Andersen, 2010).

The RESTREND method has proven to be robust and effective in many studies. The RESTREND method is however, relatively simple and it contains several limitations. It assumes there is a linear relationship between vegetation and precipitation; that the identified relationship is not overly influenced by the presence of degradation; and that the vegetation measure (e.g. NDVI) can represent all forms of degradation of interest. Each of these limitations has been addressed in various studies. The assumption of a linear relationship between vegetation (NDVI) and precipitation appears robust within dryland systems, however the assumption breaks down in wetter environments. Wessels et al. (2007) applied a log transform to the precipitation data to account for the lower responsiveness of vegetation to precipitation in high precipitation areas. With this change they found that the RESTREND method was successful in identifying the degrading regions across a large climatic gradient in South Africa. Others found similar non-linear relationships between vegetation and precipitation when applied outside the drylands (e.g. Paudel and Andersen, 2010). For practical reasons, a number of studies use a single precipitation accumulation period when calculating the vegetation-precipitation relationship. This necessarily obscures differences in this relationship between vegetation types, and generally reduces the significance of any

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relationship found. Evans and Geerken (2004) addressed this by independently calculating the relationship for each pixel, reducing, though not eliminating, the impact of mixed vegetation types within the relationship. Omuto (2010, 2011) addressed this issue by combining the RESTREND method with mixed-effects modeling of the NDVI-rainfall relationship. Rather than calculating an independent NDVI-rainfall relationship for each pixel, as in Evans and Geerken (2004), the mixed-effects model calculates the average relationship for an entire area and variations around this for designated sub-regions, in this case vegetation types.

The RESTREND method assumes that the relationship seen in the observations between the vegetation (NDVI) and precipitation contains the climate influence, while the residuals of this relationship contain all other influences on the vegetation. However, if an area has been degraded during the observation record this will be evident as a smaller vegetation response to the same precipitation input, and the identified vegetation-precipitation relationship will embody some of this response. That is, the RESTREND method necessarily underestimates the degradation (or improvement) that has occurred. This limitation has been addressed by estimating the “potential” vegetation given the climate conditions. This potential vegetation has been estimated in a number of ways. Wessels et al. (2008) used the 90th percentile of Net Primary Production (NPP) for each biophysical land unit to define the potential production of the land. Xu et al. (2010) used NPP calculated using the Carnegie-Ames-Stanford Approach (CASA) model to determine the potential of an area, while Zhang et al. (2011) used this approach with estimates from the Thornthwaite Memorial and Synthetic models as well. Hickler et al. (2005) and Seaquist et al. (2009) used a dynamic vegetation model to calculate the potential vegetation response to climate. In each case deviations below the potential indicate a degraded landscape.

In drylands degradation may occur without any discernible change in the vegetation measure, usually NDVI. Probably the most important of these changes is caused by over-grazing. The livestock will preferentially graze on the most palatable species which can result in the replacement of these species by less palatable species. Such a

change in vegetation composition may not produce any change in the observed NDVI but may dramatically reduce the livestock carrying capacity of the land. It has been proposed that by utilizing the phenological cycle as embodied in time series of remotely sensed vegetation indices, such as NDVI, it is possible to differentiate vegetation species. Most conventional studies only use particular phenologically relevant variables such as green period, peaking time, or onset/end of green period (e.g. Boyd et al., 2011). Others have used decision (regression) tree techniques based on a collection of the above variables (Hansen et al., 2000; Bradley and Mustard, 2008). These approaches do not make full use of the information available over the entire phenological cycle. One technique used to extract information from the full time series is to treat the multi-temporal data set as if it is a multi-spectral data set and apply spectral-unmixing algorithms to it (Singh and Glenn, 2009), another is to decompose the shape of the phenological cycle using Fourier Transforms and base the classification on the Fourier components which is the approach used in Geerken et al. (2005) and Evans and Geerken (2006).

Geerken et al. (2005) presents the Fourier Filtered Cycle Similarity algorithm in which the user identifies optimum weightings for Fourier harmonics calculated from a reference vegetation cycle. These optimum parameters are applied to the entire area of interest and pixels of similar shape to the reference cycle are deemed to be the same vegetation type. The measurement of similarity is based on a linear regression between the reference cycle and the target cycle. To overcome the high computational burden and subjective nature of this algorithm Evans and Geerken (2006) introduced the Fourier Component based shape Similarity Measure. This similarity measure is based directly on the Fourier Components and is designed mathematically such that it is invariant to modifications unrelated to the plants phenology, that is, the similarity measure is invariant to vertical displacements caused by different backgrounds (soils), temporal shifts caused by changes in the onset of the wet season or across strong climate gradients, and magnitude variations which can be caused by changes in the coverage or vigour of the vegetation. These techniques have been further developed

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by Geerken (2009). They have been found to provide accurate vegetation type classifications down to the level of differentiating shrub types. Geerken (2009) also introduced a change detection algorithm that differentiates between coverage changes and vegetation type changes making these shape similarity based approaches very promising as a means to detect early stages of degradation in drylands such as the replacement of palatable shrubs by unpalatable ones.

To date, attempts to differentiate between human-induced and climate caused degradation have focused on precipitation as the representing the required climate variations. However in cold dryland regions where temperatures reach below zero Celcius, temperature changes may also play an important role. Liu et al. (2012) apply the RE-STREND method to the Mongolian Steppe and found that inter-annual changes in temperature had a significant impact on the vegetation and so must be included along with precipitation in order to account for the full climate effect. As global warming continues it is likely that the long-term change in temperature will play a larger role in many environments and should not be neglected. Liu et al. (2012) found that once the climatic effects had been removed, regions of degradation in the Mongolian Steppe could largely be attributed to over-grazing due to the increase in Cashmere goat population. The challenge now is to determine what actions can be taken to stop, or even reverse, the degradation without damaging one of Mongolia's largest export markets and before the degradation becomes irreversible.

3.2 Remote sensing of water budget components in drylands

As already noted, the variability and distribution of water availability in the landscape is of paramount importance for drylands. There have been a number of exciting developments in monitoring tools useful for ecohydrological research over the last decade. For example, field deployable laser based spectroscopy approaches that determine the ratios of hydrogen and oxygen isotopes (Lee et al., 2005; Wang et al., 2009c, 2012b), cosmic-ray (Zreda et al., 2008) and magnetic field based plot-scale soil moisture monitoring, and remote sensing based estimates of key hydrological variables such as soil

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moisture, ET and water level (Alsdorf et al., 2000) are revolutionizing the scales and precision of information sources to inform ecohydrological measurement and investigation.

Remote sensing provides the capacity to bridge the point scale focus of many ecohydrological investigations to the larger spatial extents required for whole system assessment. One of the key advantages of remote sensing platforms is the availability of data to allow extrapolation not just in space, but also across the temporal domain, offering insight into pattern change and development through time. Recent advances in hydrological remote sensing (Lettenmaier and Famiglietti, 2006) have seen research efforts that seek to address the outstanding problem of observation based hydrological closure (Sheffield et al., 2009). Such efforts to quantify the fluxes and stores of water within the terrestrial system have relevance to better understanding the hydrological implications of climate change and also the coupling and feedback mechanisms that directly impact upon ecohydrology studies.

Of considerable interest in large scale ecohydrological applications are hydrological related variables associated with the estimation of soil moisture, precipitation, vegetation and water stress, and the linked process of ET.

3.2.1 Soil moisture

Soil moisture sensing from space has been employed for a variety of applications in the hydrological sciences, most regularly in updating the state parameter for land surface modelling applications (Pauwels et al., 2001; Pan et al., 2008). From an ecohydrological perspective, characterizing the antecedent condition of a system along with the mean, range and variability of soil moisture dynamics within it, are of primary interest. Both active and passive sensor microwave based systems are available for remote estimation of soil moisture, with each representing a compromise between spatial and temporal resolution. Microwave based soil moisture retrieval is, however, limited by the depth of measurement (on the order of just a few centimetres). As such, sensors only have the capability to inform upon the near surface soil moisture: although techniques

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to extrapolate through the soil column are used (Hoeben and Troch, 2000). Although routine daily global scale estimation of soil moisture is possible, making them ideal for large scale studies, a clear limitation of passive microwave sensing is the coarse resolution of retrievals (McCabe et al., 2005). While active radar systems provide a higher spatial resolution (up to a few kilometres), the repeat time is generally on the order of a few days. Passive systems on the other hand can provide sub-daily retrievals, but with a coarser spatial coverage (approximately 25 km). Efforts to merge data sets from multiple systems and sensors have the potential to offer improved insight for large scale ecohydrological investigations (Liu et al., 2011b).

From an in-situ monitoring perspective, perhaps the most important recent advance in soil moisture estimation is the development of the COSMOS monitoring system (Zreda et al., 2008). Based on the release of both fast and slow neutrons from interactions between water in the soil column and a regular flux of cosmic rays from space, the COSMOS system provides for the first time, a typical model resolution (model GRID resolution) estimate of the soil wetness in a system. The COSMOS installations represent a revolution in terms of bridging the spatial divide that often exists between remote sensing and in-situ measurement approaches. Efforts to develop a distributed network of these systems globally will see an improved capacity to monitor ecosystem change and development in ways not previously available.

3.2.2 Precipitation

Spatial and temporal maps of rainfall distribution provide key constraints on the health and development of ecohydrological systems. While there is a wide range of satellite based retrievals that can provide enhanced characterisation of ecosystem condition or state, a number of these space based platforms provide multi-decadal sequences of important terrestrial variables. Remote measurement of precipitation has an extensive history, with numerous hydrological investigations being informed by the two decades long Tropical Rainfall and Measurement Mission (TRMM) satellite system (Kummerow et al., 2000) and other related sensors.

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Using both microwave based passive and active radar systems, together with infrared based sensors on board geostationary platforms, researchers have been able to provide improved spatial and temporal detail on precipitation structure and pattern. Over the coming years, the next generation of satellite rainfall systems, referred to as the Global Precipitation Mission (GPM), will provide a much needed update to the space based rainfall monitoring capacity. With GPM, the spatial and temporal resolution of rainfall retrievals will eclipse previous incarnations, and offer the needed level of detail to enable a range of hydrological and ecohydrological investigations.

3.2.3 Evapotranspiration

Together with precipitation, ET represents the major water flux exchanges occurring within the Earth system. Encompassing water loss from open water, bare soil and canopy components (E) and plant water release through the process of transpiration (T), the two terms can routinely exceed 90 % of the water lost from dryland ecosystems. For this reason, considerable effort has been directed at developing approaches for its accurate estimation at a range of spatial and temporal scales (McCabe and Wood, 2006). Indeed, for dryland systems, a compromise between the spatial resolution of measurements and the temporal frequency is often required, given the rapid rate at which water is exchanged through the system.

Numerous techniques for flux estimation exist, with the recent review by Kalma et al. (2008) providing a good overview of these different approaches. While many satellite based algorithms rely on the use of surface temperature and air temperature gradients to gauge heat flux potential, the surface temperature itself is a useful proxy for surface condition and stress: particularly if the diurnal temperature range can be retrieved (Sobrino and El Kharraz, 1999; Stisen et al., 2008).

Recent efforts towards better understanding the global distribution of fluxes and variability in flux retrieval approaches has been undertaken under the auspices of the Global Energy and Water Cycle Experiment (GEWEX) Landflux project (Jiménez et al., 2011; Mueller et al., 2011). Landflux is an effort to provide the community with a

climatological record of land surface heat fluxes that can be used not only for global climate model evaluation, but also more process and system scale modeling and interpretation activities. Most importantly, it will seek to be consistent with a suite of other climate system product data sets, reducing the risk of introduced bias as a response to independent forcing data sets.

These data have considerable utility in describing the long term variability and range of ecosystem behavior around the globe, allowing intercomparison of climate regions (arid, semi-arid, humid) with reduced bias as a result of data consistency. Also, other GEWEX data streams such as radiation, precipitation and ultimately soil moisture will allow further ecosystem analysis to be undertaken in a consistent framework, reducing one of the large uncertainties in mass and energy flux assessments (McCabe et al., 2008), the inherent variability in forcing data and subsequent response on model simulations.

Figure 3 illustrates a recent example of a multi-satellite Global Land surface Evapotranspiration (GLEAM) product, developed by the Vrije University of Amsterdam (Miralles et al., 2011). The approach, along with a number of other global ET data sets (see Jiménez et al., 2011 for further details), allow for the calculation of simple ecosystem and catchment indices such as the aridity index (P/ET) or evaporative fraction, providing a baseline characterization of ecohydrological response across spatial and temporal domains (see Fig. 4 for an example of such an index). Such data sets provide a much needed source of information on global patterns of evaporative response, and will prove useful in establishing change and trend detection in the hydrological cycle.

3.2.4 Vegetation and water stress

In addition to standard maps of land use and land cover which provide insight into the (often slowly) changing nature of the terrestrial surface, there are a wide range of vegetation based indices that present detail on the state and condition of vegetation across the globe. The range of available data can also include information on plant phenology, density and distribution (Badeck et al., 2004; White et al., 2009; Simard

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maximize the productive water loss (T) and minimize the unproductive water loss (E) (Wang and D'Odorico, 2008). Separating E and T however, has always been a difficult task, especially at larger scales.

Stable isotopes of water provide a useful tool to separate E and T , as E and T carry distinct isotopic signatures. The water stable isotopic compositions are traditionally measured by the isotope ratio mass spectrometry (IRMS), while the vapor phase measurements are usually based on cryogenic water vapor collection coupled with the IRMS method. Such methods are labor intensive and time consuming, as ^{18}O measurements require offline $\text{CO}_2\text{-H}_2\text{O}$ equilibrium. The typical vapor equilibrium takes about 24 h. Over the past decade, a revolutionary change in isotope measurements has seen the appearance of spectroscopy based isotope instruments capable of making continuous measurements of water vapor isotopic compositions. This new type of instruments does not usually require pretreatment and have precisions similar to traditional cryogenic based mass spectrometry methods (Lee et al., 2005; Wen et al., 2008; Wang et al., 2009c; Griffis et al., 2010). Recent research has indicated that plant derived volatile compound induced spectral contamination in leaf and stem water measurements could affect the accuracy of water stable isotopic compositions significantly (West et al., 2010; Zhao et al., 2011), limiting the application of spectroscopy-based method of plant sample measurements. There are studies which developed post correction method for modest spectral contamination of plant tissues (Schultz et al., 2011), but this remains a topic requiring future developments. Nevertheless, the continuous measurement of water vapor isotope compositions allows for the direct quantification of the water vapor and will expand the capabilities of using stable isotopes for echohydrological research.

To assess ET partitioning using stable isotopes, three isotopic end members need to be quantified: the isotopic composition of ET (δ_{ET}), T (δ_T) and E (δ_E). Recent efforts have focused on developing, refining and assessing estimation methods of all three end members. δ_{ET} is typically measured using the Keeling plot approach coupled with traditional cryogenic methods (e.g. Yopez et al., 2005). With the development of new

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laser technique, Wang et al. (2012c) extended this technique to direct chamber measurements coupling with laser instruments. Good et al. (2012) quantified and compared the uncertainties using different δ_{ET} estimation methods and showed that the eddy covariance method has the largest uncertainties, while the Keeling plot and flux gradient methods have smaller and similar uncertainties.

δ_E is typically calculated by the Craig-Gordon model (Craig and Gordon, 1965), which describes δ_E as a function of humidity, kinetic and equilibrium isotope fractionation, the isotopic composition of water of the evaporation surface, and atmospheric vapor. Soderberg et al. (2012) showed that the traditional Craig-Gordon model could be improved by adding water potential terms for dry soils.

Typically, δ_T is assessed indirectly using stem water measurements or leaf water measurements with leaf enrichment corrections. The use of stem water measurements is based on the assumption that leaves operate under isotopic steady state, so that δ_T is equal to the isotopic composition of plant source water. The assumption that leaves are in isotopic steady state is generally valid during midday because the magnitude of T relative to the volume of leaf water is large and there is a rapid turnover of water in transpiring leaves. However, non-steady state isotopic enrichment is also common in many natural systems, especially during the early morning and late afternoon (e.g. Lai et al., 2005), when T fluxes are lower. Taking advantage of the new laser technique, Wang et al. (2010a) reported the first direct measurements of δ_T using a customized leaf chamber and off-axis integrated cavity output spectroscopy water vapor isotope analyzer with pure nitrogen as purging gas. This method, however, has two limitations: (1) the limited availability of ultra-purity nitrogen gas makes this method unsuitable for many field applications. (2) The water-free and CO_2 -free inline environment affects stomata openings since humidity and CO_2 have opposite effect on stomata openings, which may alter instantaneous δ_T values. Building on this configuration, Wang et al. (2012b) developed a new framework to remove the need for dry air by employing a mass balance approach for both isotopes and water vapor inside the leaf chamber. This direct

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and continuous δ_T quantification method has been shown to effectively capture the fast δ_T responses to radiation variations (Wang et al., 2012b).

Figure 5 presents an example of using laser-based isotope monitoring technique to partition ET inside Biosphere 2 and assess the T/ET ratio changes with increasing vegetation cover (Wang et al., 2010b). With these new developments on the estimation of isotope end members, stable water isotopes provide a promising tool for partitioning ET across a range of spatial scales, which will significantly enhance water budget estimation in drylands.

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Table 1. The classification, percentage of global land area, percentage of global population of each dryland type. Data are from United Nation’s Millennium Ecosystem Assessment (2005) and Gibert (2011).

Dryland classification	Aridity Index (AI)	Global land area (%)	Global population (%)
Dry subhumid	$0.50 < AI < 0.65$	9.9 %	15.3 %
Semi-arid	$0.20 < AI < 0.50$	17.7 %	14.4 %
Arid	$0.05 < AI < 0.20$	12.1 %	4.1 %
Hyper-arid	$AI < 0.05$	7.5 %	1.7 %

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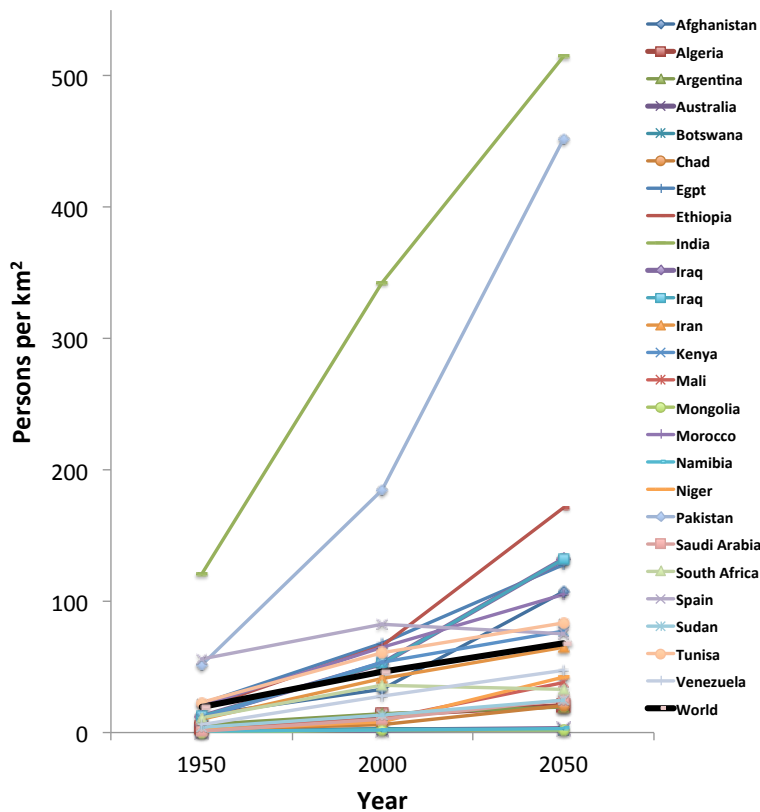


Fig. 1. Average global population density growth and population density growth for dryland dominated countries (defined as where dryland areas are larger than 50% of the total areas). The population data is from the Department of Economic and Social Affairs of the United Nations (2004), <http://www.un.org/esa/population/publications/longrange2/WorldPop2300final.pdf>.

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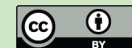
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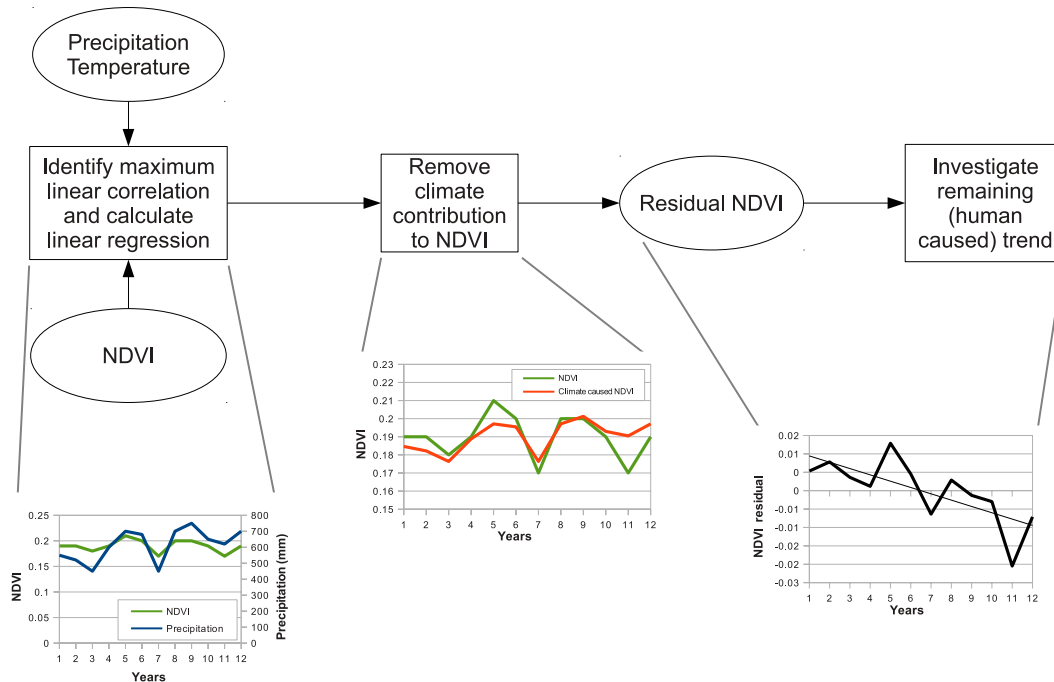


Fig. 2. Flowchart of RESTREND method to differentiate human vs. climate induced desertification. First, climate (precipitation, temperature) and vegetation (NDVI) time series are used to find the climate averaging window that produces the highest correlation. Using this window, the linear regression is calculated. Then, this regression equation is used to calculate the climate caused NDVI component. The difference between the observed NDVI and the climate caused NDVI is the NDVI residual. The trend in this residual indicates the human caused changes in the vegetation.

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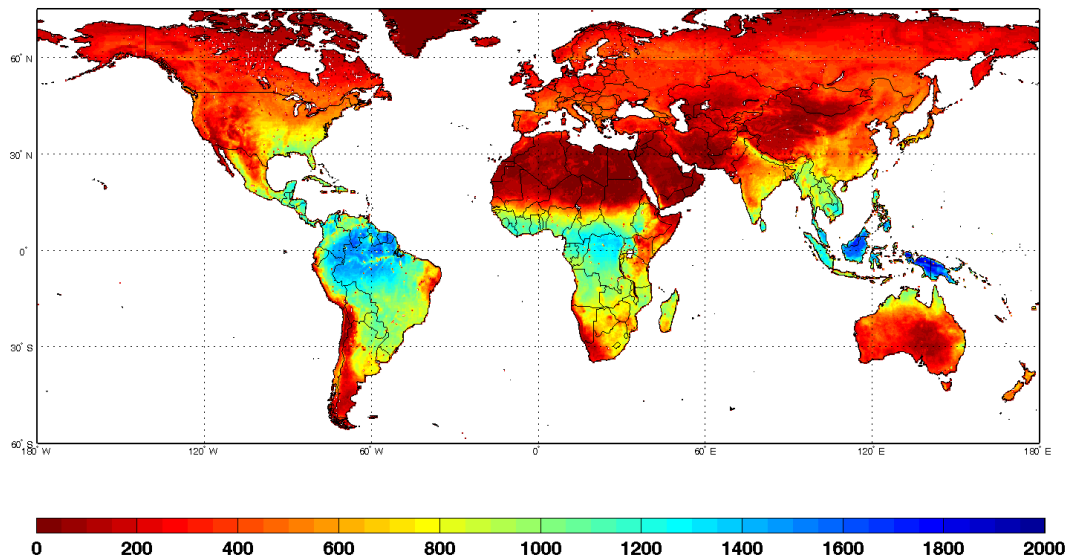


Fig. 3. Map of global evapotranspiration distribution for the year 2006. Data are from Global Landsurface Evaporation: Amsterdam Method (GLEAM).

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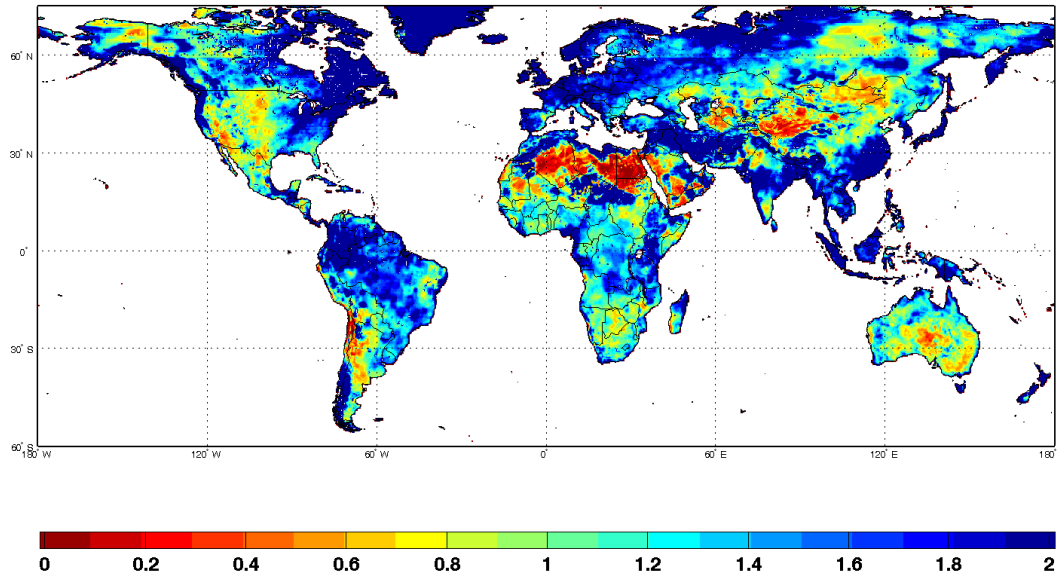


Fig. 4. Map of aridity index for the year 2006, using Global Precipitation Climatology Centre (GPCC) descriptions of rainfall at $0.5 \times 0.5^\circ$ and the Global Landsurface Evaporation: Amsterdam Method (GLEAM) for evapotranspiration. Note that evapotranspiration rather than potential evapotranspiration values were used in calculating the aridity index for this map.

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$$\frac{T}{ET} = \frac{\delta_{ET} - \delta_E}{\delta_T - \delta_E}$$

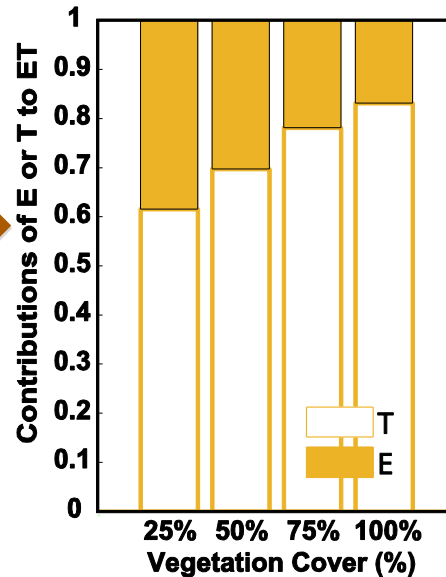


Fig. 5. An example of using laser-based stable isotope monitoring techniques to partition evapotranspiration. The left panel showed the experimental setup inside Biosphere 2 as well as the mass balance equation to calculate the evapotranspiration partition, and the right panel showed the increased transpiration/evapotranspiration ratio as vegetation cover increases, which was modified from Wang et al. (2010b).

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