Trading Carbon and Water Through Vegetation Shifts

by

John Hyungchul Kim

Graduate Program in Ecology
Duke University

Date:_______________________
Approved:

____
Robert B. Jackson, Supervisor

____
Esteban G. Jobbágy

____
Gabriel G. Katul

____
Brian C. Murray

____
Daniel d. Richter

Dissertation submitted in partial fulfillment of
the requirements for the degree of Doctor of Philosophy in the Graduate Program in
Ecology in the Graduate School
of Duke University

2011

ABSTRACT

Trading Carbon and Water Through Vegetation Shifts

by

John Hyungchul Kim

Graduate Program in Ecology
Duke University

Date:_______________________
Approved:

____
Robert B. Jackson, Supervisor

____
Esteban G. Jobbágy

____
Gabriel G. Katul

____
Brian C. Murray

____
Daniel d. Richter

An abstract of a dissertation submitted in partial
fulfillment of the requirements for the degree
of Doctor of Philosophy in the Graduate Program in Ecology in the Graduate School
of Duke University

2011

Abstract

In this dissertation I explore the effects of vegetation type on ecosystem services, focusing on services with significant potential to mitigate global environmental challenges: carbon sequestration and groundwater recharge. I analyze >600 estimates of groundwater recharge to obtain the first global analysis of groundwater recharge and vegetation type. A regression model shows that vegetation is the second best predictor of recharge after precipitation. Recharge rates are lowest under forests, intermediate in grasslands, and highest under croplands. The differences between vegetation types are higher in more humid climates and sandy soils but proportionately the differences between vegetation types are greater in more arid climates and clayey soils. My extensive field estimates of recharge under paired vegetation types in central Argentina and southwestern United States provide a more direct test of the relationships between vegetation and recharge. The field data confirm the strong influences of vegetation on recharge, and its interactions with abiotic factors, observed in the global synthesis. These results indicate that vegetation shifts have a proportionately larger potential to affect recharge in more arid climates and clayey soils.

Using the same study systems, I compare my field estimates of recharge to organic carbon stocks (in biomass, litter and soil) under the different vegetation types to evaluate tradeoffs between carbon sequestration and groundwater recharge as affected

by vegetation shifts. To determine net values of vegetation shifts, I combine the changes in carbon and water with reported economic values of the ecosystem services. Based on physiological tradeoffs between photosynthesis and transpiration in plants, I hypothesize that vegetation promoting carbon storage would reduce recharge and vice versa. Changes in water and carbon services are inversely proportional, with rain-fed cultivation increasing groundwater recharge by 13 mm/yr but decreasing carbon storage by 34 Mg/ha compared to the grasslands they replace on average, whereas woody encroachment does the opposite, decreasing recharge by 5.3 mm/yr and increasing carbon storage by 23 Mg/ha on average. In contrast, cultivated plots irrigated with ground water decrease both ecosystem services. Higher precipitation and clay content both exacerbate changes in carbon storage with grassland conversions to woodlands or croplands, whereas higher precipitation accentuates, but higher clay content diminishes, changes in recharge. Regardless of the nature of the vegetation change, most of the net values of grassland conversions are negative, with the exception of woody encroachment. Vegetation shifts represent increasing costs in the following order: woody encroachment, rain-fed cultivation, and irrigated cultivation. Values of changes in carbon are greater in magnitude than those of recharge, indicating that establishment of carbon markets may drive land-use changes in grasslands over water markets.

Lastly, I examine the effects of changes in subsurface hydrology resulting from grassland conversion to croplands on soil inorganic carbon stocks in the same U.S. study systems. I find that inorganic carbon stocks are significantly lower under both rain-fed and irrigated croplands by 100 to 1100 Mg/ha than in the grasslands they replaced. These losses are visible to at least 6 m depth in the soil profile and are uncharacteristically rapid for a carbon pool that is considered to be relatively inert. Based on the negative relationship observed between the inorganic carbon stocks and recharge rates, higher estimated exports of bicarbonates in recharge under croplands, and increasing alkalinity trends in shallow ground waters in more arid parts of Kansas and Texas derived from national long-term monitoring datasets, I postulate that increased recharge with cultivation may result in dissolution and leaching of grassland soil carbonates. These results call for detailed follow up research to confirm the observations and to determine the mechanisms, pathways and fate of carbon in the soil carbonates. The extent and value of the ecosystem services quantified here and their relationships to biotic and abiotic factors should enhance our understanding of the tradeoffs and interactions between the two services accompanying vegetation shifts.

To my parents and my sister, who have shared many of my joys and fascinations of the natural world.

Contents

List of Tables

List of Figures

Acknowledgements

For his guidance, understanding, generous support and opportunities, and exemplary leadership, I would like to express my gratitude to my advisor, Rob Jackson. I am indebted to him for his trust and respect that forms a basis for the independent thinker within me. In my time at Duke University, many scientists and thinkers have left strong impressions on me. Among them are Gaby Katul, Dan Richter, Rebecca McCulley, Dush Jayawickreme, and Esteban Jobbágy. My committee offered their time and suggestions for this dissertation at numerous opportunities, for which I am grateful.

I owe Kathleen Farley, Zbignew Kabala, Gervasio Pineiro, Marcelo Nosetto, David LeMaitre, Brian Murray, Justin Baker, Tim Bleby, and Aaron Jenkins thanks for numerous thought-provoking discussions. Roxana Aragon, Victoria Marchesini, Celina Santoni, Silvina Ballesteros, German Baldi, Isabel Miranda, and Pedro Tognetti deserve my thanks for making my experience in Argentina an enjoyable as well as a productive one.

I appreciated assistance in the field from Ricardo Paez, Antonio Brun, Marcos Jannello, Matt Cleary, and Laura Beth Konopinski and assistance in the lab from too many people to list but especially from Chinling Chen, Stewart Hoelscher, Jennifer Huang, Zhe Ma, Laura Mendenhall, Maura Nowalk, Virginia Palacios, Nancy Scott, and Will Cook. This work could not have been completed on time without their help.

For reasons of their privacy, I do not list all of the land owners here, but I want to acknowledge everyone who gave me and my field assistants access to their properties and permission to take samples for this dissertation. I would also like to thank the people at Research Extension Centers in Tribune, KS, Goodwell, OK, San Angelo, TX, Vernon, TX, Riesel, TX, and San Claudio, Buenos Aires, Argentina for their logistical support and for permission to take samples on their premises.

Financial support from EPA, NSF, a Billings fellowship, and various organizations within Duke University was crucial in allowing me to pursue my questions independently and with ample resources.

I am obliged to all the Jackson lab, Bernhardt lab, and Richter lab members for their constructive comments and discussions on formulating hypothesis and research plans and writing proposals and manuscripts. My gratitude also extends to the administrative staff in the Biology Department and Nicholas School of the Environment, who work hard to make our job easier and more enjoyable.

It would be unacceptable to conclude this section without acknowledging special thanks to Catarina Moura, my greatest happiness, for her love, patience, and support.

1. General introduction and overview

Ecosystem services sustain our societies by maintaining processes that provide basic necessities, such as water, food, and fiber, and more complex benefits such as climate regulation. Some services are recognized in their worth and importance, yet there are many that we overlook that may have just as serious consequences for us and the ecological community at large. As one of the dominant kingdoms on the Earth in terms of biomass, plants affect diverse, if not all, facets of terrestrial ecosystem processes. Vegetation shifts and land-use changes, more ubiquitous now than ever, can therefore strongly affect multiple ecosystem services. Plants have particularly strong grips on the carbon and water cycles by coupling them and facilitating the exchanges of materials between the lithosphere and the atmosphere. Carbon sequestration and water provisioning are two ecosystem services regulated by vegetation and that can mitigate climate change and freshwater scarcity, two urgent and worsening environmental problems worldwide. In the current era of the Anthropocene where globalization and increasing demands drive large-scale engineering of the planet, understanding the responses of ecosystem services to natural and anthropogenic changes becomes of paramount importance for a sustainable use of the environment.

As my contribution towards better understanding of ecosystem services in the changing world, I integrated experimental and synthesis approaches to understand how vegetation and abiotic factors affect ecosystem services, focusing on responses of carbon sequestration and groundwater recharge to land-use changes across wide ranges of climate and soil variables. I quantified the importance of vegetation type on groundwater recharge, merging existing and new field experiments in the first global synthesis of its kind. I also used the new field estimates of recharge and carbon storage to evaluate ecological and economic tradeoffs between changes in the two ecosystem services with land-use changes. Finally, I tested the hypothesis that changes in recharge resulting from land-use changes can affect otherwise-stable pools of soil carbonates by undertaking the most comprehensive inventory to date of this carbon pool under different land uses. My analyses quantified both relationships between ecosystem services and generic vegetation types and changes in ecosystem services from paired comparisons of land uses. I examined services whose relevance was well-recognized, but along the way I also discovered unforeseen interactions between water- and carbon-related ecosystem services.

I started my query with groundwater recharge, a process that replenishes essential sources of water for many ecological and human communities, but whose relationship with vegetation has traditionally been regarded as secondary to physical factors such as climate and soil texture. In Chapter 2, I synthesized recharge rates under different vegetation types, and broad ranges of climate and soil factors to show the relative importance of these variables on predicting recharge. Because I found few studies with paired comparisons of recharge under different land uses in the synthesis, I made a parallel assessment of recharge rates under paired land uses in 15 sites across Argentina and the U.S. My independent field estimates of recharge provided opportunities to confirm conclusions from the synthesis and to directly test the relationships between recharge, vegetation, and abiotic factors.

Recharge rates from Argentina and U.S. were compared to carbon stocks in the same study systems to evaluate tradeoffs between carbon sequestration and groundwater recharge in Chapter 3. I tested the hypothesis that these two services would change in opposing directions following land-use changes due to physiological coupling of carbon and water in plants and due to land management practices. I observed a clear tradeoff between the two ecosystem services with grassland conversion to croplands or woodlands. Combining the measurements of ecosystem services with economic values of the services in our U.S. study region, I calculated net values of land-use changes in terms of water and carbon.

In Chapter 4, I investigated the effect of greater recharge under croplands on soil carbonate storage. Soil carbonates have been regarded as slow-forming pools and hence ineffective sinks for atmospheric carbon, yet the large influx of soil water detected under cultivated plots compared to grasslands seemed too dramatic of a change to be irrelevant to carbonate chemistry. This research opened new questions about the cycling and fate of soil carbonates with respect to atmosphere and aqueous environments.

This work attempts to quantify relationships between ecosystem services, vegetation, climate and soil, as well as between different ecosystem services. In my search for policy-relevant results, I examined two important ecosystem services, but also uncovered unexpected relationships between water provision and carbon storage using comparisons between paired and general vegetation types. I quantified the effects of biotic and abiotic factors on important linkages and examined new interactions between carbon and water-related ecosystem services, guided by a desire to find and promote a sustainable use of the environment.

2. A global analysis of groundwater recharge for vegetation, climate, and soils

2.1 Introduction

Ground water sustains the lives of one quarter of the human population (Ford and Williams 1989; White et al. 1995) and is vital for industrial, agricultural, and recreational activities and for the health of other species and ecosystems (Postel and Carpenter 1997; Jackson et al. 2001). Its importance is most apparent but by no means limited to arid and semi-arid regions, where a paucity of surface waters often leads to greater groundwater exploitation. Given the increasing use and scarcity of ground water in many locations, and its relatively slow replenishment, sustainable groundwater use and management are necessary to meet the needs of people and ecosystems (Shiklomanov 1997; Shah et al. 2000; Vorosmarty et al. 2000).

Relationships between groundwater recharge and physical variables have long been of scientific and practical interest, traceable back to ancient Roman times (Priyadarshi 2008). Previous studies have identified climatic and geologic factors as major environmental controls on rates of groundwater recharge. In general, recharge increases with the amount and the intensity of rainfall, which both influence how much water enters soil and rocks (Lvovitch 1970; Freeze and Cherry 1979; Bredenkamp 1988; IAEA 2001; Jan et al. 2007; Stonestrom et al. 2007), and recharge decreases with increasing potential evapotranspiration (*PET*), an expression of the amount of energy available to evaporate water (Thornthwaite et al. 1957). Once in the soil, water movement is influenced by soil texture and structure, with sandier soils tending to have greater rates of recharge, and greater clay content increasing tortuosity and limiting water movement in soil (Athavale et al. 1980; Kennett-smith et al. 1994). Such general relationships are already important for use in some global and local models (e.g., Doll et al. 2003).

One aspect of recharge that is less well understood and rarely incorporated into global land-surface models is the effect of vegetation on recharge (Jackson et al. 2000; Gerten et al. 2004). Examples of key uncertainties include the primary effect of vegetation type compared to physical climate and soil variables, as well as how changes in vegetation interact with climate and soils to alter recharge. Though considerable research has examined physical factors as controls of recharge, earlier work has rarely emphasized the effects of vegetation (but see Lull and Munns 1950 and Petheram et al. 2002 for a review of Australian studies). Several studies have included vegetation in attempts to model groundwater recharge at various scales (i.e. Finch 1998; Keese et al. 2005; Doll and Fiedler 2008), though most have found or assumed the relationship to be of secondary importance compared to the effects of physical factors such as climate and soil on recharge.

Plants mediate water fluxes between soil and the atmosphere through uptake of soil water by roots and through evapotranspiration (ET) from leaves, with plant traits such as rooting depth, leaf area, and phenology affecting the magnitude and duration of these fluxes (Skiles and Hanson 1994; Neilson 1995; Milly 1997; Kergoat 1998; Peel et al. 2001; Jackson et al. 2008). Pervasive land-use and land-cover changes from anthropogenic and natural forces could have large consequences for groundwater recharge and potentially for downstream effects such as salinization (Walker et al. 1999). Building on earlier studies of land use and recharge, including studies in Australia (Petheram et al. 2002) and in arid and semi-arid regions (Scanlon et al. 2006), we examined the relative importance of vegetation in the relationship between recharge and physical factors.

We compiled a new global synthesis of groundwater recharge rates and data for different climate, soil, and vegetation types to understand how different vegetation types affect recharge. We hypothesized that vegetation would exert as strong an influence on recharge as climate and soils do. Moreover, we expected strong interactions among plant, climate, and soil factors that would create predictable patterns of recharge under different vegetation types. Among vegetation types in this review, we emphasized croplands, grasslands and forests, as shifts between these common land covers represent most of the ongoing land-use changes today (Meyer and Turner 1994; Klein Goldewijk and Battjes 1997). To test the synthesis data and to compare recharge under paired vegetation types, we also collected new field data from paired land uses across precipitation gradients in central Argentina and the southwestern U.S. We apply our findings to examine how land-use/cover changes may affect recharge across climatic and soil factors.

2.2 Conceptual model of recharge

A conceptual model of recharge suggests several important soil and climate factors that affect recharge:

$$R = \alpha \cdot P - ET - dS/dt$$

where R is recharge (mm/yr), P is precipitation and irrigation (both in mm/yr), and dS/dt is any change in soil moisture storage (mm/yr). The term α is the proportion of precipitation that would become throughfall, and ET is an evapotranspiration term that is a function of soil water availability and energy available for evaporation (mm/yr). Vegetation characteristics are likely to affect α through the interception of rainfall by leaves and branches, and affect ET through such factors as the coupling of vegetation to the atmosphere (e.g., through more leaves and/or taller vegetation stature) and to soil moisture (e.g., through deeper roots) . Studies reviewed globally and our specific sites were located in flat landscapes to minimize the effect of runoff, which is not considered in this conceptual framework.

Since available soil moisture can become either ET or R, the potential rate at which water moves through the soil matrix and therefore, out of the zone of root uptake, is another important determinant of recharge. When there is uniform matric potential, recharge is affected largely by the gravitational gradient according to the Darcy's law:

$$R = K_s \cdot (\theta / \theta_s)^{(2 \cdot b + 3)},$$

where K_s is saturated hydraulic conductivity, θ is the soil moisture below the root zone (dependent on evaporation from the soil surface, root water uptake and downward flux of water out of the root zone as determined by the water potential gradient), θ_s is soil moisture content at saturation, and b is an empirical parameter that varies with soil texture.

Because θ, θ_s and K_s are not always reported in published studies, we estimated K_s in our regression model of recharge based on soil texture information that the studies provided more frequently. Furthermore, because *ET* is dependent on available soil moisture, we used the energy available for evaporation or potental evapotranspiration (*PET*) as a proxy for *ET*. Though our approach was statistical, we chose the predictors for the regression model based on this conceptual framework. Predictors we chose were precipitation, *PET*, K_s, and seasonality of rainfall, as well as vegetation type.

2.3 Methods

2.3.1 Literature review

We examined studies of recharge and physical variables associated with land use or vegetation type, identified using literature searches involving the keywords "groundwater recharge," "deep drainage," or "residual flux," henceforth collectively referred to as "recharge" (Petheram et al. 2002; Scanlon et al. 2006). From tables, digitized graphs, and text, we recorded recharge estimates, precipitation during the study period or reported long-term mean (*P*), potential evapotranspiration (*PET*), soil texture (% clay/sand or textural classes), saturated hydraulic conductivity (K_s), vegetation type, species present, and amount of irrigation (*I*), when present. In studies where recharge estimates included data from multiple years and locations, such as those using permanent boreholes for the same vegetation and soil type, we used the mean of the estimates. A map showing the location of the studies included in our synthesis is shown in Figure 2.1. Across the dataset, 46% of the data points came from Oceania, 19% from North America, 15% from Asia, 10% from Africa, 6% from Europe, and 3% from South America.

Because we wanted to compare the effects of biological and physical variables on recharge, we excluded data from sites with significant sources or sinks for runoff, such as sinkholes, playas, and streams. Studies that estimated recharge for <1 year were also excluded from our analysis. Due to a large number (>2500) of studies in the search, we sorted results by relevance in ISI Web of Knowledge and included studies until less than two out of ten additional studies yielded data on the following variables: recharge, vegetation types, precipitation, irrigation, soil texture or K_s.

For the vegetation analyses, we divided plant types into five broad categories: cropland, grassland, woodland, scrubland, and no vegetation. Annual agricultural fields were classified as "croplands," grasslands and pastures as "grasslands," forests and woodlands as "woodlands," scrublands, heathlands, shrublands, steppes, fynbos, and savannas as "scrublands," and sparse or no vegetation areas as "NoVeg.".

Most studies did not provide data for *PET* or seasonality of rainfall, and these variables were therefore obtained from the high spatial resolution (10′ x 10′) Climate Research Unit global dataset (New et al. 2002; http://csi.cgiar.org/cru/), using locations of sites given in the studies. We calculated *PET* using the Penman-Monteith equation from the monthly climate dataset. Seasonal amplitude of rainfall and synchrony of rainfall with *PET* are both important considerations in the water balance (Milly 1994; Potter et al. 2005). We defined two variables associated with seasonality of rainfall: 1) the difference between the maximum and minimum mean monthly rainfall (amplitude) and 2) the number of months between maximum mean monthly temperature and precipitation (*Phase*). Water input (*WI*), Aridity index (*AI*), and potential water excess (*PWE*) were calculated as *P*+*I*, (*P*+*I*)/*PET*, and *P*+*I*–*PET*, respectively.

We estimated saturated hydraulic conductivity using soil texture classes (Rawls et al. 1982). Where different soil horizons existed within the depth of soil examined, the estimated K_s for the top layer was used. To ensure that our estimates of *PET* and K_s were reasonable, we compared them to values of *PET* and K_s from the subset of studies where reported. Our estimates matched well with reported *PET* and K_s across the studies (n=220, 71; R^2=0.71, 0.70; P<0.0001, 0.0001, for *PET* and log(K_s) respectively).

Proportional recharge (P/WI) between each pair of vegetation and soil types was compared using a Kruskal-Wallis test. Proportional recharge was used for this analysis instead of recharge because it allowed comparisons after controlling for the effect of *WI*. A nonparametric test was used because the data were not normally distributed. Grouping the data into two soil texture categories was done for some analyses to examine the effects of soil texture on recharge more easily: "clays" were defined as soils whose estimated K_s was <0.25 m/day (silt loam and more clayey soils) and "sands" were texture classes with higher K_s.

We tested all climate variables (*WI*, *AI*, *PET*, *PWE*, *seasonality*) and models (linear, logarithmic, exponential and sigmoidal) to determine the best predictor of recharge. This approach was taken to choose a single best predictor variable to easily represent and compare the synthesis data with new field data (see below). Due to the relatively low sample size (n<50) and limited ranges in climatic variables (e.g. *WI* 159-937 mm/yr) for scrublands and NoVeg, those two vegetation types were excluded from the curve-fitting and regression analyses (see below). All of the models tested were susceptible to the influence of relatively few data points at the most humid end of our data range (n=5 for the hyper-humid region data); we therefore limited our curve-fitting and regression analyses to a dataset without these extremely humid regions.

We tested for effects of *WI*, *PET*, vegetation types, K_s, seasonality of rainfall, and accompanying interactions on recharge using multiple regression analyses. Because of heteroscedasticity, we log-transformed recharge and examined appropriate models to relate recharge to each of the predictor variables. The Breusch-Pagan test was used to test for homoscedasticity, and log-transformation of recharge gave the most homoscedastic relationships with the predictor variables out of all the transformations of recharge values (untransformed, natural log, and square root). We examined appropriate models (linear, exponential, logarithmic) to relate recharge to the predictor variables individually and found that a logarithmic model explained the most statistical variation in log-transformed recharge using *WI* and *PET* and that a linear model maximized the fit of log-transformed recharge with K_s and seasonality of rainfall (amplitude and *Phase*). Thus, we log-transformed *WI* and *PET* to linearize them with respect to log-transformed recharge for the multiple regression. We used *WI* and *PET* instead of *PWE* or *AI* for our multiple regressions to tease out the relative importance of *WI* and *PET.* Stepwise regression with whole effects was used to determine which main and interaction terms to retain in the model and to determine the relative importance of each term on recharge.

Finally, to test the reliability and predictive capability of our regression model, we used 3-fold cross-validation, in which a model based on a subset of the data is tested against the remainder of the data (Kohavi 1995).

2.3.2 Site description

In addition to the literature synthesis, we collected an extensive new field dataset as an independent test of our global dataset, using paired comparisons of adjacent vegetation types in Argentina and the United States. In Argentina, we located six sites in the Pampas on relatively level landscapes across a precipitation gradient that ranged from 382 to 1215 mm/yr. When available, rain-fed cultivation and woody plant invasion (WPI) plots were paired to an adjacent or nearby (<1km) natural grassland plot at each site. Cultivation and WPI plots correspond to cropland and woodland vegetation designations in our literature synthesis (Table 2.1,2.2).

We also selected five sites along a precipitation gradient (407-860 mm/yr) in the southern Great Plains of the United States. Land uses selected as paired plots were natural grasslands, rain-fed crops, and irrigated crops. In both U.S. and Argentina, most plots have 30+ years of relatively continuous land-use history (Table 2.1). Land-owners or farm managers were surveyed for land-use history at each site, including cropping schemes (species, rotations) and fertilizer/pesticide/irrigation inputs. Tree stand ages were verified with aerial photos or tree ring cores taken during our sampling campaign (2008-2010). Precipitation data were obtained from long-term (30+ years) records maintained by weather stations onsite by the farm managers (INTA ; NOAA) or from separate stations 1-30 km away.

In addition to our new field data, we also estimated additional recharge rates based on soil chloride data from four paired grassland and woody encroached sites located across a precipitation gradient in the southwestern U.S. Detailed descriptions for these latter five sites are available in Jackson et al (2002) and McCulley et al (2004). We collectively refer to these and the Southern Great Plains sites as our southwestern U.S. sites.

2.3.3 Soil sampling

At the Argentinean sites, soil samples were taken using three to eight boreholes 6-9-m deep or to the depth of groundwater as well as 4-6 shallow cores (30-cm deep) at each land-use plot. Augered samples were taken every 20 cm to 1 m, then every 30 cm to 4 m depth, then every 50 cm afterwards. Soil samples were homogenized and subsampled in the field and then frozen until analysis. At our U.S. sites, we used a direct-push mechanical coring rig (Geoprobe Systems, Salina, KS) for five to eight cores per plot to a 8.5 m depth. At only one plot near San Angelo, TX were soil samples not retrieved to 8.5 m depth because of indurated caliche found around 5 m depth that blocked further coring. Soil cores were weighed in the field, subsampled for soil moisture and bulk density using intact cores and for elemental analysis using homogenized soil cores, then shipped to Duke University for analysis.

In the laboratory, soil samples were oven-dried for gravimetric moisture content and analyzed for chemical constituents. 1:1 ratio of dried and homogenized soil samples were shaken with equal weight of double deionized water for four hours. The mixture was centrifuged, the supernatant filtered, and the filtrate analyzed for anion contents (Cl^-, Br^-, NO_3^-, SO_4^{2-}, and PO_4^{3-}) by ion chromatography (Dionex ICS-2000). Cl^- concentrations in the soil porewater were calculated by dividing the soil Cl^- contents (mg Cl/kg soil) by gravimetric soil moisture. Soil texture was determined by the pipette method (Klute 1986) and ranged in texture from sandy to clay (Table 2.1).

2.3.4 Groundwater recharge calculations

Recharge rates at our sites were estimated by chloride mass balance (CMB) from soil samples in the unsaturated zone (Allison and Hughes 1983). Total atmospheric inputs of Cl^- were obtained from Piñeiro et al. (2007) and Santoni et al. (2010) for the Argentinean sites and from deposition networks in the U.S. (NADP 2011; US EPA 2011). To estimate Cl^- deposition rates at our sites, we used distance from the ocean (Junge and Werby 1958; Keywood et al. 1997), which correlated well with Cl^- deposition at our Argentina and U.S. study regions (Figure 2.2; $P<0.001$, 0.001; n=12, 6; R^2=0.99, 0.99 for U.S. and Argentina, respectively). Dry deposition at U.S. sites was estimated based on the relationship between the dryfall:wetfall ratio in precipitation across the study region (Figure 2.3; $P<0.001$, n=9, R^2=0.82). Anthropogenic inputs of Cl^- under cultivation were calculated by multiplying Cl^- contents in the fertilizer, pesticide, and irrigation samples obtained at the sites with the average application rates revealed in the surveys (Table

2.2). Assuming steady-state conditions, recharge rate was calculated using the following mass balance equation:

$$Q_{in} \times Cl_{in} = Q_{out} \times Cl_{out}$$

where Q_{in} is the average volume of rain and irrigation water entering the root zone per ground area per year ($mm^3 \cdot mm^{-2} \cdot yr^{-1}$), Cl_{in} is the average atmospheric and anthropogenic Cl^- input expressed as concentration in precipitation ($mg \cdot mm^{-3}$), Q_{out} is the volume of water exiting the root zone per ground area per year ($mm^3 \cdot mm^{-2} \cdot yr^{-1}$), and Cl_{out} is the concentration of Cl^- in the soil water exiting the root zone ($mg \cdot mm^{-3}$). Assuming no dispersion and diffusion of Cl^- and Cl_{out} to be the average Cl^- concentration of soil porewater below the root zone, Q_{out} is the groundwater recharge rate (mm/yr). The root zone was defined as the top 2.1 m, below which we found a linear relationship between cumulative Cl^- and cumulative soil moisture content, except for some cultivated plots where we assumed the root zone to be the top 1 m (Phillips 1994). At the Tribune, Vernon, and Riesel sites, where we did not observe complete leaching of the original Cl^- peak with cultivation, we also used the chloride displacement method to calculate recharge rates based on the migration of the original grassland Cl^- and on water profiles (Walker et al. 1991). Calculations for the Cl^- tracer displacement (CTD) method was:

$$q_{CTD} = \theta \frac{z_1 - z_2}{t_1 - t_2}$$

where q_{CTD} is the recharge rate (mm/yr), and z_1 and z_2 are the depths (mm) of the chloride fronts corresponding to land uses at years t_1 (new: rain-fed cultivation) and t_2

(old: grassland), and θ is the average soil moisture content over this depth interval. 8.5 m was used as z_1 for profiles without a clear chloride peak, providing a lower-bound estimate for recharge.

We compared results from our global dataset and independent field data to estimate the influence of vegetation shifts on recharge for different water availabilities globally. We calculated absolute and relative changes in recharge with land-use changes. For the field data, relative change (Δ) was defined as:

$$\Delta = \frac{\text{Recharge}_{\text{veg2}} - \text{Recharge}_{\text{veg1}}}{\text{Recharge}_{\text{veg1}}}$$

For the global synthesis data, we used recharge predicted from the best-fit curves to calculate absolute and relative differences in recharge between vegetation types.

2.4 Results

Vegetation and soil types had strong effects on proportional recharge (*R*/*WI*) globally (chi-square=73.7, 13.9, respectively; $P<0.0002$). On average, proportional recharge was 0.18, 0.11, 0.08, 0.06, and 0.05 under NoVeg, croplands, grasslands, woodlands, and scrublands respectively ($P<0.0005$ for all pairwise comparisons except scrublands to woodlands and grasslands to croplands; Table 2.2). Sandy soils had twice as much proportional recharge as clayey soils on average.

Potential water excess (*PWE*) fitted to an exponential model was the best single predictor of recharge across the dataset (Figure 2.4). Recharge increased with *PWE* for croplands, grasslands, and woodlands in both sands and clays (average R^2=0.52; P<0.0001 for all vegetation-soil types; Figure 2.4). Differences among vegetation types were evident in the fitted curves. For example, at *PWE* of –250 mm/yr in clays, predicted recharge under croplands, grasslands and woodlands were 112, 61, and 35 mm/yr respectively (n=220, 138, and 109 respectively).

Water inputs (*WI*) in the multiple regression explained 29% of the statistical variation in recharge across the dataset (P<0.0001, Table 2.3). Other significant variables in the order of decreasing importance were vegetation type (16%, P<0.0001), *PET* (12%, P<0.0001), and K_s (6%, P<0.0001), with amplitude and phase (seasonality) of rainfall contributing a statistically significant but minor 1% of variation (P<0.0001; Table 2.3). Recharge increased with *WI*, K_s and seasonality of rainfall and decreased with *PET*. Overall, recharge was greatest under croplands, about two- and fifteen-times greater than for grasslands and woodlands, respectively (P<0.0001; Table 2.3; *Veg* term).

The interaction terms of vegetation types with climate or soil variables collectively explained an additional 8% of the variation in recharge (Table 2.3; Figure 2.5). Of all vegetation types, cropland recharge increased the most with *WI*, but grassland recharge increased the most with increasing K_s and decreasing *PET*. In contrast, woodland recharge was the least sensitive to K_s and *PET*, indicating that recharge under different vegetation types responded differently to climate and soil factors. These responses accentuated differences in recharge between vegetation types in humid region and in sandy soils (Figure 2.5). The cross-validation analysis of the regression model and the dataset produced comparable results, giving confidence in the model's reliability (Figure 2.6).

Our new field dataset from central Argentina and the southwestern United States independently confirmed the strong differences in recharge for grasslands, croplands, and woodlands that the global synthesis revealed. Croplands had significantly lower average soil porewater Cl^- concentrations below the root zone (>2.1 m) while woodland plots had significantly higher soil porewater Cl^- compared to those in their grassland pairs (Table 2.4, Figure 2.7; Signed Wilcoxon test; $P<0.0020$, 0.0039 for Grass-Crop and Grass-Wood comparisons, respectively). This result indicated that the greatest recharge occurred under croplands, intermediate recharge occurred under grasslands, and the lowest recharge occurred under woodlands. The strong biological control over soil water fluxes is in close agreement with our global review (Figure 2.8, Table 2.4). Crop cultivation using ground water as an irrigation source resulted in very high net discharge of ground water (Table 2.4).

Our field dataset also confirmed the interactive effect of vegetation and water availability on recharge that the global synthesis revealed. For our global synthesis, absolute differences in recharge between vegetation types using *PWE* as the best-fit predictor variable were small in arid climates and grew with increasing *PWE* and were larger in sandy soils than in clays (Figure 2.8). However, relative differences were largest in arid climates and in clays (Figure 2.9), suggesting that proportionately greater hydrological effects of land-use change may occur in more arid regions and in clayey soils. Similarly to the global synthesis, our new field estimates of recharge gains or losses due to land-use conversion of natural grasslands increased in magnitude with *PWE* (Figure 2.8), revealing interactions between land use and the abiotic environment in determining recharge. As in the global synthesis, our field-based estimates of *relative* changes in recharge showed an increasing importance of vegetation effects towards lower precipitation and higher clay content areas, suggesting that while land-use changes have potentially large effects on recharge in humid and coarse-textured regions, vadose zone processes may be particularly sensitive to land-use changes in relatively arid and fine-textured areas. (Figure 2.8, 2.9).

2.5 Discussion

Although the role of vegetation in global terrestrial water fluxes is well-recognized (Hutjes et al. 1998; Kucharik et al. 2000; Arora 2002; Jackson et al. 2005), this synthesis is to our knowledge the first attempt to quantify the relative importance of vegetation on recharge rates globally. Vegetation was the second most powerful predictor of recharge after water input (*WI*), explaining about 1.3- and three- times as much variation in recharge as potential evapotranspiration (*PET*) and saturated hydraulic conductivity (K_s), respectively, indicating that vegetation types are often more important for determining recharge than most physical variables are (Table 2.3). As a result, vegetation should be one of the key components of analyses or models addressing scales sufficiently large to include multiple vegetation types.

The treatment of vegetation parameters in global land-surface models are sometimes cursory and are rarely process-based with regard to recharge (Gerten et al. 2004). Our global synthesis should help parameterize such models and could contribute as inputs or for independent testing of global water balance or climate models. For example, studies modeling the reciprocal effects of ground water on climate (e.g., Niu et al. 2007) may benefit from better constrained estimates of recharge under different vegetation types.

Changes in recharge with land-use changes in our field data followed the patterns of recharge observed under different vegetation and soil types across our global synthesis (Figure 2.8,2.9). Overall, agreement between the field and synthesis results suggests that vegetation is responsible for a large portion of the variation in recharge and that distinct patterns of recharge between vegetation types are typically clear and reproducible when covarying site factors such as soil properties are controlled for. Agricultural conversion of grasslands or woodlands would therefore likely bring about greater recharge, whereas woody plant invasion or afforestation into grasslands or croplands would likely reduce recharge. These hydrological changes may be especially severe for land-use changes to and from woodlands, as the capacity of woody plants to limit recharge leads to large differences in recharge between woodland and the other vegetation types (Figure 2.4, Table 2.3). Loss of renewable water yield to planted or invading woody plants could be detrimental to groundwater-dependent communities, both human and natural, over long time scales. In contrast, cultivation generally increases recharge but may pose a risk of salinization or degradation of groundwater quality in some regions through associated leaching of salts in the vadose zone (Smettem 1998; Boumans et al. 2005; Jobbagy and Jackson 2007; Scanlon et al. 2007). Such disruptions to the hydrological cycle should be recognized in land management and policy decisions.

The effect of vegetation on recharge was further evident along the entire climate gradient and across soil types (Figure 2.4). In our synthesis, we observed large absolute differences in recharge among vegetation types in mesic regions (high *WI*, low *PET*) and in sands (high K_s) but larger relative differences in arid climates and in clays (Figure 2.8, 2.9). Relative differences between grasslands and the other vegetation types in clays, for example, were as much as –70% and +250% (woodlands and croplands, respectively) in arid climates compared to only –20% and +60% in humid areas (Figure 2.9). Though the large absolute differences in recharge between vegetation types in humid climates highlight the importance of land-use changes on water yields in these climates, large relative differences in drier climates forecast proportionately important hydrological changes in arid regions, as observed previously for stream flow (Farley et al. 2005). Mirroring the synthesis data, the observed 70% reduction of grassland recharge with woody plant invasion and >500% gain in recharge with cultivation of arid grasslands with clayey soils in our field data indicate that near-complete loss of ground water recharge or flushing of accumulated vadose zone solutes may be possible with land-use changes (Figure 2.8,2.9). The different responses of recharge among vegetation types to climate and soils warrant careful consideration of these interactions to avoid adverse hydrological consequences of land-use changes.

Vegetation type explained a similar amount of variation in recharge as important physical variables did, and its interactions with physical variables contributed additional explanatory power. Recharge was correlated with high K_s (Table 2.3), but we observed this effect primarily in grasslands, which have relatively shallow root systems (2.5m; Canadell et al. 1996). The analogous increases in woodland recharge were less pronounced. Woody plants grow deeper roots in areas with sandy soils (high K_s), in part to capture water throughout the soil profile (Schenk and Jackson 2002, 2005); these deep woody roots may limit recharge despite higher K_s. In croplands, with the shallowest expected rooting depth (generally <2m), recharge was generally higher than for other vegetation types but did not vary substantially with K_s. This result may be due to particular management practices in croplands, such as tillage increasing deep drainage and weakening the overall positive effect of K_s on recharge under croplands (Daniel 1999; Scanlon et al. 2008). Interactions between vegetation and physical variables such as K_s and *PET* collectively explained >8% of the statistical variation in recharge and helped identify potential mechanisms responsible for differences in recharge among vegetation types.

Irrigation is often used to enhance crop productivity and to meet the increasing food demand given decreasing productive land area (Kendall and Pimentel 1994), but it also causes a large net discharge of ground water, as we observed at our southwestern U.S. sites. A simple exercise, assuming rain-fed croplands represent the upper limits for recharge and irrigation sources ground water, may aid landscape-level land management decisions. Building on our global synthesis, we observe that despite being the land use with highest recharge, rain-fed cultivation only allowed marginal recharge compared to net discharge (irrigation – recharge) of irrigated cultivation (Figure 2.10). Across a gradient of water availability, the area of rain-fed cultivation needed to sustainably supply ground water for1 ha of irrigated agriculture decreases from 70 ha in arid to 0.5 ha in humid climates (Figure 2.10), providing first-order approximation of irrigated:rain-fed cropland ratio necessary for sustainable groundwater management. It points to challenges associated with providing enough ground water for irrigated crops in a sustainable fashion, especially in more arid regions where the aridity results in both larger irrigation needs and lower recharge rates under rain-fed cultivation.

For the range of potential water excess (*PWE*) and the recharge values analyzed, the exponential model gave the best overall fit but should not be extrapolated beyond the ranges presented in this study. For instance, with inclusion of the very limited data from perhumid regions, the sigmoidal model gave the best overall fits (data not shown), indicating that the increase in recharge with *PWE* may taper off in very humid regions due to increasing importance of runoff on the water balance (Milly 1997). Moreover, irrigation (*I*) reported at most of the sites were often estimates without long-term monitoring, introducing uncertainty in our observed relationship between recharge and *WI*. The average uncertainty associated with irrigation from studies that reported ranges of irrigation was about 190 mm/yr. Though the effect of this uncertainty on estimated parameters of our multiple regression were not statistically significant (data not shown), high explanatory power of *WI* in our model highlights the importance of obtaining best possible irrigation and precipitation data for recharge predictions.

In conclusion, vegetation and its interactions with other factors have a strong effect on groundwater recharge, explaining ~24% of global variation in recharge—more than other variables except water input (*WI*). An average of 11% of *WI* becomes recharge under croplands, whereas only 8% and 6% do under grasslands and woodlands, respectively. Vegetation types had predictable effects on groundwater recharge, and the differences in recharge among vegetation types also varied predictably across the climate and soil variables. Independent field estimates of recharge under paired land-use plots confirmed our global synthesis results and provided a direct test of relationships between vegetation and recharge. Significant gains and losses in recharge are possible with conversion of natural vegetation to crops and with afforestation, respectively, and absolute changes in water yield accompanying land-use changes are likely to be larger in humid or sandy areas. However, proportionately large *relative* hydrological consequences await land-use changes in arid or clayey regions, as observed previously for stream flow (Farley et al. 2005). Quantifying and predicting changes to water yield from land-use changes are necessary steps for sustainable and holistic management of water resources; our results highlight importance of land-use change for the vadose zone and groundwater resources.

Table 2.1: Site information for our new field data in Argentina and the southwestern U.S.

Site	Lat., long. (degrees)	Precip. (mm/yr)	Soil[†]	Vegetation type[‡]	Years since change[§]
Nahuel Mapa	-34.8, -66.2	382	Fine sand	G, W	80
Caldenadas	-33.8, -65.8	506	Fine sand	G, W, C	60, 10
Dixonville	-34.7, -65.5	525	Fine sand	G, W, C	60, 15
Parera	-35.1, -64.5	682	Loam	G, W, C	100+, 80
San Claudio	-35.9, -61.2	1011	Sandy loam	G, C	40
San Antonio	-34.2, -59.4	1219	Loam	G, W, C	40, 60
Sevilleta	34.3, -106.7	277	Turney loam, sandy loams	G, W	50
Goodwell	36.6, -101.6	407	Gruver clay loam	G, C, C+I	60, 60
Tribune	38.5, -101.6	479	Richfield silt loam	G, C, C+I	30, 50
San Angelo	31.4, -101.3	514	Angelo clay loam	G, C, C+I	100, 40
Quanah	34.3, -99.8	679	Sagerton clay loam	G, C, C+I	100, 60
Vernon	33.9, -99.4	660	Tillman clay loam	G, W	40
Riesel	31.5, -96.9	890	Heiden Clay	G, W, C	100+, 100+
Engeling	31.9, -95.9	1070	Loamy fine sand	G, W	50

[†]Soil texture based on samples from the top 1 m of the soil profile.
[†]G, W, C, and C+I denote vegetation types sampled: grassland, woodland, cropland, and irrigated cropland, respectively.
[§]Number of years since land-use conversion of grassland. The numbers listed correspond to the order of land-use changes given in the previous column.

Table 2.2: Comparison of the proportion of water input that becomes recharge (R/*WI*) and potential water excess (*PWE*) among vegetation and soil types.

Veg[†]	**R/*WI***	***PWE* (mm/yr)**	**n**
Crop	0.111 ± 0.007[a‡]	-677± 32[a]	220
Grass	0.083 ± 0.009[b]	-637± 41[a]	138
Scrub	0.049 ± 0.011[c]	-1116 ± 56[b]	73
Wood	0.062 ± 0.009[c]	-475 ± 46[c]	109
NoVeg	0.178 ± 0.03[a‡]	-1009 ± 77[b]	39
Soil[†]			
Clays	0.073 ± 0.007[a]	-606 ± 36[a]	205
Sands	0.103 ± 0.006[b]	-763 ± 26[b]	374

[†]Veg denotes vegetation types, soil denotes soil types.
[a],[b],[c] denote significant differences between each pair (mean ± standard error) within the vegetation or soil types using a Kruskal-Wallis test ($P<0.0061$ for all significantly different comparisons) for *R/WI* and using a Student's t-test for *PWE* ($P<0.023$).
[‡]Comparison of proportional recharge between cropland and no vegetation is marginally significant ($p<0.07$).

Table 2.3: Results from stepwise and least squares multiple regressions of log-transformed recharge.

Terms†	Parameter estimates‡	Stepwise Seq. SS§	R^2	F-test F ratio	P
Intercept	2.87±1.88				
log(*WI*)	2.71±0.159	626	0.29	292	<.0001
Veg	1.19±0.0762, 0.419±0.0857, -1.61±0.0896	336	0.45	193	<.0001
log(*PET*)	-2.61±0.190	260	0.57	189	<.0001
K_s	0.0002336±0.0000258	134	0.63	82	<.0001
Veg×log(*WI*)	-1.01±0.157, -0.510± 0.178, 1.52±0.188)	119	0.69	37	<.0001
Veg×log(*PET*)	0.993±0.219, 0.172±0.242, -1.17±0.237	43	0.71	16	<.0001
K_s×*Phase*¶	-0.0000597±0.0000131	17	0.72	21	<.0001
log(*PET*) ×K_s	0.000304±0.000071	22	0.73	18	<.0001
Amplitude¶	0.00408±0.00105	12	0.73	15	0.0001
Veg×K_s	10^{-5}×(-11.6±3.38, 8.17±3.58, -3.39±3.79)	18	0.74	6	0.0020
log(*WI*)× *Phase*	-0.188±0.0668	10	0.75	8	0.0051
Phase	0.0577±0.0282	6	0.75	4	0.0413

†Interaction terms involving continuous variables are centered around their means for computational purposes, e.g. *Veg*×(log(*WI*)–6.39). Average values for the interaction terms were 6.39, 7.11, 1912, 3.19 for log(*WI)*, log(*PET*), K_s, and *Phase*, respectively. Factors in the stepwise regression were selected using Bayesian Information Criterion (Schwarz 1978).

‡Parameter estimates are given for the three vegetation types (cropland, grassland, and woodland, respectively) ± standard errors.

§Sequential sum of squares.

¶*Amplitude* denotes difference between maximum and minimum mean monthly precipitation, and *Phase* refers to the number of months between maximum mean monthly precipitation and temperature.

Table 2.4: Cl^- inputs and soil Cl^- values used for the groundwater recharge calculations.

Site	Vegetation type	Cl^- input N (mg/L)[†]	Irrigation (mm/yr)	Cl^- input A (mg/L)[‡]	Soil water Cl^- (mg/L)[§]	Recharge[¶] rate (mm/yr)
Nahuel	Grass	0.437			34.5	4.81
Mapa	Wood				142.6	1.16
Caldenadas	Grass	0.285			19.5	7.38
	Wood				62.6	2.30
	Rain-fed			—	11.7	12.3
Dixonville	Grass	0.357			9.2	20.5
	Wood				17.1	11.4
	Rain-fed			—	6.5	28.8
Parera	Grass	0.349			7.75	29.7
	Wood				9.7	23.8
	Rain-fed			—	4.0	57.6
San Claudio	Grass	0.287			13.5	21.0
	Rain-fed			—	7.5	41.0
San Antonio	Grass	0.348			12.1	30.1
	Wood				16.3	22.4
	Rain-fed			—	4.6	79.0
Sevilleta	Grass	0.244			1875	0.037
	Wood				4429	0.012
Goodwell	Grass	0.109			564	0.078
	Rain-fed			0.001	38.8	1.45
	Irrigated		432	16.5	247	59.6
Tribune	Grass	0.089			255	0.317

	Rain-fed			—	30.3	1.55 (4.76)
	Irrigated		584	5.3	47.4	146
San Angelo	Grass	0.194			266	0.47
	Rain-fed			—	45.6	3.17
	Irrigated		254	112	709	60.0
Quanah	Grass	0.149			278	0.36
	Rain-fed			—	70	2.86 (6.36)
	Irrigated		610	243	1071	279
Vernon	Grass	0.161			325	0.34
	Wood				1000	0.14
Riesel	Grass	0.297			137	2.4
	Rain-fed			—	37	7.2 (9.0)
	Wood				330	0.76
Engeling	Grass	0.302			11.7	28.8
	Wood				23.1	14.0

†N: Natural Cl^- input from atmospheric deposition, expressed as mg/L in precipitation.
‡A: Anthropogenic Cl^- inputs from fertilizers, pesticides and irrigation, expressed as m/L in total water input (precipitation + irrigation), for cultivated sites only. Most of the 30+ agricultural chemicals analyzed were not significant sources of Cl^-. — denotes negligible Cl^- inputs (<0.001 mg/L) from fertilizer and pesticide applications.
§Average Cl^- concentration in the soil pore water below the root zone (>2.1 m)
¶Average recharge rates based on chloride mass balance are presented, with those based on chloride tracer displacement method in parentheses.

Figure 2.1: Map showing locations of the study sites included in the global synthesis.

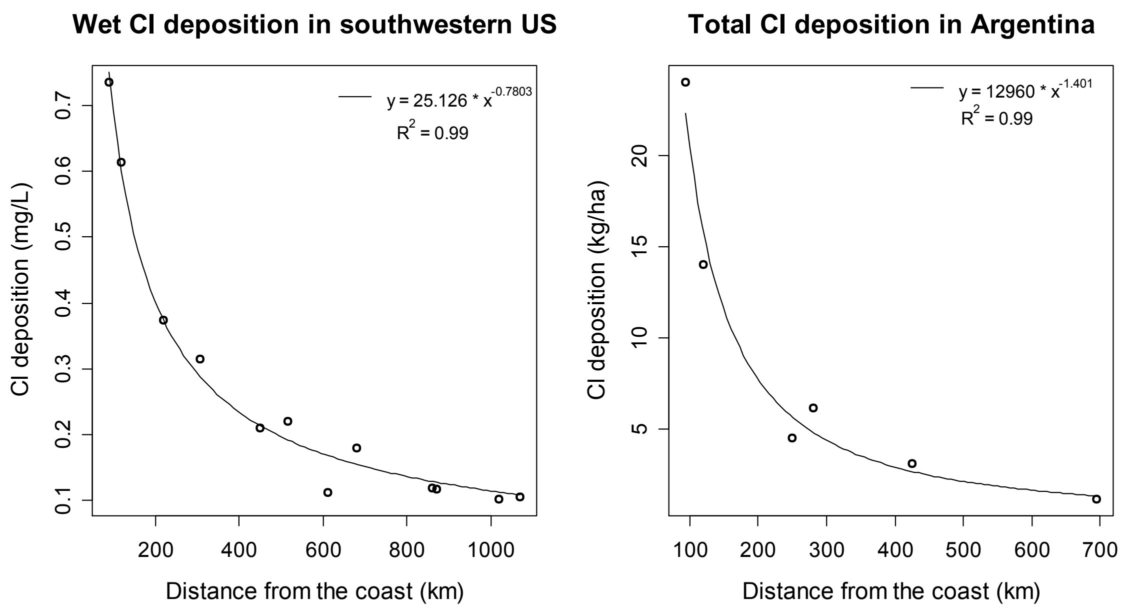

Figure 2.2: a) Wet Cl^- deposition from monitoring networks in southwestern U.S. and b) total Cl^- deposition from monitoring networks in Argentina. Only monitoring sites with 20+ years of data were included in the analysis for the U.S. network.

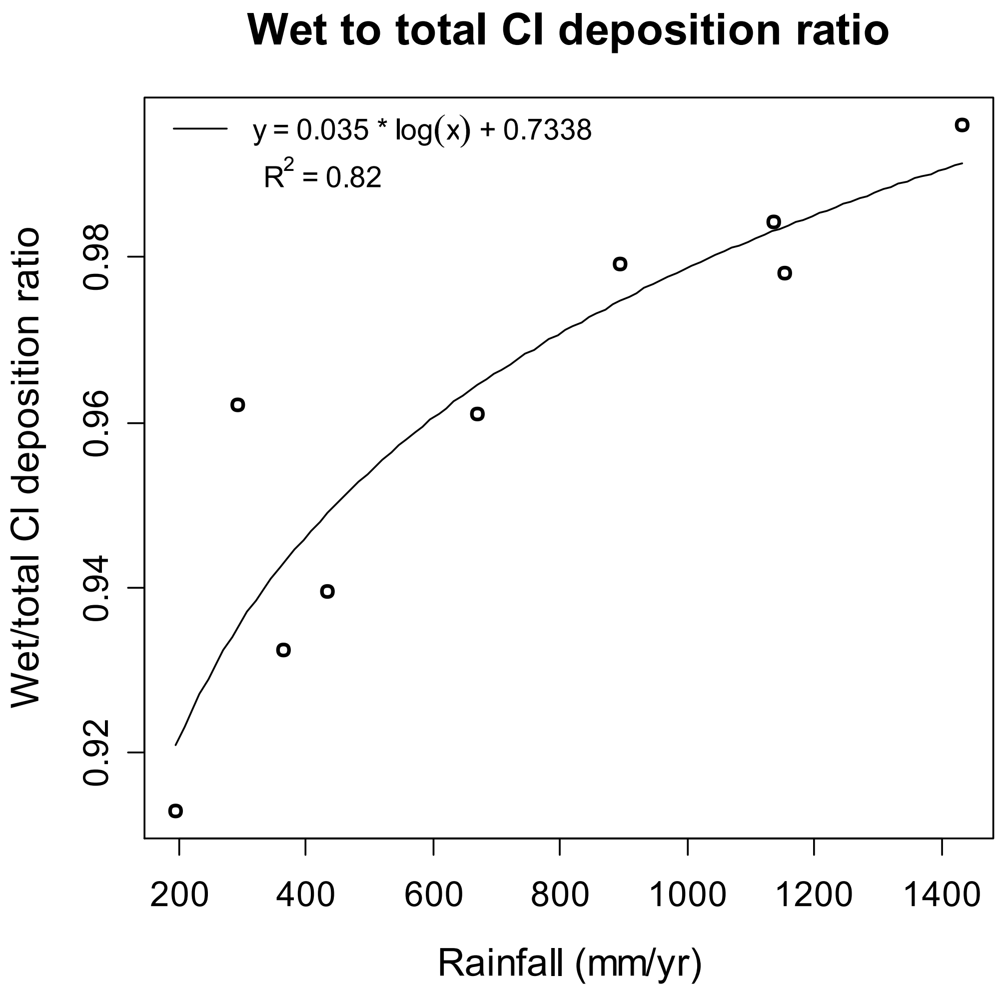

Figure 2.3: Relationship between mean annual precipitation and the ratio of wet to total Cl^- deposition from monitoring networks in the southwestern U.S. used to estimate total Cl^- deposition at our U.S. sites.

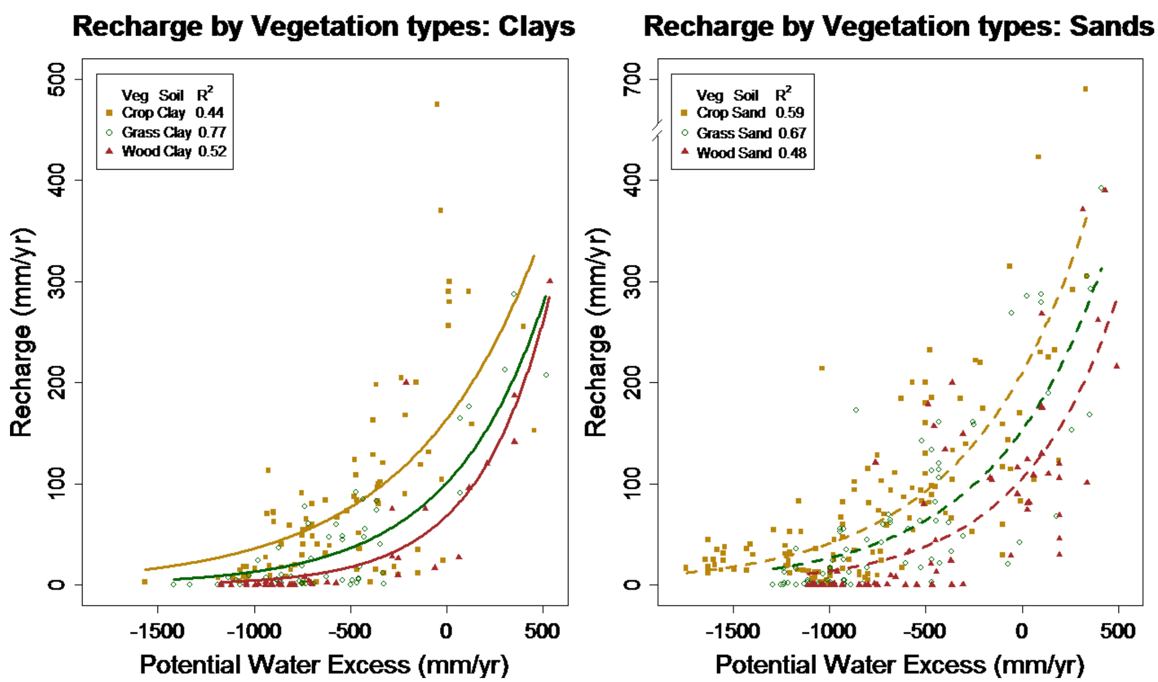

Figure 2.4: Recharge and potential water excess (*PWE*) fitted to an exponential model for three vegetation types in two soil types.

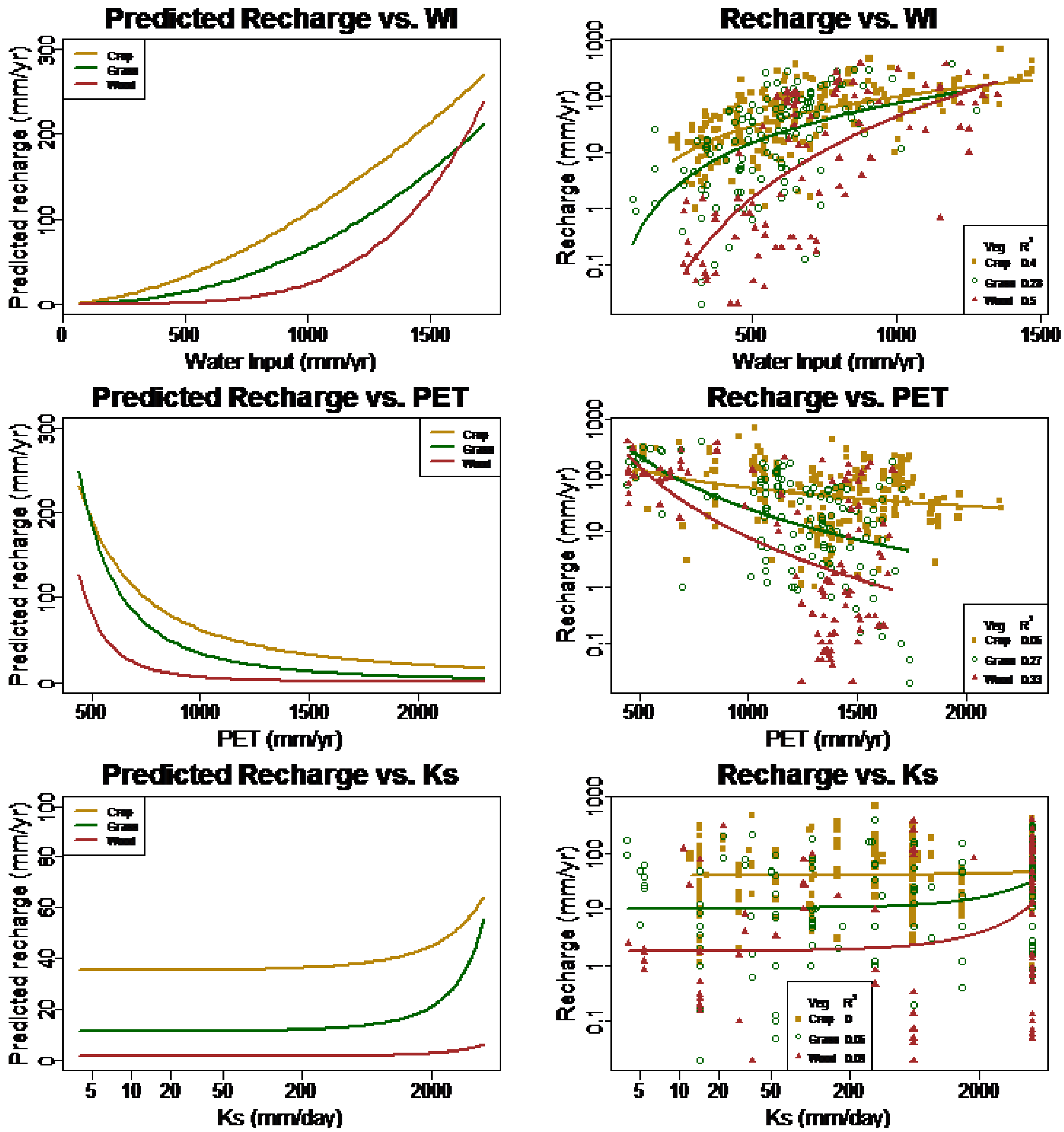

Figure 2.5: (a,b,c) Predicted recharge from interaction terms of vegetation types and physical variables (*WI*, *PET* and K_s; Table 2.3) in the multiple regression analysis, and (d,e,f) log-transformed recharge fitted to the same variables in the dataset. Recharge values were predicted from the multiple regression model holding all other terms constant around their means. As described in the methods, log-transformed recharge was fitted without data at the very highest values of *WI* because of insufficient data across vegetation types. Note the different y-axis scales.

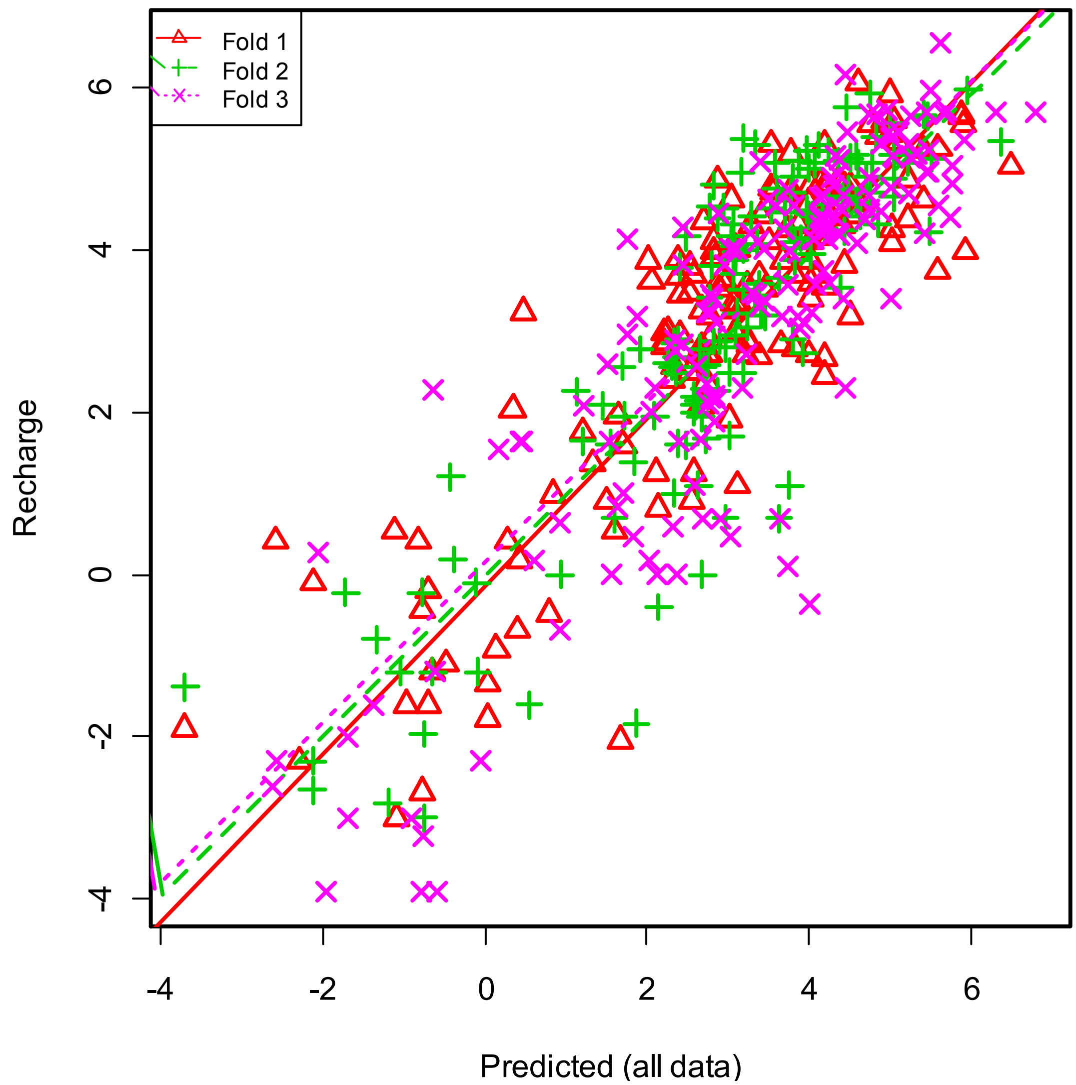

Figure 2.6: Observed vs. predicted recharge rates (log-transformed) from 3-fold cross-validation of our multiple regression model. We performed cross-validation to test the robustness of our model and our data and to test the model's predictive power. Fold 1,2,3 represent models fitted without a third of the data corresponding to lines and points of same colors. Similarity in the lines indicates good reproducibility of our regression model.

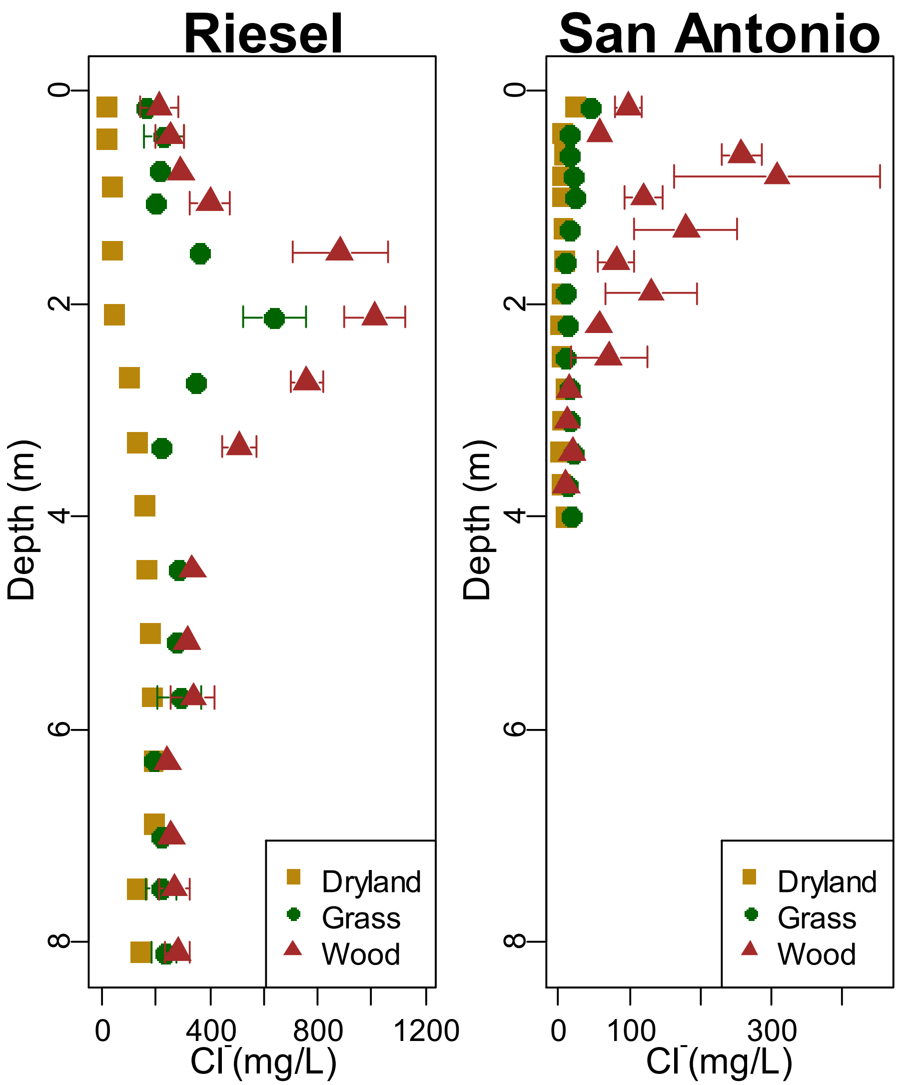

Figure 2.7: Depth profiles of chloride concentration in soil porewater profiles under three paired land uses at a) a site in the southwestern U.S. (Riesel), and b) a site in Argentina (San Antonio).

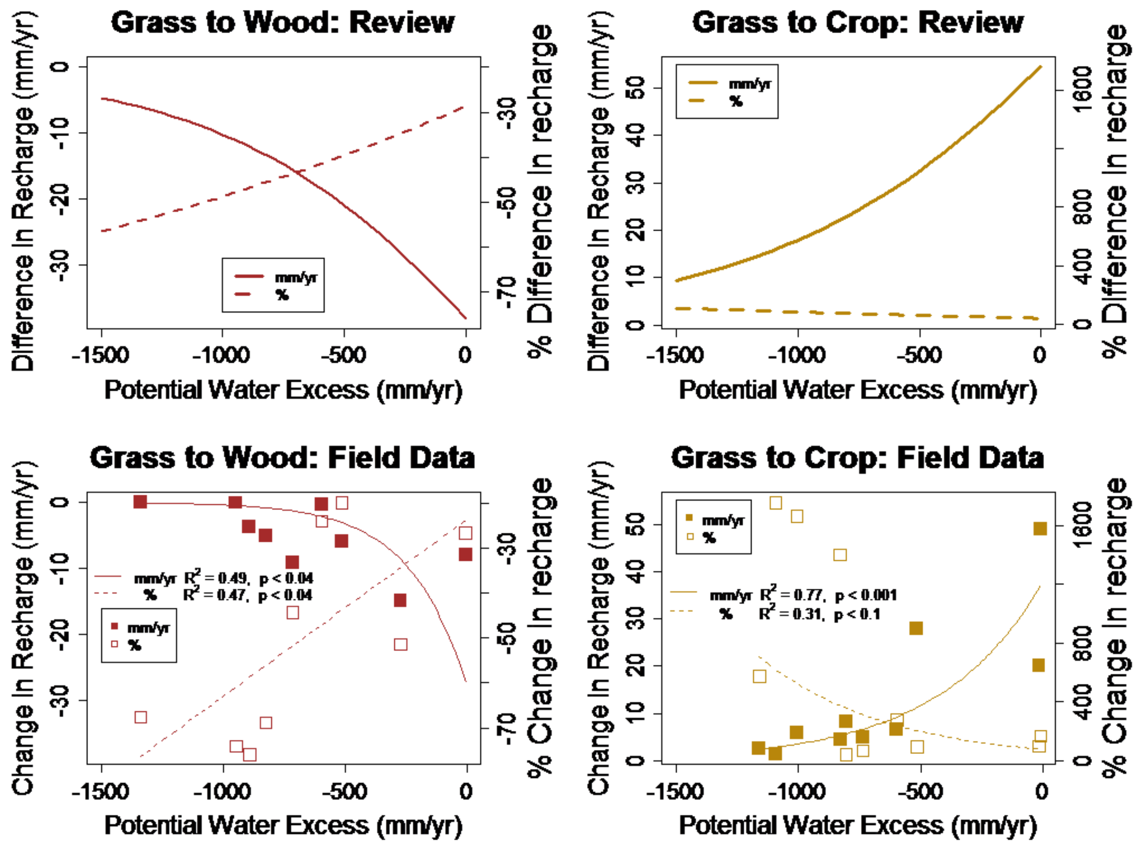

Figure 2.8: Absolute and relative differences and changes in recharge between grassland, cropland, and woody vegetation from synthesis and field data. Solid lines and filled symbols denote absolute differences in recharge, and dashed lines and open symbols denote relative differences. Fitted lines for the field data (bottom panels) were chosen from linear regressions on log- or un-transformed differences in recharge.

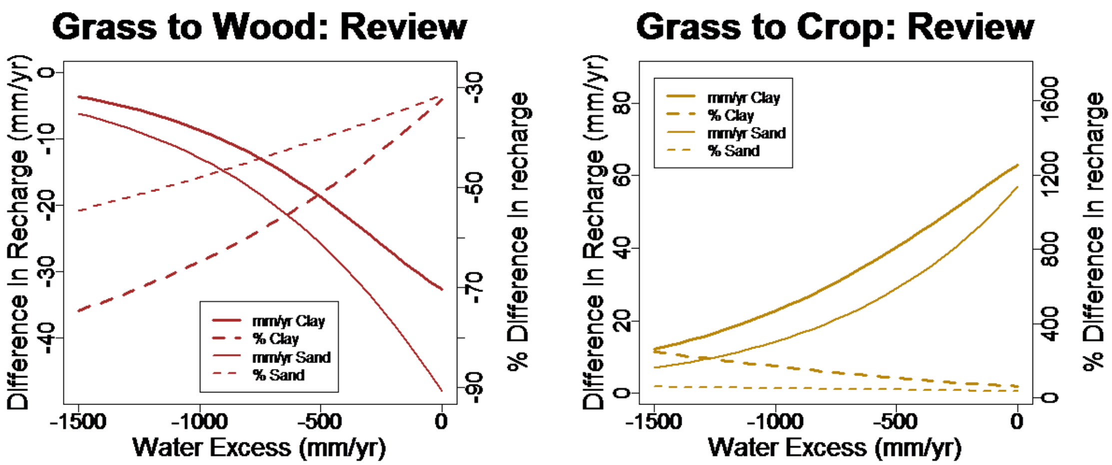

Figure 2.9: Absolute and relative differences and changes in recharge between grassland and the two other vegetation types from synthesis and field data, respectively. Thick lines for the synthesis panels (top) are for clay soils and thin lines are for sandy soils. Solid lines and filled symbols denote absolute differences in recharge, and dashed lines and open symbols denote relative differences. Fitted lines for the field data (bottom panels) were chosen from linear regressions on logged or unlogged dependent variables.

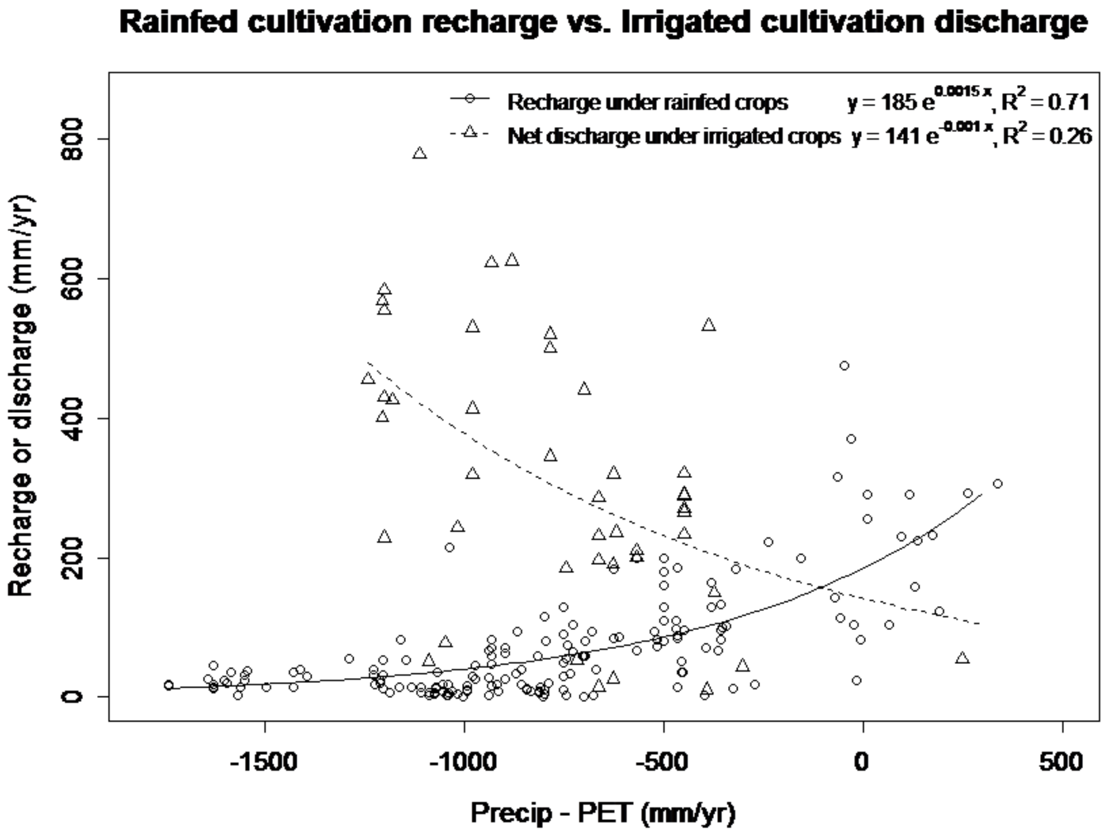

Figure 2.10: Recharge under rain-fed cropland (*Rr*) and net discharge (*Di*=*I* – *R*) under irrigated cropland across water availability (Precipitation – *PET*). Net discharge under irrigated crops is up to two orders of magnitude greater than recharge. The ratio of *Di*:*Rr* represents land area of rain-fed cultivation needed to provide a unit area of irrigated cultivation.

3. Water and carbon tradeoffs with grassland conversions along gradients of precipitation and soil texture

3.1 Introduction

Land-use changes for producing food, fiber, and fuel are the dominant landscape conversions on Earth and drive changes in many ecosystem processes (Vitousek et al. 1997, DeFries et al. 2004, Searchinger et al. 2008). The effects of land-use change on multiple ecosystem services is an emerging topic for long-term human well-being (MEA 2000, Palmer et al. 2004, Foley et al. 2005, Schröter et al. 2005). For example, land-use changes can lead to tradeoffs between the economic benefits provided by the new land use (e.g., onset of agriculture or commercial forestry) and ecosystem services (e.g., carbon sequestration or water provision; Matson et al. 1997, Farley et al. 2005). Furthermore, conditions that promote one ecosystem service may sometimes decrease another, creating a need to balance multiple products and services (Christensen et al. 1996, Jackson et al. 2005, Rodriguez et al. 2006). Our goal was to examine these tradeoffs between two ecosystem services, specifically those associated with carbon and water cycles that accompany land-use changes.

Vegetation affects many different ecosystem services, but it has particularly strong and sometimes opposing influences on carbon storage and water provision. By transforming atmospheric carbon into biomass and simultaneously releasing soil water to the atmosphere, vegetation tightly couples of both cycles in opposing directions between soil and the atmosphere (Cowan 1982, Lambers et al. 1998). More than 90% and 60% of global terrestrial carbon and water fluxes are facilitated by such couple by vegetation respectively (Schimel et al. 1995, Jackson et al. 2001).

For example, soil and plant biomass are two largest and dynamic surface pools of carbon, containing 50% and 20% of non-oceanic carbon, respectively (Schimel et al. 1995). As a major driver of climate change, land-use change releases 15% of current anthropogenic carbon dioxide (CO_2) emissions to the atmosphere (IPCC 2007, LeQuéré et al. 2008, Richter and Houghton 2011). For this reason, carbon sequestration in plants and soil organic matter is an ecosystem service that has stimulated global policy mechanisms for mitigating climate change (McCarl and Schneider 2001, Pacala and Socolow 2004).

Similarly, vegetation is the major factor in recycling precipitation to the atmosphere and limiting recharge to ground water, which contains 99% of liquid fresh water on earth and is a major source of water for one quarter of global population (Ford and Williams 1989, Wickens 1998, Scanlon et al. 2005a). However, groundwater replenishment may be sensitive to land-use changes in arid and semiarid regions, where groundwater recharge is outpaced by extraction for human use (Shiklomanov 1997, Vorosmarty et al. 2004, Farley et al. 2005, Kim and Jackson, In review).

Grasslands in arid and semi-arid regions are convergences of aforementioned issues regarding land-use changes and carbon- and water-related ecosystem services.

Grasslands contain ~40% as much carbon in their soil organic pools as the atmosphere does and those in arid climates and flat landscapes also suffer from lack of surface water, making ground water an essential source for humans over a large part of this biome (White et al. 1995, Jobbágy and Jackson 2000). Grasslands are also subject to widespread replacement with annual crops (agricultural expansion) or woody plants (encroachment or afforestation), which may lead to semi-permanent desertification in arid systems (Archer et al. 1988, Schlesinger et al. 1990, Van Auken 2000, Newman et al. 2006, Ramankutty et al. 2008). Approximately 80% of natural grasslands globally are currently cultivated and an estimated one third of U.S. rangelands are undergoing woody encroachment (Williams et al. 1968, Houghton et al. 2000). Production pressures such as emerging biofuel markets and higher grazing intensities associated with woody encroachment are likely to continue such land-use conversions in the remaining grasslands. Understanding how these vegetation shifts alter water provisioning and carbon sequestration through changes in groundwater recharge and carbon stocks should provide insight into tradeoffs between major ecosystem services with land use.

How then might land-use changes affect carbon sequestration and groundwater recharge in grasslands? Vegetation-mediated carbon and water fluxes in opposing directions across soil and atmosphere may result in tradeoff between carbon sequestration and groundwater recharge. General trends of carbon losses and increased water yields with cultivation of annual crops are respectively well-documented in the literature (Guo and Gifford 2002, Scanlon et al. 2005b), though few studies have examined both concurrently. Effects of woody plant invasion (WPI) are less clear, though changes in ecosystem carbon and water balances with afforestation may serve as proxies (Knight and Thurow 1991, Briggs et al. 2005, Farley et al. 2005, but see Jackson et al. 2002, Knapp et al. 2008, and Eldridge et al. 2011). WPI generally increases overall carbon storage with greater productivity and plant biomass, but greater transpiration from woody plants may reduce recharge, though there is debate on generality of these trends (Huxman et al. 2005, Wilcox and Huang 2010). Because both climate and soil are important determinants of fluxes and storages of carbon and water (Horton 1933, Thornthwaite et al. 1957, Lvovitch 1970, Schimel et al. 1994, Schimel 1995, Houghton 2000), a predictive understanding of ecosystem service tradeoffs is likely to depend on the interaction of biotic and abiotic factors.

The potential for tradeoffs between carbon storage and water supply makes the economic valuation of the services important for establishing their net benefits with land-use change. Quantifying and valuing such changes are first steps towards incentivizing provision of the services through policy or market mechanisms. Resolving optimum land uses for multiple ecosystem services would be challenging without such joint valuation. Unfortunately, these services are traditionally undervalued, if at all, which has led to degradation or unsustainable use of the services (MEA 2000, NRC 2005). The two ecosystem services considered here not only offer potential to mitigate two serious and worsening global environmental problems, climate change and freshwater scarcity (Vorosmarty et al. 2000, IPCC 2007, UNEP 2007), but are already recognized in their values and marketability (Wilson and Carpenter 1999, Laurance 2007, Turpie et al. 2008). A holistic approach combining biogeochemical, hydrological, and social aspects of ecosystem services would contribute greatly towards sustainable management of these environmental externalities.

The goal of our research was to examine how magnitudes and values of carbon sequestration and groundwater recharge vary with land-use changes in grasslands, focusing on the potential tradeoffs between the two. We compared carbon storage and groundwater recharge in paired grassland, woody plant invaded, rain-fed and irrigated cultivation plots to examine tradeoffs between changes in the two ecosystem services arising from land-use changes. We also examined the changes in the two ecosystem services across regional climate and soil texture gradients, allowing us to test relationships between the shifts in ecosystem services and abiotic factors. We combined our results with literature values of the same ecosystem services to estimate the net economic value of the land-use changes, presenting a direct comparison between changes to carbon storage and water provisioning.

3.2 Methods

3.2.1 Site description

In Argentina, we located five sites on flat landscapes across a precipitation gradient (382 to 1215 mm/yr) in the Pampas grasslands, and the Parera site in the Dry Forest ecotone (*Prosopis caldenia* woodlands). When available, rain-fed cultivation (cropland) and woody plant invasion (woodland) plots were paired to an adjacent or nearby natural grassland plot within 1 km at each site (Table 3.1,3.2).

In the southern Great Plains of the United States, we selected five sites along a precipitation gradient that ranged from 407 890 mm/yr. Land uses selected as paired plots here were natural grasslands, rain-fed croplands, and irrigated croplands. Although the grassland plots were largely covered by herbaceous plants, there was slight woody encroachment (*Prosopis glandulosa*) at some of the sites, with average stem density of 100-200/ha.

In addition to our new field data, we also analyzed soil cores from 5 paired grassland and woodland sites located across a precipitation gradient in the southwestern Great Plains. Detailed information and methods for these sites are available in Jackson et al (2002) and McCulley et al (2004).

Most plots have 30+ years of constant land-use history. Land-owners or farm managers were surveyed for land-use history at each site, including cropping regimes (e.g., species, rotations) and fertilizer/pesticide/irrigation inputs. Tree stand ages were verified with aerial photos or tree ring cores taken during our sampling campaign from 2008 to 2010. Precipitation data were obtained from long-term (30+ years) records maintained by weather stations near the sites within 30 km or maintained onsite by the farm managers (INTA , NOAA). Most of our sites in the southern Great Plains were located in agricultural research extension centers with good local records of the weather and land management history (Table 3.1).

3.2.2 Soil sampling

At the Argentinean sites, soil samples from three to eight deep boreholes (to nine meters or to ground water table) and up to six 30-cm shallow cores were taken at each land-use plot using a manual auger and a soil corer with a slide hammer attachment. Augered samples were taken every 20 cm to 1 m, then every 30 cm to 4 m depth, then every 50 cm afterwards. Soil samples were homogenized and subsampled during collection in the field, then frozen until analysis. At every meter, depth increments, auger diameter and oven-dried weights of the augered soil samples were recorded for bulk density calculations. In our U.S. sites, we used a direct-push coring rig (Geoprobe Systems, Salina, KS) for five to eight cores per plot up to 8.5 m depth. The only plot soil samples were not retrieved to 8.5 m depth was at a grassland plot near San Angelo, TX, due to indurated material found around 5 m depth. Soil cores were weighed in the field, subsampled separately for both soil moisture/bulk density using intact cores and elemental analysis using homogenized soil samples and shipped to the lab for analysis.

3.2.3 Aboveground biomass

Aboveground herbaceous biomass, including litter, was harvested in three to eight randomly placed 0.5 m x 0.5 m quadrats or 0.3 m diameter rings in every grassland and woodland plot. In addition, basal diameter or diameter at breast height and height of individual woody stems were measured in up to three 20 m x 20 m quadrats in every woodland plot. We used species-specific allometric equations and these measurements to estimate tree volume and biomass (Table 3.2). Crop biomass and grasses that were regularly mowed were not considered as carbon storage due to their quick turnover.

3.2.4 Lab analysis

Soil and biomass samples were dried in the oven for gravimetric moisture content calculations and ground for organic carbon measurement on Carlo Erba Elemental Analyzer using the two temperature combustion method (Chichester and Chaison 1992). Oven-dried and homogenized soil samples were shaken with equal weight of double deionized water for four hours. The mixture was centrifuged, the supernatant filtered, and the filtrate analyzed for anions (Cl^-, Br^-, NO_3^-, SO_4^{2-}, and PO_4^{3-}) by ion chromatography (Dionex ICS-2000). Cl^- concentrations in the soil porewater was calculated by dividing the soil Cl^- contents (mg Cl/kg soil) by gravimetric soil moisture. Soil texture was determined in the laboratory by the pipette method (adapted from Klute 1986) and ranged from sandy to clay (Table 3.1). Bulk density was estimated from weight and volume of the dry soil sample. We estimated soil organic carbon storage by multiplying soil organic carbon contents with bulk density estimates by depth down to 2 m depth to standardize our comparison across all the sites.

3.2.5 Groundwater recharge calculations

Recharge rates at our sites were estimated by chloride mass balance and tracer displacement method from unsaturated zone soil samples (Allison and Hughes 1983, Walker et al. 1991, Phillips 1994). More details on methods are outlined in Chapter 2.

3.2.6 Statistical analysis

We used Mann-Whitney U-test to compare the recharge rates and soil organic carbon stocks derived from individual cores between the different land uses. We also made paired comparison of carbon stocks (SOC, biomass, and total) and average groundwater recharge rates between the different land uses with paired T-test or Wilcoxon signed rank test depending on whether the data satisfied assumptions of normality. The tests were performed in JMP® (SAS Institute). We also analyzed, by regression, absolute and relative changes in recharge and carbon stocks (soil, biomass, total) due to land-use changes of natural grassland across precipitation and clay contents of the surfacial 1 m of soil. Relative changes from grassland ecosystem services were calculated as:

$$\frac{ES_{new} - ES_{grass}}{ES_{grass}} \qquad \textit{Eq. 3.1}$$

where *ES* is either carbon storage or recharge, and subscripts *new* and *grass* denote new land use and grassland, respectively. We used regression analysis to test for relation between carbon storage and groundwater recharge under each land use, as well as between changes in the two services following the land-use changes of grasslands to test for tradeoffs between the two ecosystem services. For some of our regression analyses, we grouped our sites into two soil textural classes, based on sand and clay contents in the 1m of soil to illustrate more easily the relationship between land use and abiotic factors (i.e., Figure 3.3,3.6,3.5,3.8,3.9). Sand, loamy sand, sandy loam, and loam were grouped as “sands”, and silt loam, clay loam, sandy clay loam, silt and clay were grouped as “clays”. Regressions were performed in R (R Foundation for Statistical Computing ; Vienna, Austria).

3.2.7 Economic analysis

If land-use changes result in tradeoffs between ecosystem services, whether the land-use changes represent a net cost or benefit would depend on the relative values of the services. Using benefit transfer methods, where economic values for ground water are inferred by transferring reported values of the resource in comparable market contexts and locations (Rosenberger and Loomis 2003, Murray et al. 2009), we estimated the total value of changes in carbon sequestration and groundwater recharge by multiplying our biophysical measurements of the services with shadow prices of the services acquired from the literature for our U.S. study region. We used grassland as the baseline activity in our analysis, capturing costs or benefits of land-use changes such as woody encroachment and cultivation. For our carbon prices, we used a value of carbon sequestration based on social costs of CO_2 emissions (IPCC 2007, USEPA 2008), about 2010 US$15, that was assumed to be similar across our sites.

For water valuations, rather than equating the marginal cost of water extraction to be its value, we assume that groundwater recharge has a value by preventing or slowing the drawdown of the water table from extractive uses of ground water, which has led to widespread and unsustainable drawdown of the water table and increase in pumping costs in more arid parts of the study region (Nieswiadomy 1985, Galloway et al. 1999, Dodds et al. 2004). For these reasons, we assume that hypothetical water markets in this analysis would seek to reverse or steady the decreasing water levels by accounting for the social cost of water, which would encourage recharge and discourage withdrawal. Water values at locations near the research sites were obtained from studies of market values of water transfers (Brookshire et al. 2004), actual costs of water provision (Olenick et al. 2004), production values of irrigation water (Ruesink 1979, Gilson et al. 2001, Johnson et al. 2007), and farm sales (Gilliland et al. 2004). Local water values, reported in 2010 $US, were negatively correlated with potential water excess, a climatic proxy for water yield, and were assumed to be specific to the sites (Figure 3.1). Market values of water transfers between private and municipal parties in the mid Rio Grande valley were used as water values for our two New Mexico sites. Studies that examined production values of irrigation water derived the water values based on the differences in net farm incomes of lands that were rain-fed and irrigated but otherwise comparable, taking into account the differences between the inputs and outputs of the two cultivation types (KS, OK, and TX sites). For the farm sales method, the average difference in farm sales prices of croplands with and without water rights were equated to the difference in present values of the future earnings (projected for 30 years) from the lands with and without irrigation, which gives the value of irrigation water per unit basis when divided by average irrigation applied (Torell et al. 1990). The cost of additional water provision by removing woody encroached vegetation near our San Angelo site was comparable in value from that of farm sales method (2010 US$118 vs. 105), though these methods represent different metrics, with the former examining the cost of increasing the supply and the latter examining the perceived value of the water. We assume that these values reflect the marginal value of water that one might be expected to derive from water trading or from irrigated crop production given the recent market and hydrogeological conditions.

For example, land-use change that increases recharge relative to the grassland conditions would result in additional water for extraction and use without drawdown of the water table, while one that decreases recharge would indicate that less water is available for extraction compared to grassland conditions, conferring a cost. In the case of irrigated cultivation, extracted water that is not returned to the water table by recharge represents drawdown of the water table and an opportunity cost for other users relative to the grassland conditions. In our San Angelo site, where ownership of water rights is necessary for irrigation, irrigation is not an externality like it is at other irrigated sites. In our analysis, we quantify the social values of changes in groundwater recharge and carbon storage regardless of whether they are externalities or not. Although the values we use do not reflect non-extractive values such as providing baseflow to streams and the associated ecological benefits or preventing land subsidence from lowering of the water table, maintaining sustainable water table and groundwater use is an explicit goal in this analysis, where addition or loss of recharge from grassland conditions due to land-use changes confers social benefit or cost for use of that resource.

As a part of our valuation efforts, we simulated changes in carbon storage and groundwater recharge for the 30 years following grassland conversions using sigmoidal curves of the following form:

$$ES(t) = \frac{A}{1 + e^{(B-t)\cdot C/30}} + D \qquad \textit{Eq. 3.2}$$

where $ES(t)$ is carbon stock or groundwater recharge at time t since the land-use change in years (Mg/ha or mm/yr), with t=0 indicating original native grassland conditions, A is the difference in carbon stock or groundwater recharge rate between grassland and the new land use (Mg/ha or mm/yr), both taken to be at or near equilibrium, B is the year at which half of the difference between the new land use and the original grassland value has been reached, C is the number years it takes to approach the new equilibrium (rate of change), and D is the grassland carbon stock or groundwater recharge (Mg/ha or mm/yr). Flexibility of the sigmoidal curve allowed us to represent diverse temporal dynamics (Figure 3.2). To represent realistic scenarios of the changes in carbon storage, a subset of the curves were based on studies examining chronosequences of soil and biomass carbon storage following vegetation shifts (Turner et al. 2005, Davidson et al. 1993, Dalal et al. 1986). Changes in carbon storage after land-use changes are typically rapid initially, usually peaking within the first 20 years, whereas changes in recharge are buffered by a thick vadose zone, operating at longer time scales often necessary for the new recharge to reach the groundwater table. The time for the new recharge to reach the water table was calculated as:

$$T = \frac{S_{new}}{R_{new}} \qquad \textit{Eq. 3.3}$$

where T is time for recharge under the new land use to reach the water table (yrs), S is integrated soil moisture storage in the unsaturated soil zone between the bottom of the root zone and above the water table (mm), *new* is a subscript indicating new land use, and R_{new} indicates recharge under the new land use (mm/yr). We used USGS well records from near our sites for our estimates of the water table depths. Modeling of soil carbon stocks and groundwater recharge rates using the Rothamsted carbon model and simulation of heat, water, and solutes in variably-saturated media using HYDRUS 1-D, respectively at some of our sites confirmed these patterns (Coleman et al. 1997, Simunek et al. 2008).

Simulated changes for 30 years following the land-use changes were multiplied by the shadow prices discounted at 4% annually (adjusted for inflation), yielding net present values of the land-use changes (Jenkins et al. 2010). Because carbon stocks were used to infer fluxes of carbon whereas groundwater recharge itself is a flux, changes in carbon stocks between two consecutive years, $ES(t)$-$ES(t$-1), were used as carbon sequestration for year t attributable to land-use changes, whereas differences in recharge between years t and grassland recharge, $ES(t)$-$ES(0)$, were used as changes in recharge due to land-use changes. We compared the net values of changes in carbon storage and groundwater recharge following cultivation and WPI, with our specific aims being to determine whether these externalities represent net costs or benefits to the local human communities, which ecosystem service is likely to drive land-use decisions, and under which environmental conditions markets of ecosystem services would encourage grassland conversions to other land uses. We focused our analysis for sites in the southwestern U.S., due to relative dearth of economic valuations of water in our study region in Argentina, but we expand on our results to discuss policy implications for both regions.

3.3 Results

Changes in grassland carbon storages accompanying cultivation and woody encroachment were negatively correlated with changes in groundwater recharge, indicating a tradeoff between the two ecosystem services with grassland conversions

(Figure 3.3, $P<0.004$). Rain-fed croplands had 34 Mg/ha lower total carbon storage on average than grasslands had, driven largely by losses in cropland soil organic carbon (Figure 3.4, Table 3.3, paired T-test; t-ratio=-4.74, $P<0.0011$). However, croplands also had 13 mm/yr more groundwater recharge, a six-fold increase on average compared to native grasslands (Figure 3.5, Table 3.3, signed rank test; $P<0.001$). Woody encroached sites had 23 Mg/ha more total carbon storage than native grasslands (paired T-test; t-ratio=1.87, $P<0.0469$) but only half of the groundwater recharge, 5.3 mm/yr less on average (signed rank test; $P<0.002$). The opposing directions of changes in ecosystem carbon storage and groundwater recharge indicate a clear tradeoff between the two with land-use changes, as well as strong effects of vegetation on the two ecosystem services. In contrast to rain-fed crops and woody encroached sites, however, irrigated croplands tended have lower carbon storage *and* groundwater recharge—by 14 Mg/ha and 330 mm/yr on average respectively—the latter attributable to net discharge of ground water from irrigation use (Table 3.3).

Changes in the ecosystem services were also associated with precipitation and clay content in soils, indicating these abiotic factors can help predict effects of grassland conversions on the services. Losses in total organic carbon due to rain-fed cultivation became larger at sites with higher precipitation and clay content (Figure 3.6,3.7). Total organic carbon gains with woody encroachment decreased at sites with higher clay content and eventually became losses in a few sites with the highest clay contents

(R^2=0.47, P<0.028). Although differences in total carbon storage between woody-encroached grassland pairs showed overall no trend with precipitation (P<0.84, data not shown), this lack of response resulted from the opposing trends of greater soil organic carbon losses and greater biomass gains with more precipitation (Figure 3.6,3.8; SOC, R^2=0.64, P<0.0054; biomass, R^2=0.40, P<0.048).

Absolute changes in groundwater recharge also increased with potential water excess but tended to be smaller in clayey soils compared to sandy soils at both woody encroached and cropland sites (Figure 3.5; WPI, R^2=0.42, P<0.057, Cultivation, R^2=0.78, P<0.001), revealing an interaction of effects of land use and physical variables on recharge. Interestingly, relative recharge changes showed the opposite association with these environmental variables, with stronger effects of land-use change on recharge at sites with more arid climates and more clay content (Figure 3.5; WPI, R^2=0.57, P<0.018, Cultivation, R^2=0.32, P<0.087), suggesting that relative effects of land-use changes on groundwater recharge are likely to be greater in drier climates and finer-textured soils.

These relationships suggest that abiotic factors can highlight the relative importance of carbon storage and water provision following land-use changes. For example, grassland conversions in humid climates resulted in large changes in both carbon and water compared to those in drier climates. Sandy sites tended to have larger changes in water whereas clay sites tended to have larger changes in carbon, usually losses, indicating that the large losses of carbon may outweigh the small changes in recharge in clayey soils (Figure 3.3,3.9).

Grassland conversions at most sites in southwestern U.S. resulted in negative net changes in terms of ecosystem carbon storage and water provision, indicating that these land-use changes represent costs (Table 3.4). Land uses in the order of least to most negative net values were woody encroachment, rain-fed cultivation, and irrigated cultivation. Moreover, aside from irrigated cultivation, net present values of changes in carbon storage tended to overshadow those of groundwater recharge regardless of the carbon prices used. The average factor by which carbon benefits/costs overwhelmed recharge benefits/costs varied between two- to 100-fold (on average by $500/ha) depending on the land use and carbon price (Table 3.4).

For example, values of carbon and water changes were most comparable in magnitude for woody encroachment, reflecting relatively small changes in both carbon and water associated with the two land use (Table 3.4). For rain-fed cultivation, we observed clearer dominance of carbon over water but these changes represented smallest costs in drier grasslands, reflecting the higher value of the additional recharge and less carbon lost compared to humid grasslands. Compared to rain-fed cultivation, values of carbon changes in water and carbon were similar in magnitude for irrigated cultivation due to the large losses in recharge associated with irrigation (Table 3.4).

Our analysis was also robust despite diverse temporal dynamics simulated for changes in carbon and water. Whether these changes represented net costs or benefits in terms of carbon and water were consistent regardless of the scenarios were same except for some woody encroached plots, indicating that our conclusion of carbon's dominance of water is applicable for wide range of hydrological and biological conditions given the values of carbon and water explored in our analysis (Table 3.4).

3.4 Discussion

A tradeoff between carbon sequestration and groundwater recharge with land-use changes was evident in our data and indicates that economic or policy incentives that encourage one ecosystem service will likely influence the other. For example, woody plant invasion (WPI) of grasslands may qualify as carbon sequestration, but accompanying loss of groundwater recharge raises concern for local communities that may depend on that resource. On the other hand, cultivation results in emissions of CO_2, which represents an environmental cost, but this cost may be offset by enhanced groundwater recharge in areas where water is highly valued. In our analysis, irrigated cultivation tended to decrease both carbon storage and groundwater recharge, indicating that this land use may result in negative environmental externalities. Ultimately, the question of whether these changes actually represent costs or benefits is a multi-faceted problem, which we address in this section.

Knowledge of how ecosystem services change along environmental gradients can be used to help identify regions where certain land-use changes are likely increase or decrease ecosystem services. For example, the larger losses of carbon but smaller changes in recharge associated with more clayey soils that we observed suggest that clay soils are more likely to result in losses in the net value of the two services than sandy soils are (Figure 3.3,3.9). However, negative values associated with recharge, such as its potential to contribute to flooding under certain geologic and climatic conditions, or solute leaching and contamination of shallow aquifers with salts, reveal an interesting dichotomy between carbon sequestration and groundwater recharge as ecosystem services. Uncertainties regarding benefits of groundwater recharge, such as the risk of flooding or aquifer contamination, create asymmetry in the tradeoff of carbon and water, where land uses that gain carbon but lose water (e.g., WPI) may represent a win-win situation under certain conditions, and land uses that do the opposite (e.g., rain-fed cultivation) represent a lose-lose situation in other conditions. Hence, net value of the two ecosystem services would depend on how the magnitude and economic value of the ecosystem services change across the environmental gradients, especially given that groundwater recharge is localized and its value is context-dependent. We discuss the cases for our two study regions individually below.

For the Argentinean case, urban and rural development and productivity are limited by water in drier grasslands, but in the humid pampas, relatively abundant rainfall, shallow groundwater tables, and lack of surface drainage features owing to young geologic history can create flooding risks (Aragón et al. 2010). The most positive combination of changes in the ecosystem services thus may be expected for 1) WPI in humid grasslands with sandy soils where carbon gains are the largest but where large water losses represent small costs or even benefits through flood prevention, and 2) for rain-fed cultivation in drier grasslands where carbon losses are smallest and the gain in groundwater recharge is expected to be more valuable. The mixed agro-pastoral system of Argentina, however, is experiencing opposite trends under market pressures to cultivate crops in the productive humid pampas, increasing risks for flooding there and driving livestock production to the marginal drier grasslands and increasing risks of degradation and woody plant dominance in the drier grasslands (Schlesinger et al. 1990, Archer et al. 1994). Irrigated agriculture is also on the rise in drier areas there, presenting risks for loss of both ecosystem water and carbon. Social and policy mechanisms aimed at incorporation of these environmental externalities into markets may help reverse or slow these trends.

In the southwestern U.S., water values are similarly driven by physical scarcity at individual sites (Figure 3.1), but there is an additional consideration of salts stored in grassland soils that accumulate as a result of many years of atmospheric deposition and low recharge rates. Theses salts are vulnerable to dissolution and leaching with higher recharge rates, which may lead to their increased concentrations and deterioration of water quality, especially for residential uses of water. Unlike more recently deposited sediments in the Argentinean Pampas, soils in the Great Plains store considerably greater amounts of salts (representing 7000-14000 years of accumulation), also in part due to the higher clay content that impedes soil water fluxes. If an influx of solutes from leaching limits the current or future uses of the ground water, the degradation of water quality may negate the value of the additional recharge. Such costs may be exaggerated in drier climates and clay soils that store large amounts of salt from intrinsically low soil water fluxes. In addition, clay soils tend to release larger amounts of carbon to the atmosphere with cultivation, indicating that the land-use change is more likely to result in costs in terms of carbon and water in clay soils.

Because many grasslands have already been converted to croplands around the world, we note that restoration to grasslands may represent opportunities for positive changes in terms of *both* carbon sequestration and prevention of water quality degradation for recently converted grasslands. Transition from grassland to woodlands in clay soils also tended to result in losses of both carbon and water services, indicating that woody encroachment in clay soils also is likely to lead to losses of both carbon and water from the grasslands.

Because of the simplified definition of groundwater recharge as an ecosystem service that we used, and the limited number of water valuation studies, we applied economic analysis to only a subset of our sites. Irrigated agriculture was the dominant user of ground water in the U.S. study region where the potential degradation in water quality is not an imminent threat for such uses, unlike for residential consumption. Moreover, unconfined shallow aquifers such as the ones we examined are generally not used for residential purposes due to naturally unsuitable groundwater quality or risks associated with such water sources (NRC 1997). However, we emphasize that cultivation of natural grasslands as a means to enhance water provision would be inappropriate due to associated risks of water quality degradation in areas where shallow unconfined aquifers are sources for residential uses. The hydrogeological and social settings that most closely match our interpretation of value of groundwater recharge are regions with negligible runoff and risk of flooding and for sites that have undergone agricultural conversion, areas with low soil salt storage or where use of shallow ground water is solely by a sector unaffected by potential changes in water quality. Of our U.S. sites, Jornada, Sevilleta, San Angelo, Quanah, and Vernon most completely satisfy these assumptions. Though we present economic value of the changes in the two ecosystem services with land-use changes at all U.S. sites, our valuation should be treated more as case studies for the aforementioned sites specifically.

Our results from valuation of the changes in the carbon and water ecosystem services point to carbon sequestration being the dominant ecosystem service in terms of its economic value (Table 3.4). This was true across a range of carbon prices and diverse temporal dynamics of carbon and water changes, indicating that a carbon market or tax could be a more important driver of land-use decisions than water markets are. Water prices or water yield changes necessary to offset value of carbon storage changes would be up to 300 times greater than those found in our study, indicating water would be unlikely to overshadow carbon without significant departures in hydrology or valuation of ground water in the region (Table 3.4). Though our analysis incorporated relatively low water values for agricultural, as opposed to residential, uses, transfers of water rights from farmers to large municipalities through water markets are unlikely to be the norm in much of our study region; water trades are most common when such entities share access to the resource and have the means for such trades to occur (Griffin 2006). Emergence of carbon market thus may have large consequences on land-use decisions compared to groundwater resources as they are currently valued.

Predominantly negative net values of the land-use changes in terms of carbon and water further indicate that market incorporation of the ecosystem services will likely reduce the extent of these land-use changes. For example, large net costs of changes that irrigated cultivation incurs on carbon and water at our cotton-producing sites would be 30-40% of profits associated with the land use, $700/ha/yr on average and net present value of $12,000 discounted for 30 years for irrigated cotton in the region (Robinson and McCorkle 2006). Rain-fed cultivation also resulted in overall costs which increased with higher rainfall, paralleling losses of carbon and suggesting higher likelihood of agricultural conversions of drier grasslands (Table 4). The marginally positive or negative net values associated with WPI at our southwestern U.S. sites may not be significant enough to warrant actions leading to large-scale changes by land-owners. This would likely result in overall more careful grassland conversions.

However, carbon dominance may also promote depletion or degradation of ground water. Establishment of carbon markets could lead to depletion of ground water by encouraging WPI, leading to long-term consequences for productivity and sustainability. For cultivation, though arid grasslands might be preferred over more humid grasslands with carbon markets, large storage of salts under grasslands in arid climates represent a risk for salinization of ground water through enhanced leaching. Our analysis was decidedly short-term and was preformed to simulate the likely policy or market incentives that would affect land-use decisions. Such short-term outlook does not explicitly address long-term sustainability of ground water and may lead to exploitation or degradation of ground water. A holistic accounting of intrinsic values of ground water may weaken the dominance of carbon over water.

Several economic and biophysical factors may have contributed to the dominance of carbon in our cost-benefit analysis. One consideration is that carbon sequestration may be initially rapid following land-use changes in aggrading woodlands or as soil organic matter in croplands decomposes after tillage, but changes in groundwater recharge may take decades to propagate through the thick vadose zone. The discounted nature of future values thus gives more weight to short term changes and favors the faster changes in ecosystem carbon against the slower changes in ground water. However, even under alternative scenarios that assumed more rapid response of water yield to land-use changes, as might be the case for changes in surface runoff, changes in recharge could not offset changes in carbon storage at most of our sites (Table 3.4). Similarly, lower or no discount rates, if we deem future provision of these ecosystem services as valuable as current provision of the services, could make values of the carbon and water more comparable in magnitude but still would result in higher carbon values than those of water (data not shown).

Another potential contributing factor to the carbon-dominance in our result is that surface runoff may represent a significant portion of the water yield. Such processes are minimized in arid and flat landscapes with sandy soils, but appreciable runoff is more likely in the humid rolling hills of Central Texas. Though shallow ground water is not widely used in these areas explicitly, it supports the baseflow of streams and rivers that are used for residential and industrial purposes downstream. However, hydrological changes up to 300 times larger would be necessary to offset changes in carbon storage economically in such landscapes (Table 3.4), equivalent to converting three times the amount of mean annual precipitation received at such sites into surface runoff. Long-term monitoring of a grassland watershed in humid climate with clayey soils in Riesel, Texas yielded estimates of a relatively small amount of runoff (137 mm/yr; Harmel et al. 2006), indicating that our conclusions of generally negative net values of the land-use changes and the carbon dominance over that of water in terms of their values are unlikely to be reversed with incorporation of surface runoff as water provision. Moreover, surface runoff is likely to be affected by land-use changes in similar fashion as groundwater recharge was. Woody invasion may reduce runoff from rainfall interception by leaves and stems and improved infiltration, or may not change runoff compared to grasslands (Huxman et al. 2005, Wilcox et al. 2005). Cultivated catchments tend to have similar or greater runoff than grassed catchments, partly from loss of vegetation and soil structure as well as soil compaction due to heavy machinery used for harvesting that allows smaller precipitation events to overwhelm infiltration capacity associated with cultivation (Smith et al. 1983, Van der Kamp et al. 1999). The fact that runoff may be unaffected or mirror changes in recharge with land-use changes further indicates that inclusion of runoff in our analysis is unlikely to alter our other main conclusion of climatic and soil texture zones that are likely to result in carbon and water changes that are beneficial with land-use changes. For example, reduction of recharge and runoff that are likely results of woody invasion would point to this land-use change as a win-win case for carbon and water, given the negative values associated with recharge and runoff due to risk of flooding in the humid Argentinean pampas.

Observed differences in carbon storage and groundwater recharge between the land uses are attributable not only to the tradeoff between transpiration and photosynthesis at the leaf physiology level, but also to management practices and functional traits associated with the land uses. For example, tillage encourages microbial decomposition of the built-up grassland soil organic carbon. Woody plants on the other hand allocate more primary production to longer-lived structural tissues compared to grasses, resulting in a build-up of carbon in the biomass (Table 3.3). Such differences in land management practices and plant traits directly affect the built-up carbon pools following land-use changes. Similarly, management practices such as tillage and fallow periods may increase groundwater recharge through higher hydraulic conductivity and more frequent opportunities for recharge respectively, while greater rooting depth of woody plants may enable more complete exploitation of soil water in order to supply the greater leaf area and transpiration compared to grasses (O'Connell et al. 1990, Daniel 1999, Seyfried and Wilcox 2006, Scanlon et al. 2008). Such vegetation traits and management practices were not a focus for this work but merit consideration for land-use decisions that seek to optimize multiple ecosystem services.

3.5 Conclusion

In our comprehensive and direct comparison of carbon sequestration and groundwater recharge, grasslands undergoing land-use changes were gaining one ecosystem service at the expense of the other, pointing to coupled tradeoffs arising from physiological and management factors. Changes in the two services varied with precipitation and soil texture, and we present broad recommendations based on these relationships and uncertainties regarding the value of groundwater recharge. In contrast to the value of carbon sequestration, the value of groundwater recharge is localized and context-dependent, and may be negative under certain conditions due to risks of flooding or degradation of water quality by leaching salts. Such uncertainties regarding the value of groundwater recharge suggest that woody encroachment may have positive values in humid regions with sandy soils, where increases in carbon storage may be large and decrease in recharge may prevent flooding. Similarly, crop cultivation in grasslands that have accumulated salts for a long period of time, especially in drier climates and with clay soils, may lead to emission of carbon to the atmosphere while deteriorating water quality with increased leaching of the solutes. Our economic analysis assuming market incorporation of these services showed that most land-use changes we examined would result in net costs in our U.S. study region, which may encourage more selective conversion of grasslands. However, due to dominance of carbon over water and because water values used in our study did not account for non-extractive value of ground water, market incorporation of the services also has potential to result in exploitation or degradation of groundwater resources, especially in more arid conditions, given the more positive net value of grassland conversions in drier climates. Efforts to manage and optimize ecosystem services could guide us to land-use configurations to maximize benefits, currently existing as externalities to both social and policy endeavors.

Table 3.1: Site information.

Site	Lat., long. (degrees)	Precip (mm/yr)	Soil	Vegetation type	Years since change
Nahuel Mapa	-34.8, -66.2	382	Fine sand	Grass, Wood	80
Caldenadas	-33.8, -65.8	506	Fine sand	Grass, Wood, Rain-fed	60, 10
Dixonville	-34.7, -65.5	525	Fine sand	Grass, Wood, Rain-fed	60, 15
Parera	-35.1, -64.5	682	Loam	Grass, Wood, Rain-fed	100+, 80
San Claudio	-35.9, -61.2	1011	Sandy loam	Grass, Rain-fed	40
San Antonio	-34.2, -59.4	1219	Loam	Grass, Wood, Rain-fed	40, 60
Sevilleta	34.3, -106.7	277	Turney loam, sandy loams	Grass, Wood	50
Goodwell	36.6, -101.6	407	Gruver clay loam	Grass, Rain-fed, Irrigated	60, 60
Tribune	38.5, -101.6	479	Richfield silt loam	Grass, Rain-fed, Irrigated	30, 50
San Angelo	31.4, -101.3	514	Angelo clay loam	Grass, Rain-fed, Irrigated	100, 40
Quanah	34.3, -99.8	679	Sagerton clay loam	Grass, Rain-fed, Irrigated	100, 60
Vernon	33.9, -99.4	660	Tillman clay loam	Grass, Wood	40
Riesel	31.5, -96.9	890	Heiden Clay	Grass, Wood, Rain-fed	100+, 100+
Engeling	31.9, -95.9	1070	Loamy fine sand, fine sandy loam	Grass, Wood	50

Table 3.2: Dominant species at each site and land use and allometric equations used to estimate woody biomass.

Site	Dominant species			Allometric equation‡	Reference
	Grasses	Crops†	Woody plants		
Nahuel Mapa	*Stipa spp.*	--	*Geoffroea decorticans*	B = -7.6 + 0.35* ab	Molinero et al. 1986
Caldenadas	*Stipa spp.*	C	*G. decorticans*	B = -7.6 + 0.35* ab	Molinero et al. 1986
Dixonville	*Stipa spp.*	C, S	*G. decorticans*	B = -7.6 + 0.35* ab	Molinero et al. 1986
Parera	*Stipa spp.*, *Bromus brevis*	C, S, W, Su	*Prosopis caldenia*	*log*(B)=-0.43+ 2.18**log*(10*bd)	Hierro et al. 2000
San Claudio	*Paspalum quadrifarium*, *Cortadeira selloana*	C, S, W	--		
San Antonio	*Bothriochloa lagureoide*, *Deyeuxia viridi-flavescens*, *Briza subaristata*	C, S, W	*Gleditsia tricanthos*	*log*(V)=0.00051+ 2.22**log*(dbh)	McHale et al. 2009
Jornada	*Bouteloua eriopoda*	--	*Prosopis glandulosa*	Biomass harvest	Jackson et al. 2002
Sevilleta	*Bouteloua eriopoda*	--	*Larrea tridentata*	Biomass harvest	Jackson et al. 2002
Goodwell		C, So	--		
Tribune		C, W, So	--		
San Angelo		Ct	--		
Quanah		Ct	--		Ansley et al. 2010
Vernon	*Stipa spp.*	--	*P. glandulosa*	Biomass harvest	Jackson et al. 2002
Riesel	*Schizachyrium scoparium*	C	*P. glandulosa*	Biomass harvest	Jackson et al. 2002
Engeling	*Andropogon spp.*	--	*Juniperus virginiana*	Biomass harvest	Jackson et al. 2002

†C-corn, S-soybean, W-wheat, Su-sunflower, So-sorghum, Ct-cotton
‡B-wood biomass, V-wood volume, dbh-diameter at breast height, db-basal diameter, ab-basal area

Table 3.3: Biomass, soil and total carbon stocks at each site and land use.

Site	Biomass Carbon (Mg/ha)			Soil Carbon (Mg/ha)			Recharge (mm/yr)		
	Grass	Crop	Wood	Grass	Crop	Wood	Grass	Crop	Wood
Nahuel Mapa	9.1	—	36	48	—	72	4.8	—	1.16
Caldenadas	9.5	—	412	52	51	64	7.4	12.3	2.3
Dixonville	4. 9	—	52	64	63	67	20.5	28.8	15.4
Parera	2.3	—	56	114	99	137	29.7	57.6	2.4
San Claudio	15.	—	—	142	118	—	5.0	32.7	—
San Antonio	3.7	—	77	120	88	85	27.4	70.3	23.6
Jornada	0.7	—	2.1	34	—	43	0.35	—	0.085
Sevilleta	0.5	—	0.9	53	—	54	0.37	—	0.078
Goodwell	—	—	—	102	85	—	0.08	1.14	—
Tribune	—	—	—	102	80	—	0.17	1.41	—
San Angelo	3.6	—	—	126	91	—	0.47	2.18	—
Quanah	2.5	—	—	132	90	—	0.32	6.36	—
Vernon	2.8	—	7.5	86	—	79	0.34	—	0.14
Riesel	4.0	—	9.4	149	85	—	2.0	9.32	—
Riesel	4.0	—	9.4	263	—	202	1.03	—	0.756
Engeling	6.0	—	50.4	74	—	42	27.6	—	14.02

Table 3.4: Ranges of net present values (NPV) of changes in ecosystem services forecasted for 30 years since land-use changes.

Site	Land-use change[†]	Carbon[‡] ($/ha)	Water[‡] ($/ha)	Realistic carbon[§] ($/ha)	Realistic water[§] ($/ha)	Net value ($/ha)	Dominant ecosystem service	Fold difference[¶]	Water values ($) and associated references[#]
Jornada	WPI	53 to 567	-17 to -2	83 to 339	-17 to -12	67 to 327	Carbon	18	Brookshire et al. 2004 (2913)
Sevilleta	WPI	11 to 118	-7 to -1	16 to 83	-6 to -1	9 to 83	Carbon	13	Brookshire et al. 2004 (2913)
Vernon	WPI	-61 to -6	0 to 0	-141 to -26	0 to 0	-141 to -26	Carbon	—	Johnson et al. 2005 (38)
Riesel	WPI	-3245 to -305	0 to 0	-2608 to -611	0 to 0	-2608 to -611	Carbon	—	(0)
Engeling	WPI	45 to 482	0 to 0	2 to 166	0 to 0	2 to 166	Carbon	—	(0)
San Angelo	Rain-fed	-2789 to -262	4 to 36	-2248 to -536	4 to 14	-2243 to -522	Carbon	342	Olenick et al. 2004 (118)
Goodwell	Rain-fed	-1003 to -94	2 to 15	-792 to -187	2 to 6	-791 to -181	Carbon	298	Gilson et al. 2001 (99)
Tribune	Rain-fed	-1806 to -170	5 to 47	-1291 to -278	20 to 47	-1271 to -231	Carbon	37	Gilson et al. 2001 (99)
Quanah	Rain-fed	-1802 to -169	3 to 27	-1454 to -347	3 to 11	-1451 to -336	Carbon	295	Johnson et al. 2005 (38)
Riesel	Rain-fed	-4819 to -452	0 to 0	-3857 to -917	0 to 0	-3857 to -917	Carbon	—	(0)
San Angelo	Irrigated	-2099 to -197	-2667 to -306	-1703 to -408	-2667 to -1909	-4370 to -2316	Water	—	Olenick et al. 2004 (118)
Goodwell	Irrigated	139 to 1479	-3966 to -454	212 to 873	-3966 to -2839	-3754 to -1966	Water	0.19	Gilson et al. 2001 (99)
Tribune	Irrigated	-919 to -86	-4667 to -535	-657 to -142	-4245 to -535	-4902 to -676	Water	—	Gilson et al. 2001 (99)
Quanah	Irrigated	-1887 to -177	-1500 to -172	-1521 to -363	-1500 to -1073	-3021 to -1436	Water	—	Johnson et al. 2005 (38)

[†]Land-use changes. WPI: Woody plant invasion, Rain-fed: rain-fed cultivation, Irrigated: irrigated cultivation.

‡Range of net present value of changes in ecosystem services for 30 years following the land-use change using all scenarios of temporal dynamics of the changes (Figure 3.2)

§Range of net present value of changes in ecosystem services for 30 years following the land-use change using realistic set of temporal dynamics of the changes (Figure 3.2)

¶Fold-difference value indicates the fold difference in net present value (NPV) necessary to offset the value of changes in the dominant ecosystem service with the other. Calculated with median NPV of carbon and water changes.

#Inflation-adjusted water values per $/acre-foot in parentheses. All currency in 2010 US$ (U.S. Census Bureau 2011).

Net present value of average farm profits from current practices.

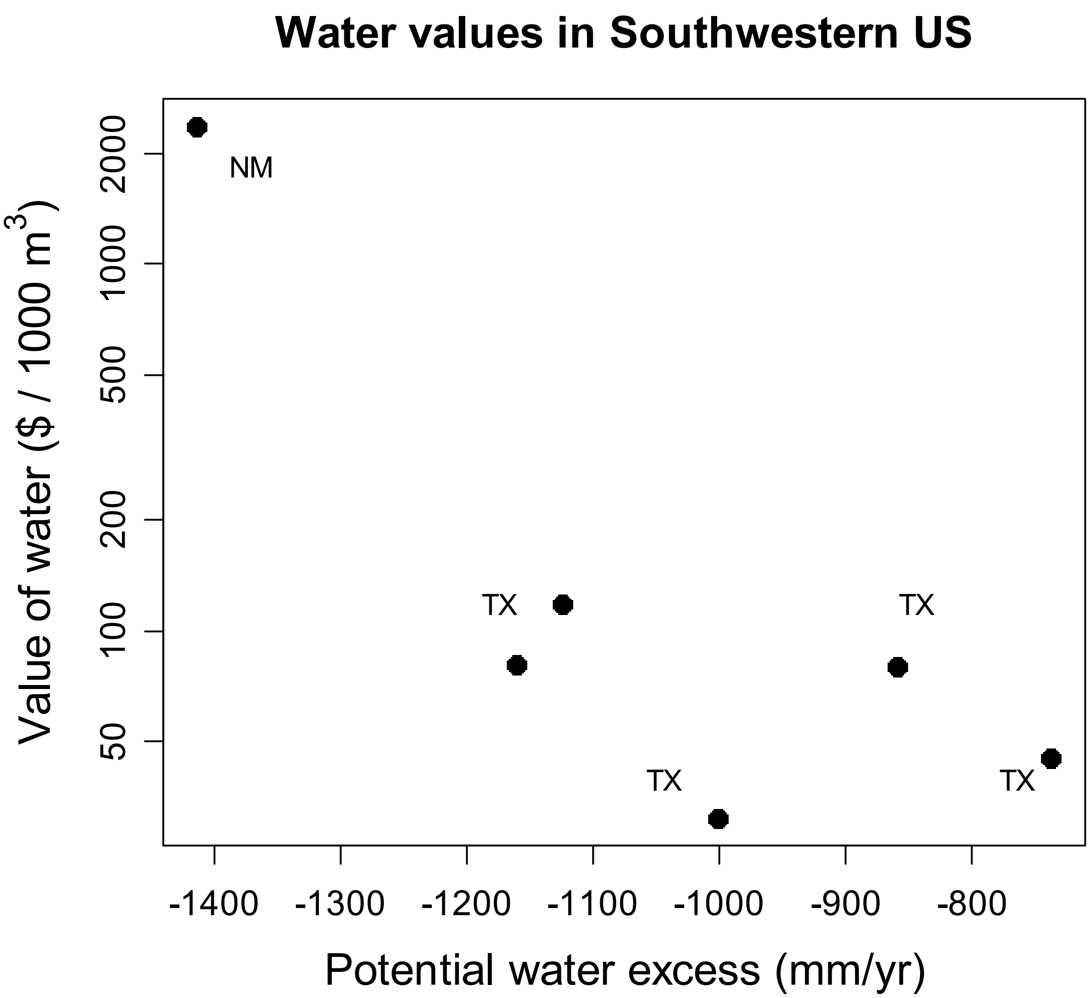

Figure 3.1: Water values in our study region from water market transactions or water valuation reports. Value of water declines with increasing potential water availability (Precipitation – Potential Evapotranspiration).

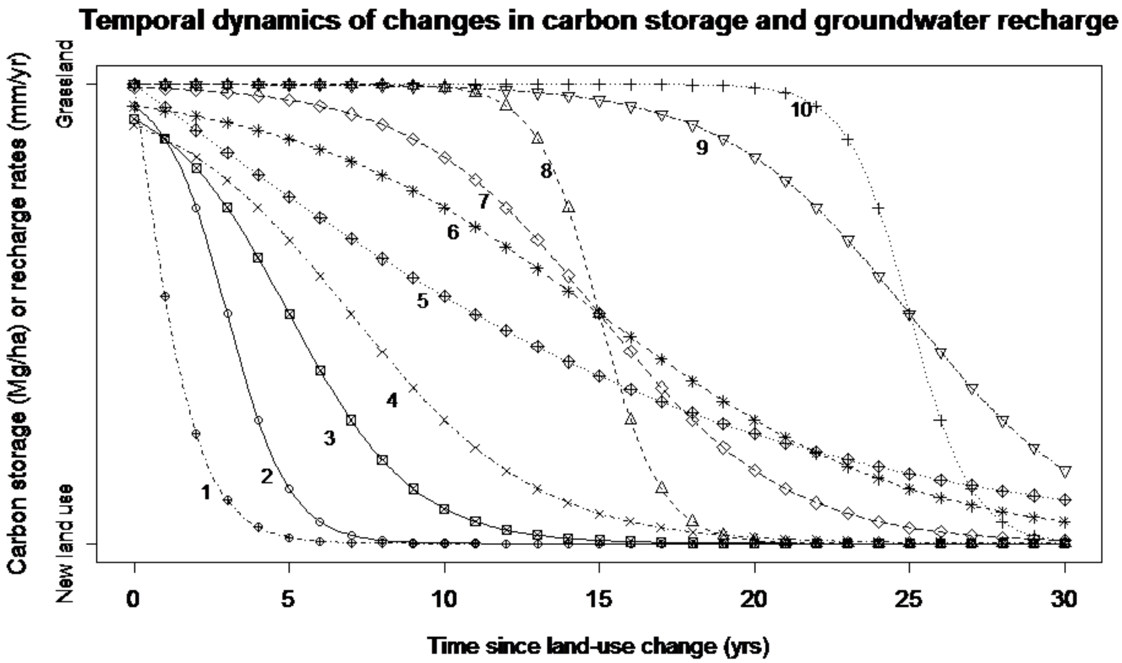

Figure 3.2: Simulated temporal dynamics of changes in carbon storage or groundwater recharge following land-use changes. Parameters for the sigmoidal function from *Eq. 3.2* were varied between 0-25 for *B* and 3-30 for *C*. Lines 3,4 were used as realistic scenarios for SOC loss, lines 6,7 for SOC gain, line 5 for biomass gain, line 1 for biomass loss, lines 1,2,3,4 for groundwater recharge loss and lines 8,9,10 for recharge gains.

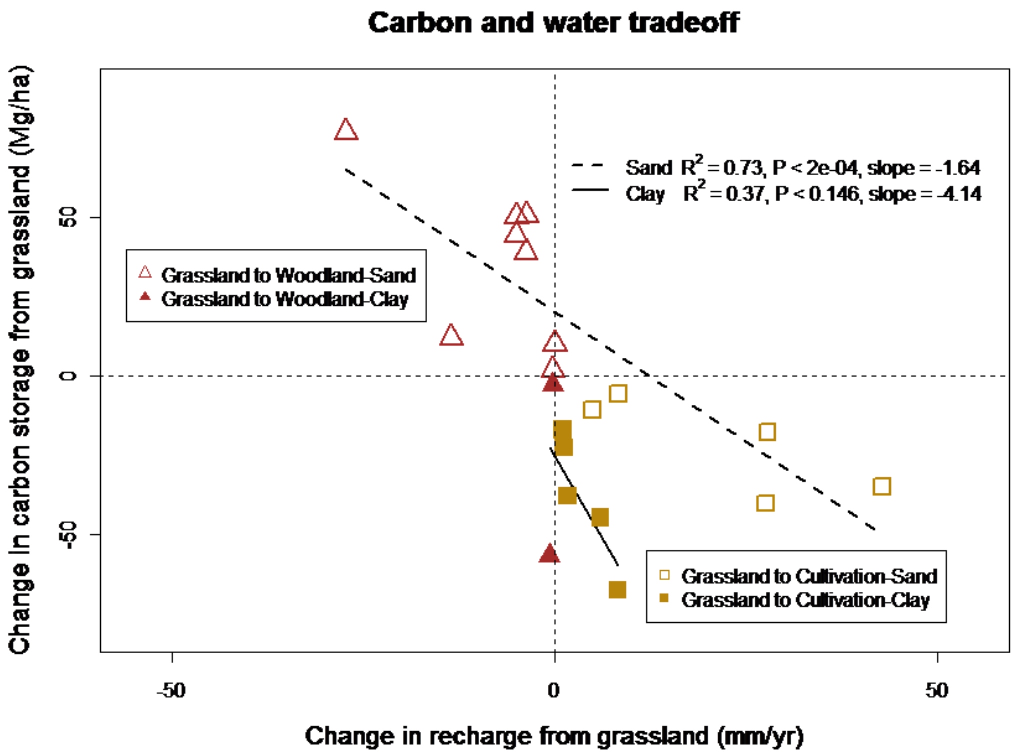

Figure 3.3: Tradeoff between changes in carbon storage and groundwater recharge with land-use changes in grasslands. Lack of data in the upper right quadrant suggests that land-use changes do not result in gains in both ecosystem services. Separate lines were fitted for data in sandy and clayey soils. Regression with pooled data is also significant to P<0.004. Slope of the lines indicate Mg/ha of carbon lost in order to gain 1 mm/yr (or 10 m^3/ha·yr) of recharge.

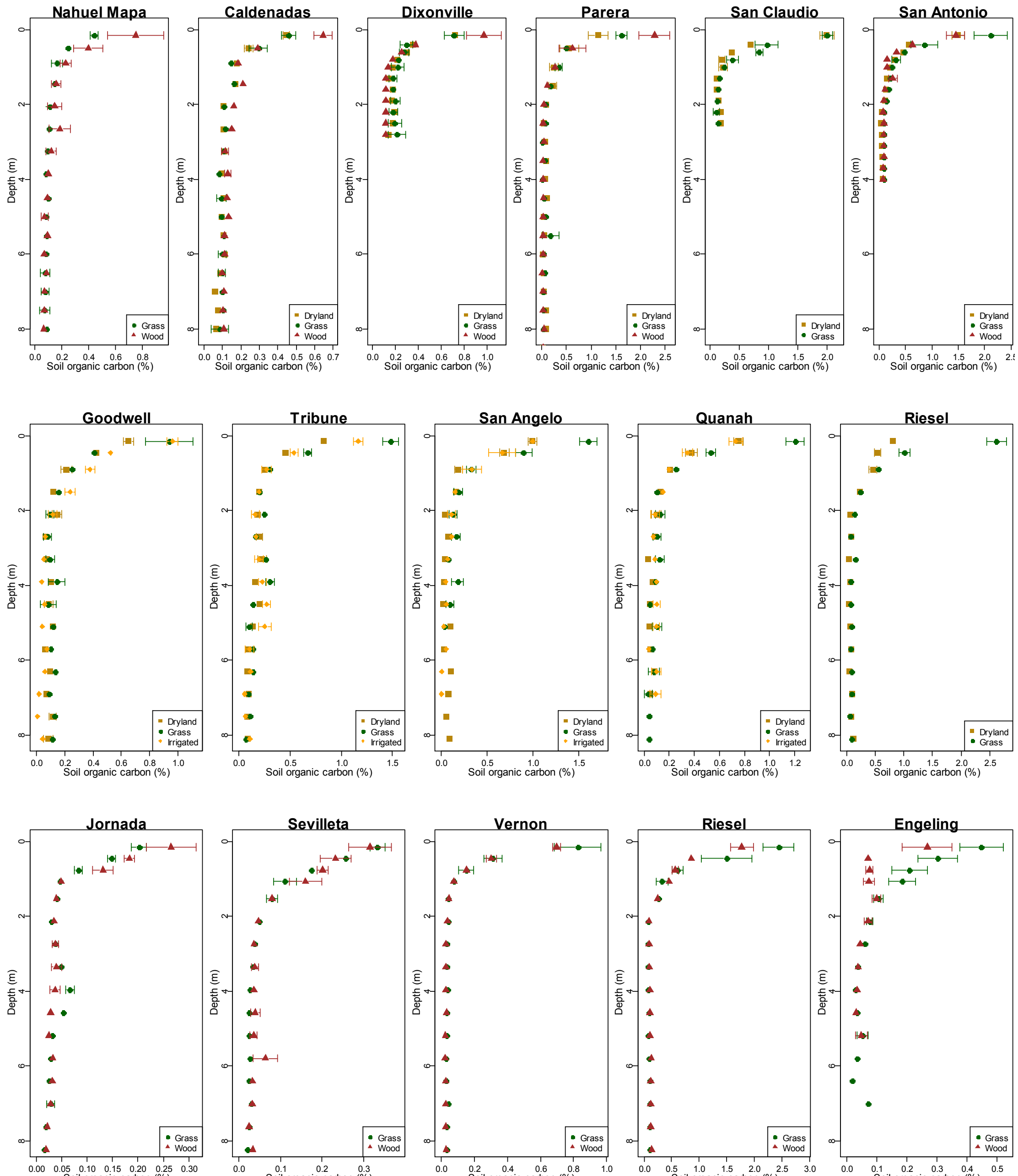

Figure 3.4: Soil organic carbon profiles at a) Argentinean and b) southern Great Plains and c) southwestern U.S. (reproduced from Jackson et al. 2002) sites.

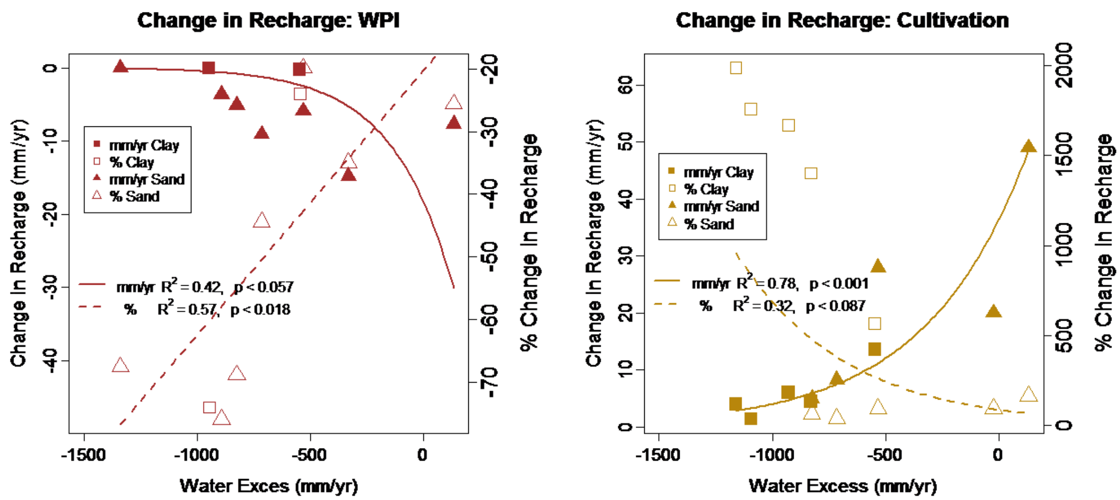

Figure 3.5: Absolute and relative changes in groundwater recharge between a) grassland and woody plant invasion plots and b) grassland and cultivation plots across potential water excess (Precipitation – Potential evapotranspiration). Solid lines and filled symbols denote absolute differences in recharge, and dashed lines and open symbols denote relative differences. Fitted lines were chosen from linear regressions on log- or un-transformed differences in recharge. Reproduced from Kim and Jackson (Kim and Jackson, In review).

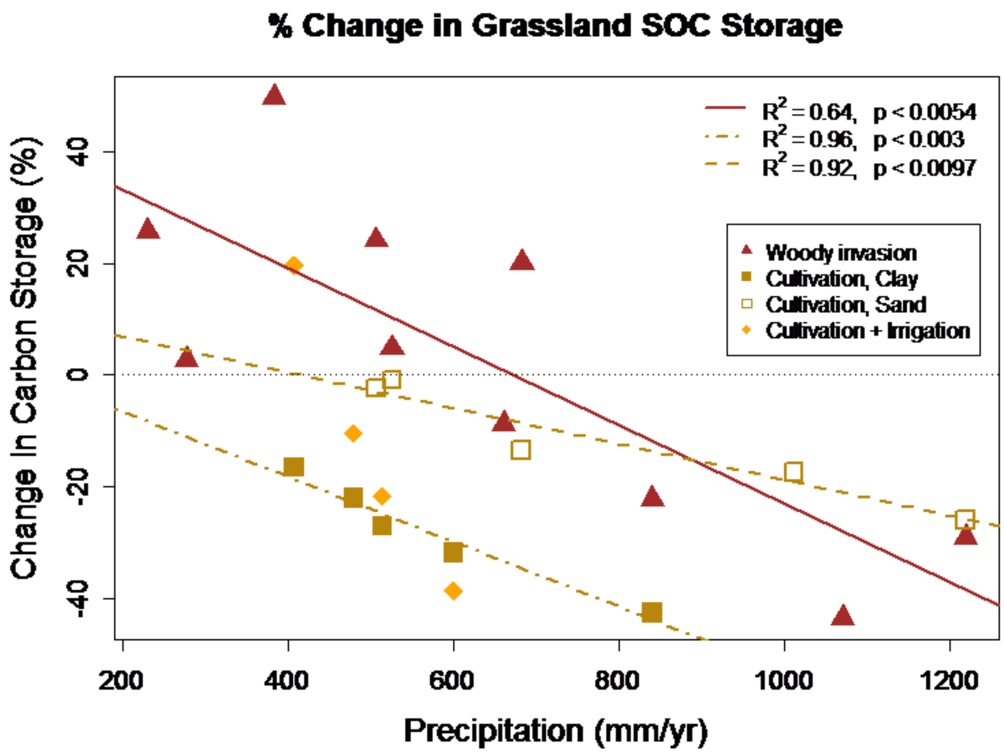

Figure 3.6: % change in grassland soil organic carbon storage with land-use changes across a precipitation gradient. Note that regressions for rain-fed cultivation are separated by soil texture categories.

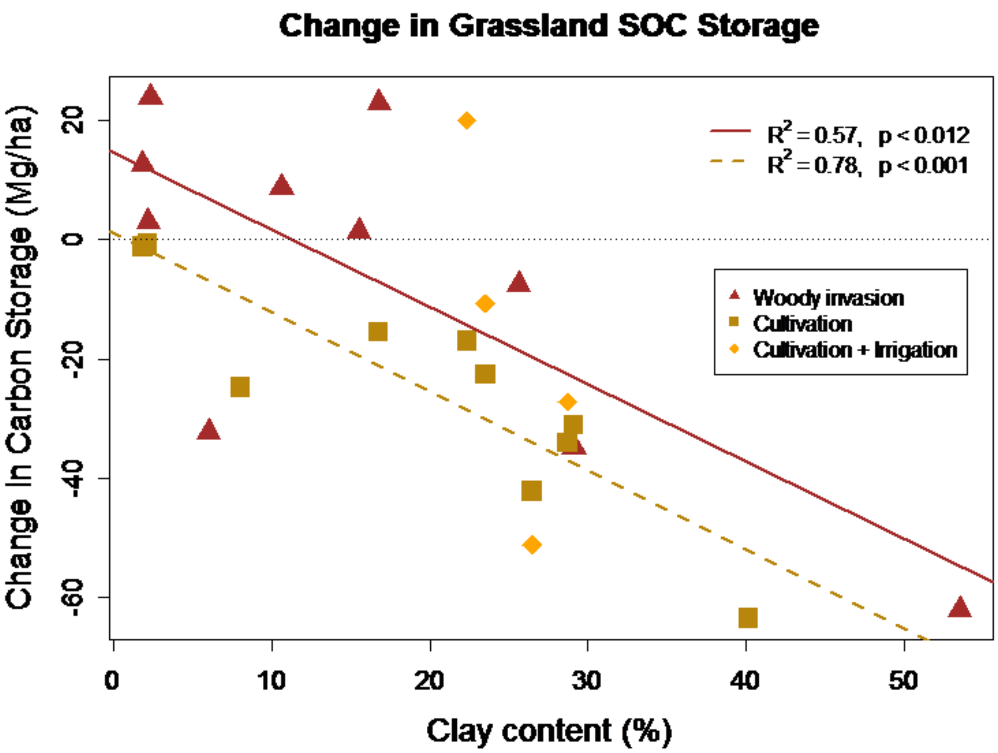

Figure 3.7: Change in grassland soil organic carbon storage with land-use changes across clay content in the top 1 m.

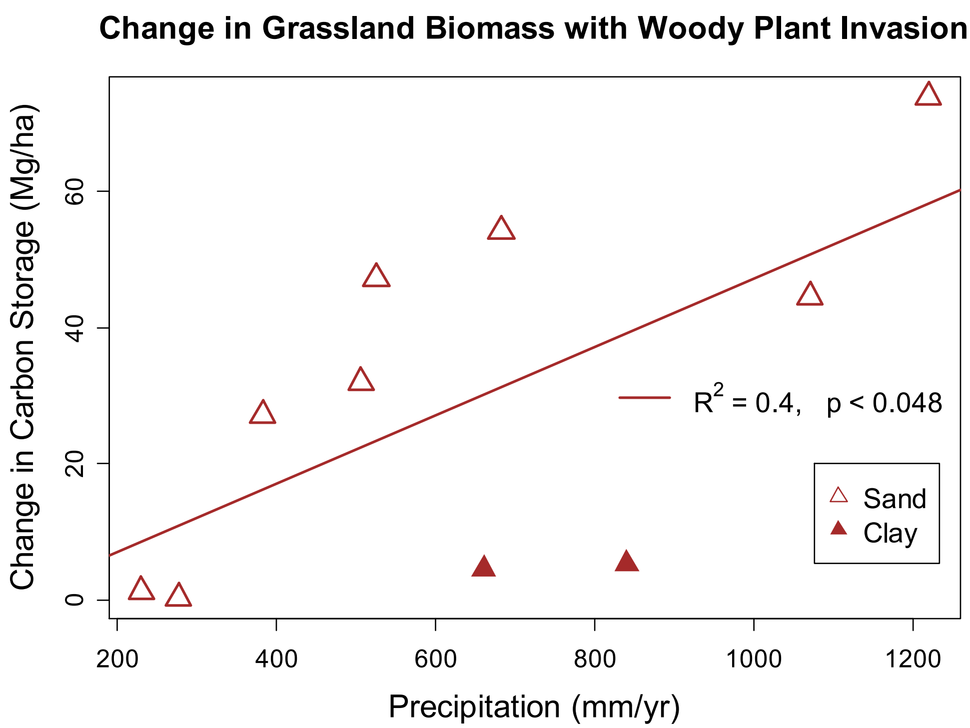

Figure 3.8: Change in biomass carbon with woody plant invasion along a precipitation gradient.

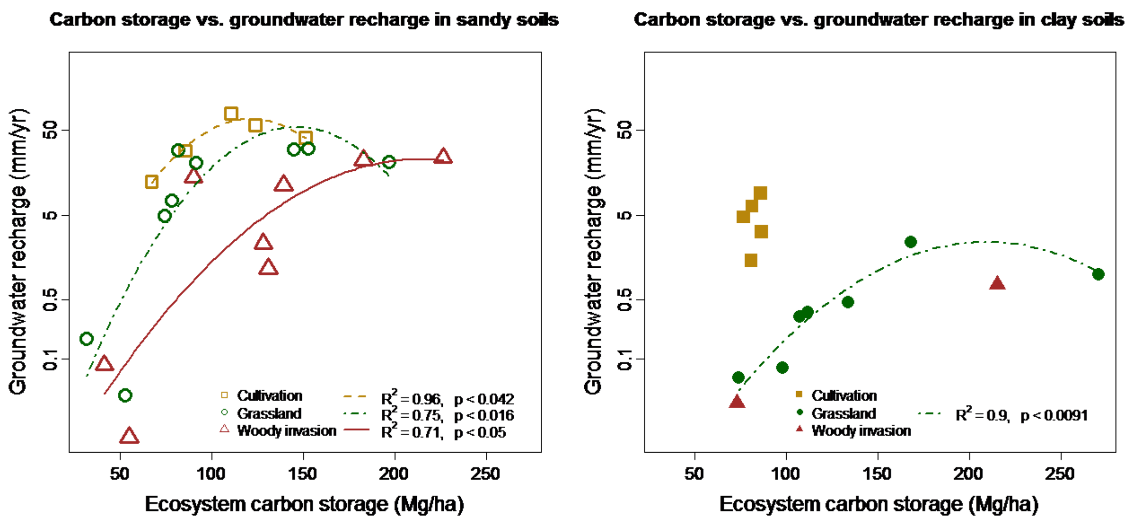

Figure 3.9: Ecosystem carbon storage vs. groundwater recharge for the three land-uses in a) sandy soils and b) clay soils. Carbon storage and groundwater recharge tended to increase together.

4. Ecosystem loss of soil inorganic carbonates from agricultural conversion of grasslands

4.1 Introduction

Carbon sequestration by storage of organic carbon in plant biomass and soils is a frequently featured policy mechanism to mitigate rising atmospheric CO_2, the key driver of climate change (Pacala and Socolow 2004, IPCC 2007). Although the global soil inorganic carbonate (SIC) pool is comparably large, equivalent to 100% and 50% of terrestrial biomass and soil organic carbon pools respectively, it has received less attention by both scientific and policy communities (Schimel et al. 1995, Batjes 1996, Eswaran et al. 2000, Jobbágy and Jackson 2000). Pedogenic carbonates form in chemical weathering reactions involving water, CO_2, and a metal (Ca or Mg), expressed as (Birkeland 1984):

$$2CO_2 + 2H_2O \rightleftharpoons 2H^+ + 2HCO_3^- \qquad \textit{Eq. 4.1}$$

$$Ca^{2+} + 2HCO_3^- \rightleftharpoons CaCO_3 + H_2O + CO_2$$

Relative dearth of scientific or policy discussions about SIC in regards to climate change mitigation partly stems from an understanding that SIC is an ineffective source or sink for CO_2 due to its slow turnover rate, forming over centennial or longer timescales (Schlesinger 1982, Berner et al. 1983, Lal and Kimble 2000, Williams et al. 2003). Some recent studies have hinted at a more active SIC pool, focusing on the sink potential in natural desert ecosystems (Monger and Gallegos 2000, Emmerich 2003, Mielnick et al. 2005, Kowalski et al. 2008, Wohlfahrt et al. 2008, Serrano-Ortiz et al. 2009, Liu et al. 2011).

However, most of these studies have not considered effects of land-use changes on the SIC pool nor considered it as a source of CO_2.

Greater soil water fluxes and acidification that usually accompany agricultural conversion of natural vegetation may have percolating effects on SIC storage. Addition of water in *Eq.* 4.1 may force the equilibrium towards dissolution of the existing SIC, which may be leached to ground water as bicarbonate (HCO_3^-) or evolved out as gas in presence of acids (Helyar et al. 1989, Moody and Aitken 1997). Previous studies that reported declining SIC storage with increasingly humid climates, and hence soil water fluxes, support the hypothesis (St. Arnaud 1979, Kelly et al. 1991).

Comparisons of SIC storage under different land uses or vegetation types have yielded variable findings. Many studies observed increases in SIC storage in the top 50 to 200 cm depths with cultivation of natural vegetation, and attributed this pattern to addition of ion-rich irrigation water or freeze-thaw cycle inducing precipitation of carbonates (Cihacek and Ulmer 2002, Denef et al. 2008, Mikhailova and Christopher 2006). In support of the latter, declining SIC with increasing temperature has been demonstrated previously, though it is unclear how land-use change interacts with the abiotic factor (Kelly et al. 1991). On the other hand, studies reporting decrease in SIC with land-use changes have attributed the loss to erosion or dissolution of the carbonates (Papiernik et al. 2007, Sartori et al. 2007).

To examine whether cultivation can result in loss of SIC from the soil, we collected soil samples to 9 meter depth in natural grassland plots paired to rain-fed and irrigated cultivation plots in 5 locations across a precipitation gradient in the southern Great Plains. Arid and semiarid grasslands contain virtually all of the global SIC (Lal and Kimble 2000), but they have also experienced widespread conversion to crop cultivation, making them ideal systems to test whether land-use change results in loss of SIC.

4.2 Methods

4.2.1 Site description

In the southern Great Plains of the United States, we selected five sites along a precipitation gradient (407-890 mm/yr). When available, rain-fed and irrigated cultivation plots were paired to an adjacent or nearby natural grassland plot within 1 km of each other (Table 4.1). Most plots have 30+ years of constant land-use histories. We surveyed land-owners or farm managers for land-use history at each site, including cropping schemes (e.g., species, rotations) and fertilizer/pesticide/irrigation inputs (Table 4.1,4.2). Precipitation data were obtained from long-term (40+ years) records maintained by weather stations within 30 km of the sites (NOAA). Most of our sites were located in agricultural research extension centers with good local records of the weather and land management history (Table 4.1).

4.2.2 Soil sampling

We used a direct-push coring rig (Geoprobe Systems, Salina, KS) to take five to eight cores per plot up to 8.9 m depth in March-April of 2010. The only plot soil samples were not retrieved to 7.2 m depth was at a grassland plot near San Angelo, TX, due to indurated material found around 4.5 m depth. Soil cores were weighed in the field, subsampled separately for soil moisture/bulk density using intact cores and for elemental analysis using sieved and homogenized soil samples, and shipped to the lab for analysis. Segregated carbonates (nodules) were recovered from sieving and kept separately for analysis.

4.2.3 Lab analysis

Soil, biomass, and segregated carbonate samples were dried in the oven for gravimetric moisture content calculations and ground for carbon measurement on Carlo Erba Elemental Analyzer using the two temperature combustion method (Chichester and Chaison 1992). Results from this method correlated well with results from acid-fumigation technique (Harris et al. 2001) on a subset of our samples (data not shown, R^2=0.98, N=97). Soil texture was determined in the laboratory by the pipette method (Klute 1986) and ranged from silty loam to clay (Table 4.1). Bulk densities were estimated from dry weights and volumes of the soil samples. % inorganic carbon contents of combined soil and the carbonate nodules were multiplied by bulk density estimates by depths to estimate soil organic carbon storage. We compared SIC storages under rain-fed and irrigated croplands and grassland triplets in the top 2.4, 4.8, and 7.2 m of soil to examine the effect of depths in our analysis. For anion analysis, dried and homogenized soil samples were shaken with equal weight of double deionized water for four hours. The mixture was centrifuged, the supernatant filtered, and the filtrate analyzed for Cl^-, SO_4^{2-}, HCO_3^-, and Br^- by ion chromatography (Dionex ICS-2000). We also analyzed for HCO_3^- by acid-titration, and for pH using pH electrodes on a subset of the samples.

4.2.4 Groundwater recharge estimations

In addition to the soil inorganic carbon data, we present groundwater recharge rates previously reported in Chapter 2 (Kim and Jackson in review) calculated from chloride mass balance and tracer displacement methods (Allison and Hughes 1983, Walker et al. 1991, Phillips 1994). We examined relationships between soil inorganic carbon storage and recharge rates by regression to test our hypothesis that higher soil water fluxes may reduce SIC storage. In addition, we determined the bicarbonate flux in deep drainage by multiplying the recharge rates by average HCO_3^- concentrations below the root zone.

4.2.5 Long-term groundwater quality monitoring

To test our hypothesis that crop cultivation may result in dissolution of carbonates and leaching of bicarbonates to the ground water, we looked for indications of long-term changes in the alkalinity in the shallow ground waters in our study region to in two online databases (TWDB , USGS). We selected groundwater quality data series with at least 20 observations and of sampling depths less than 60 m throughout Kansas, Oklahoma, and Texas, where large-scale grassland conversions to croplands occurred in the decades leading up to the 1940's (Gutmann et al. 2005).

4.2.6 Literature review

We also conducted a literature review of SIC storage under paired comparisons of different land uses to examine whether the trends observed at our sites are universal. SIC stocks, land uses, and environmental variables such as soil texture and mean annual precipitation and temperature were recorded from studies or from global climate databases based on the study locations (New et al. 2002).

4.2.7 Statistical analysis

We used t-tests to compare both SIC stocks and % soil inorganic carbon between different land uses by depths. To test for relationships between land-use changes, abiotic factors and soil carbonates, we analyzed, by regression, absolute and relative changes in SIC from grassland conversions across precipitation, potential water excess (PWE), temperature, and clay contents within the upper 1 m for both our new field data and data from the literature review. Relative changes from grassland SIC were calculated as:

$$\frac{SIC_{new} - SIC_{grass}}{SIC_{grass}} \qquad \textit{Eq. 4.2}$$

where *SIC* is soil inorganic carbon storage and subscripts *new* and *grass* denote new land use and grassland, respectively. PWE was calculated by subtraction long-term mean annual precipitation by potential evapotranspiration calculated by Penman-Monteith equation using high resolution climate dataset (New et al. 2002). We analyzed relationships between SIC storage and recharge rates with linear regression. Long-term temporal trends of well alkalinity data were also analyzed by regression.

4.3 Results

SIC storage increased with depth at all our sites, with 446 Mg/ha on average in the first 2.4 m of the soil compared to 907 and 1043 in the second and third 2.4 m of the soil (Table 4.3). Except for our most humid site, %SIC peaked at or below 2.1m depths regardless of the land uses, and we also observed greatest changes in SIC below 2.1 m (Figure 4.1, Table 4.3). The surface soil layer may represent a more reactive medium for carbonate chemistry due to heightened effects of climate and vegetation, about an order of magnitude larger than SOC at our sites (Table 3.3). The large storage in the deeper layers nevertheless represents significant potential for both storage and change.

When examining soil to 4.8 m depth or deeper, we observed significantly less SIC storage under cultivated plots compared to their grassland pairs at all our sites, indicating that cultivation of natural grasslands may result in loss of carbonate from the soil (Table 4.3). Grasslands on average lost 518±132 Mg/ha of SIC with rain-fed cultivation and 851±93 Mg/ha of SIC with irrigated cultivation. To our knowledge, this is the first documentation of such extensive and rapid loss of deep SIC with land-use changes.

Changes in SIC storage with grassland conversions also depended on the soil depths included in our analysis. At three of our five sites, SIC increased with cultivation in the top 2.4 m, but when we included soil depths past the 2.4 m in the analysis, cultivated plots had less SIC than their grassland pairs at all sites. At these three sites, grassland SIC contents were significantly higher than those of their cultivated pairs at 0.15-0.45 m depths but were lower than their cultivated pairs at 3.9-4.5 m depths (Figure 4.1), indicating that shallow sampling may not capture complete dynamics of SIC changes with grassland conversions (Table 4.3).

SIC storage and its changes from cultivation were affected by abiotic factors. SIC storage decreased with increasing humidity or potential water excess (PWE = Precipitation – Potential Evapotranspiration), indicating that aridity may enhance build-up of ions and precipitation of soil carbonates from low soil moisture water fluxes (Figure 4.2a). Loss of SIC from cultivation became less pronounced with PWE, indicating that larger storage in more arid climates may represent higher potential for losses (figure 4.2b). Moreover, we found significant correlation in our literature review between SIC changes from land-use changes and mean annual temperature (Figure 4.3), indicating that freezing-induced precipitation of carbonates may be more pronounced under cultivation (Cihacek and Ulmer 2002, Mikhailova and Christopher 2006, Denef et al.

2008). Although changes in SIC to 2.4 m in our field data did not show similar trends with mean annual temperature (R^2=0.26, P<0.16, Figure 4.3), it did with other climatic indicators of temperature such as number of days above freezing (R^2=0.55, P<0.02, data not shown), suggesting that carbonate chemistry in shallow soil layers may be driven by temperature as well as water availability.

Average SIC contents to 4.8 m was negatively correlated with deep drainage rates calculated from the matching soil samples, indicating that higher soil water flux observed under croplands compared to grasslands (Table 2.4) may lead to dissolution and leaching of grassland soil carbonates (Figure 4.4). Also in support of the dissolution and leaching hypothesis, estimated bicarbonate fluxes under cultivated plots were greater compared to their grassland pairs, indicating that cultivation may be increasing bicarbonates (Eq. 4.1) in deep drainage (Table 4.3, Eq. 4.1).

Another piece of evidence for leaching as a potential mechanism for SIC loss is that most of the statistically significant alkalinity trends in shallow (20 to 60 m) aquifers throughout Kansas and Texas showed changes in alkalinity during 1940-2000 periods (Table 4.4). Out of 44 wells analyzed, 13 had statistically significantly positive trends and 9 had negative trends. The increases occurred despite the increasing trends of precipitation in the region (Garbrecht et al. 2004), indicating that land-use change, rather than changes in precipitation may be responsible for the increase in leaching of bicarbonate to the water table. Although the data do not point to a significant increase or decrease in alkalinity for the region as a whole, most of the increases were observed at sites with drier climates, with the trends becoming more negative with increasing site humidity, suggesting dilution of alkalinity from enhanced recharge at humid climates (Figure 4.5).

4.4 Discussion

Our results raise the possibility that cultivation of grasslands may result in a loss of soil carbonates from the soil profile to at least 6 m depth, although our analysis needs much more work to determine whether land-use changes are the primary cause. If our analyses are correct, however, several important factors emerge. For instance, SIC at such depths may be a much more dynamic carbon pool than previously thought. Ignoring the contribution from the parent material, and assuming the atmosphere to be the exclusive source of Ca^{2+}, it would take 30,000-300,000 years to build up pedogenic $CaCO_3$ to the observed levels at our sites to 2 m depth (NADP, Marion et al. 1985). Our results suggest that up to 70% of that storage can be lost from the soil in comparatively short amount of time, 30-100 years, in the case of at least one site. The amount of carbon lost was an order of magnitude larger than changes in soil organic carbon from the same study system (Table 3.3, Table 4.3). However, because we do not know the fate of the carbon in inorganic losses, our results do not point definitively to the SIC pool as either a net emitter or sequester of atmospheric carbon; we discuss some likely mechanisms and implications of the loss below.

Based on our data, increased water flux from cultivation of grasslands likely induced dissolution and leaching of bicarbonates to the ground water. This agrees with the conceptual model of carbonate equilibrium, where increased solvent volume (water) would push the equilibrium towards dissolution of carbonates (*Eq*. 4.1). Decreasing SIC storage with increasing soil water fluxes (Figure 4.4) and higher bicarbonates fluxes under cultivation compared to their grassland pairs (Table 4.3) support dissolution and leaching as likely mechanisms for loss of soil carbonates. Our examination of long-term bicarbonate monitoring data of shallow aquifers in the region also suggests that the increase in alkalinity may be regionally common. The alkalinity analysis further showed increases at drier sites but decreases at wetter sites (Figure 4.5), pointing to dilution effect from increased recharge in humid areas where such fluxes are not transport-limited or where SIC storages are not significant (Williams et al. 2007). However, it is not clear whether the increase in alkalinity in drier regions is an effect of carbonate dissolution alone or if it can account for the large carbonate loss we observed. Dissolution and leaching of carbonates should result in a loss of Ca^{2+} from the soil and an increase in Ca^{2+} concentrations in ground waters. Further examinations of regional inputs and outputs of bicarbonates and calcium from long-term groundwater or surface water monitoring datasets for these trends are needed to strengthen our observations.

Although the evidences point to increased export of bicarbonates to the ground water with conversion of arid grasslands, it is not certain that this loss represents emissions or sequestration. In systems where ground water has geologically long residence time, the leached carbonates may represent a long-term storage, whereas hydrological connection to a surface water body may result in release of CO_2 during transport to the ocean, where its sequestration depends on the composition and conditions of the mixing seawater. Raymond et al. (2008) had attributed increased export of bicarbonates in the Mississippi river over the last century to liming of agricultural soils, but our results indicate that dissolution of carbonates may also contribute to riverine export of bicarbonates where ground water and surface water are well-connected. Moreover, dissolution of a carbonate mineral can actually consume CO_2 (Lal and Kimble 2000), and hence the fate of bicarbonates in the ground water is doubly important for sequestration or emission potential of the SIC pool from disturbances such as land-use changes.

Our results also indicate that SIC dynamics may be more complicated than previously thought, with different factors affecting shallow and deep SIC. The extent to which the SIC pool responded to land-use changes depended on the soil depths of our analysis, as we observed SIC gains instead of losses with cultivation at some grasslands in the top 2.4 m. We observed similar increases in our literature review, which were higher in colder climates but replaced with losses in warm climates. Because of the lower solubility of CO_2 in water with increasing temperatures, calcite tends to precipitate more in warmer temperatures, but freezing temperatures may encourage carbonate precipitation, as ion exclusion likely does, which increases bicarbonate concentrations in the soil solution during ice formation (Cerling 1984, Marion 2001). Freezing may induce precipitation of calcite from the soil solution which may be exaggerated by deeper freezing under unvegetated surfaces such as fallow cops (Anderson 1947, Teepe et al. 2000). However, these increases represent a small fraction of the deeper changes in SIC, which more than offset the gains in the top 2 m of soil. Moreover, such gains are unlikely to result from plant inputs of CO_2 alone, as such gains would suggest unrealistically large increases in productivity and belowground input of carbon as a result of cultivation. Sufficient *increases* in carbon inputs for croplands relative to grasslands to account for the increase in SIC in the top 2.4 m would be 0.5-2.8 Mg C/ha/yr, compared to total average net primary productivity of 2.8-3.6 C/ha/yr for croplands in the region, much of which is harvested off the land (Bradford et al. 2005). Instead, bicarbonates that contribute to carbonate formation under croplands are likely to have originated from decomposition of the organic carbon upon tillage. Our results overall imply that the carbonate dynamics near the soil surface may be driven by temperature and that deeper carbonate pools may be more sensitive to changes in subsurface hydrology. Exact mechanisms of interaction between abiotic factors, SIC and land-use changes need further study.

4.5 Conclusion

We report here potentially rapid and deep changes of soil carbonates in the most complete inventory of SIC changes with depth accompanying land-use changes to date. Such changes were surprising and need considerable research to examine their occurrence in other locations. We observed significant SIC changes within 30-100 years after grassland conversions, with shallow soil layers tending to gain SIC in colder climates and deeper soil losing SIC. Significant losses were also observed beyond 7 m depth, indicating that deep carbonate cycling may be more dynamic than previously thought and that deep sampling may be necessary for capturing complete SIC dynamics. The losses were likely the result of increased recharge under croplands shifting the chemical equilibrium towards dissolution of carbonate minerals. Observed negative correlations between recharge rates and SIC storage, greater bicarbonate export in deep drainage under croplands, and increases of alkalinity in some shallow wells throughout the more arid parts of the region supported the proposed mechanism of loss based on conceptual understanding of carbonate formation, although bicarbonate and calcium fluxes should be balanced in further studies to test the fate of the dissolved bicarbonates (leaching vs. evolution). In light of our findings, the fate of bicarbonates in ground water becomes critical in determining the net emission or sequestration of CO_2 with land-use conversions. Relationships between SIC and environmental factors suggest temperature as well as climate affect SIC dynamics and open new questions and opportunities regarding abiotic controls on both shallow and deep cycling of SIC.

Table 4.1: Site information.

Site	Lat., long. (degrees)	Precip (mm/yr)	Soil	Land use	Years since change
Goodwell	36.6, -101.6	407	Gruver clay loam	Grass, Rain-fed, Irrigated	60, 60
Tribune	38.5, -101.6	479	Richfield silt loam	Grass, Rain-fed, Irrigated	30, 50
San Angelo	31.4, -101.3	514	Angelo clay loam	Grass, Rain-fed, Irrigated	100, 40
Quanah	34.3, -99.8	679	Sagerton (Olton) clay loam	Grass, Rain-fed, Irrigated	100, 60
Riesel	31.5, -96.9	890	Heiden Clay	Grass Rain-fed	100+, 100+

Table 4.2: Dominant species at each site and land use.

Site	Dominant species		
	Grasses	Rain-fed†	Irrigated
Goodwell	*Bouteloua* and *Andropogon spp*.	W, So	C, W, So, Su
Tribune	*Bouteloua* and *Andropogon spp*.	W	C, So
San Angelo	*Bouteloua* and *Buchloe spp*.	Ct	Ct
Quanah	*Stipa spp.*	Ct	Ct
Riesel	*Schizachyrium scoparium*	C	--

†C-corn, S-soybean, W-wheat, Su-sunflower, So-sorghum, Ct-cotton

Table 4.3: SIC storage under each land use by depths.

		SIC storage (Mg/ha)			HCO_3^- flux (kg/ha/yr)†		
Site	Depth (m)	Grass	Rain-fed	Irrigated	Grasses	Rain-fed†	Irrigated
Goodwell	2.4	140	241	158			
	4.8	1475[a]	1299[a]	834[b]			
	7.2	2803	2492	2015	0.035	0.469	21.2
Tribune	2.4	132	214	167			
	4.8	831	623	608			
	7.2	2246[a]	2151[a]	1651[b]	0.133	0.538	39.6
San	2.4	1081	1180	1051			
Angelo	4.8	3385[a]	2950[a,b]	2766[b]			
	7.2	5229‡	4289	4120	0.750	0.737	20.9
Quanah	2.4	384[a]	125[b]	127[b]			
	4.8	1060[a]	407[b]	462[b]			
	7.2	1744[a]	1047[a,b]	832[b]	0.203	1.39	106
Riesel	2.4	491[a]	129[b]	NA			
	4.8	1125[a]	498[b]	NA			
	7.2	1424[a]	877[b]	NA	0.450	2.14	NA

†Bicarbonate flux calculated as deep drainage multiplied by HCO_3^- concentrations in the soil.
‡Estimated assuming constant SIC content below 4.8 m using average of values for 3.0-4.8 m
[a,b] denote significant differences to $P<0.05$.

Table 4.4: Temporal trends of alkalinity measurements in shallow groundwater wells in Texas and Kansas.

Latitude	Longitude	Precipitation (mm/yr)	PET (mm/yr)†	Slope
37.25	-101.92	403	1461	-0.32
37.92	-101.25	422	1403	-0.69
38.08	-101.92	405	1382	1.07
38.25	-101.42	411	1349	1.25
38.42	-99.58	513	1339	-0.8
38.42	-99.08	570	1317	1.08
39.42	-99.58	552	1247	2.98
39.75	-99.08	563	1224	1.67
37.58	-97.58	670	1296	1.07
37.92	-97.75	675	1292	1.98
39.92	-95.58	813	1088	0.8
38.08	-97.58	701	1279	4.11
38.08	-97.58	701	1279	2.75
37.25	-97.08	776	1295	-0.48
39.08	-96.75	800	1169	-1.1
37.92	-97.58	672	1285	4.68
39.08	-98.92	584	1266	1.21
39.92	-97.58	688	1153	0.67
30.42	-97.75	692	1441	-1.37
30.42	-97.75	692	1441	-1.95
30.42	-97.75	692	1441	-2.27
29.42	-100.75	418	1571	-2.74

†Potential Evapotranspiration calculated with Penman-Monteith combination equation based on climate data from New et al. (2002).

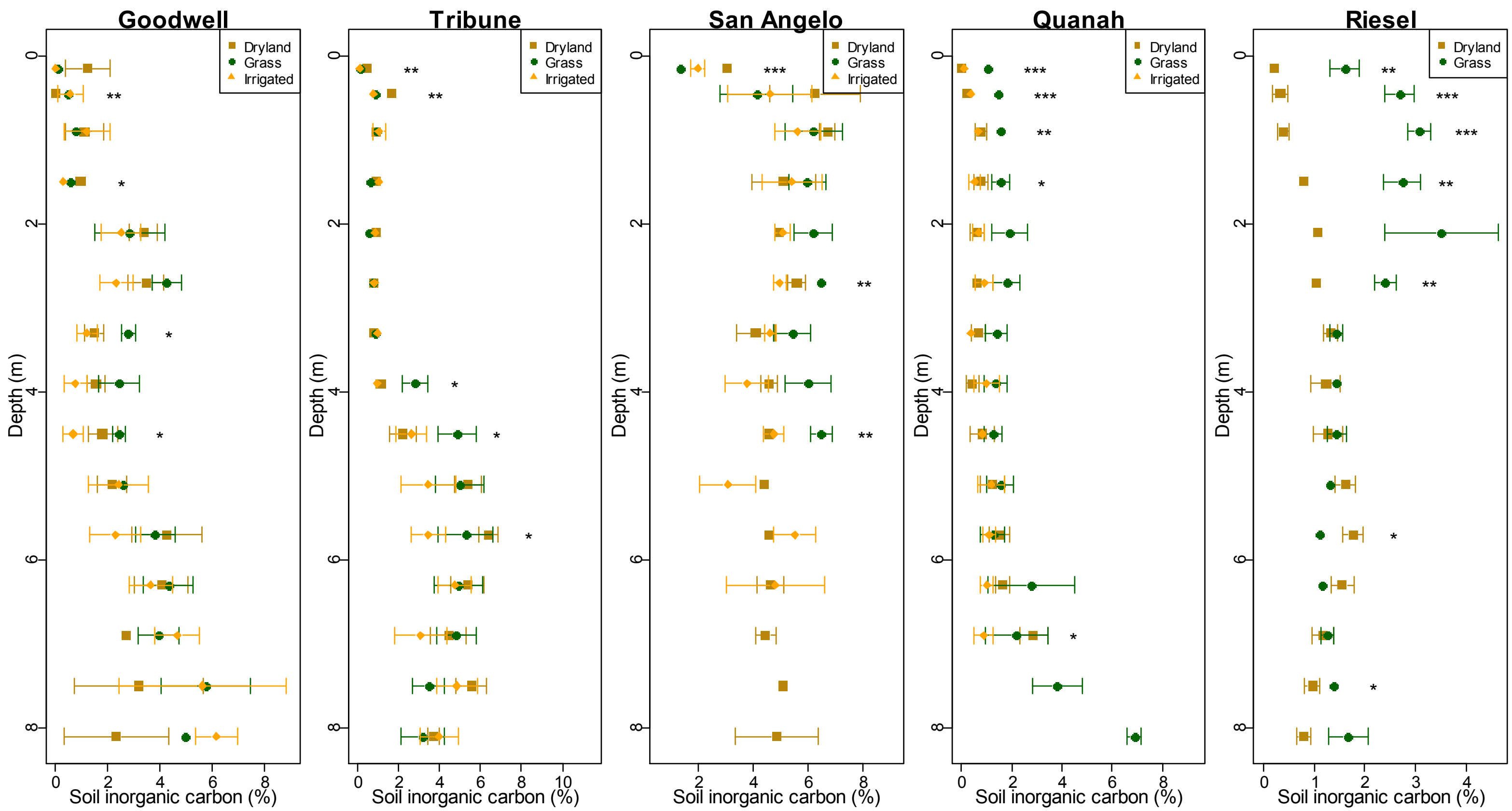

Figure 4.1: Depth profiles of SIC for the paired land uses at our five sites. Data presented are means of %SIC by weight, including carbonate nodules, and standard errors. *, **, and * indicate one or more significantly different comparisons between the land uses for specific depth at P<0.05, 0.01, and 0.001, respectively. Sites are ordered from left to right by increasing mean annual precipitation.**

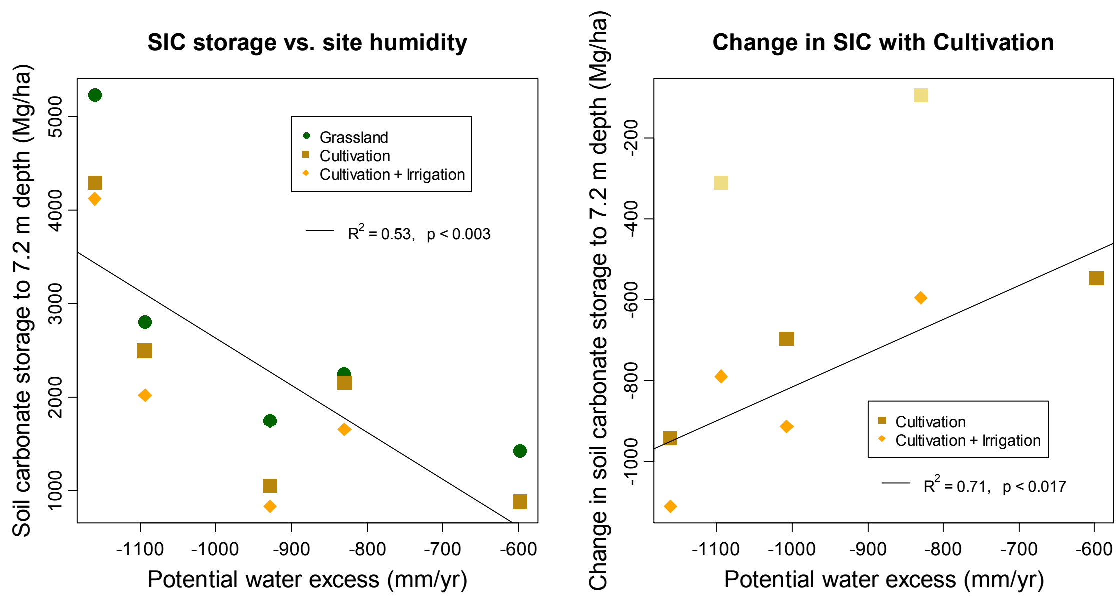

Figure 4.2: a) Soil inorganic carbon (SIC) storage to 7.2 m depth in grassland, rain-fed- and irrigated cultivation and b) changes in grassland SIC storage to the same depth from rain-fed- and irrigated cultivation across a potential water excess (Precipitation – PET) gradient. SIC storage decreased with increasing site humidity (left to right on the x-axis). Grasslands located in more humid climates lost less SIC than those in more arid climates. Irrigated cultivation pairs tended to lose more SIC than their rain-fed pairs. Line for b) drawn without the two greyed out points belonging to two sites with most recent conversions to rain-fed cultivation (30-40 years) and incomplete wetting of the whole profile (Chapter 2). Statistics with the points included are: R^2=0.23 and P<0.14.

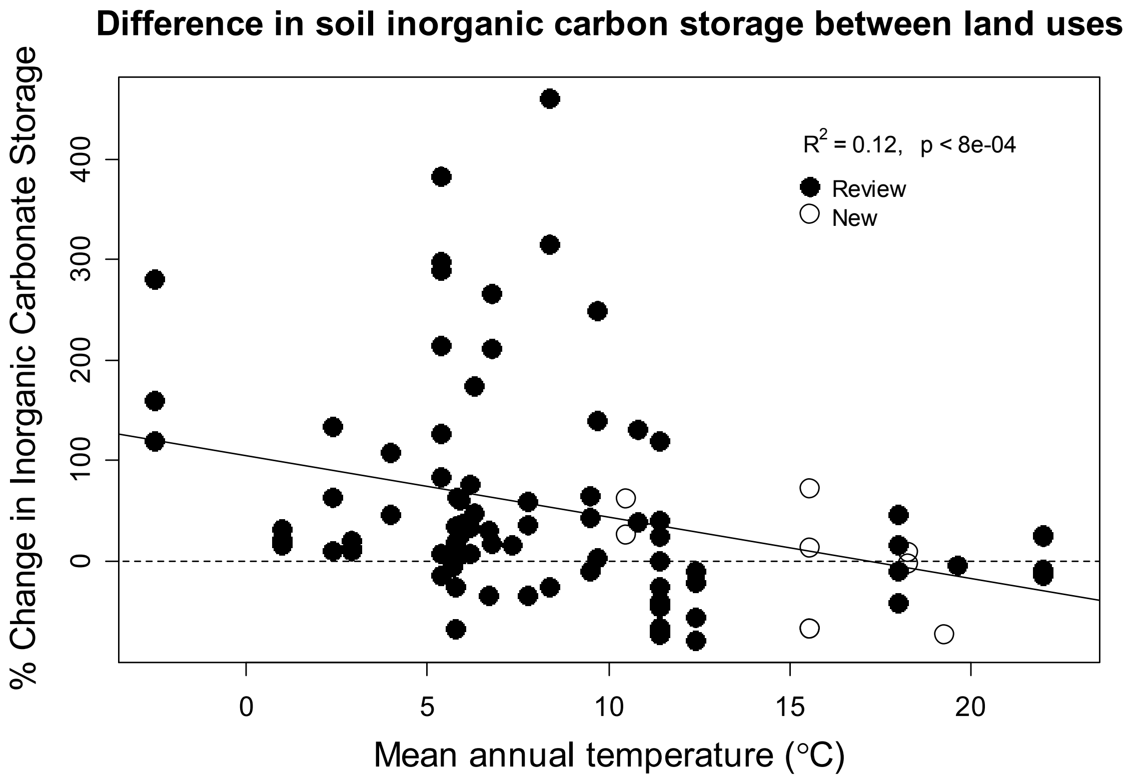

Figure 4.3: Relative changes in soil inorganic carbon storage from studies with paired land uses from our literature review. Most paired land uses were grassland-cropland, but some were cropland-irrigated cropland, forest-cropland, natural-degraded comparisons. Open symbols represent our new field data from the Southern Great Plains, limited to 2 m depth for comparison with the literature review data. Linear regression shown is significant to P<0.0001 (n=84), and does not include data from our field observations (n=9). Most of the studies in the literature review came from cooler climates.

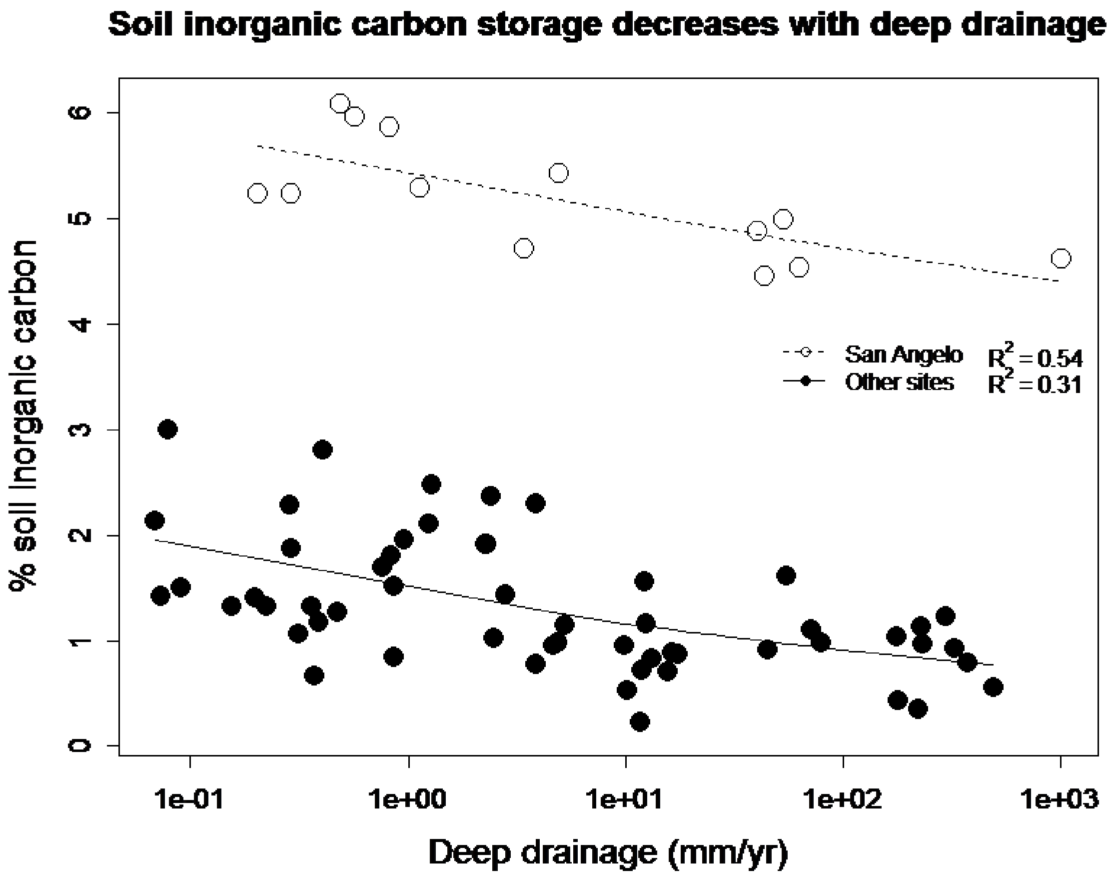

Figure 4.4: Soil inorganic carbon (SIC) storages to 4.8 m depth and deep drainage rates from corresponding soil cores. SIC decreased significantly with higher soil water flux. Separate regression for San Angelo data was done due to the higher SIC storage observed there than the rest of the sites. Though we pooled the rest of the data for this analysis, all correlations within each site were also statistically significant.

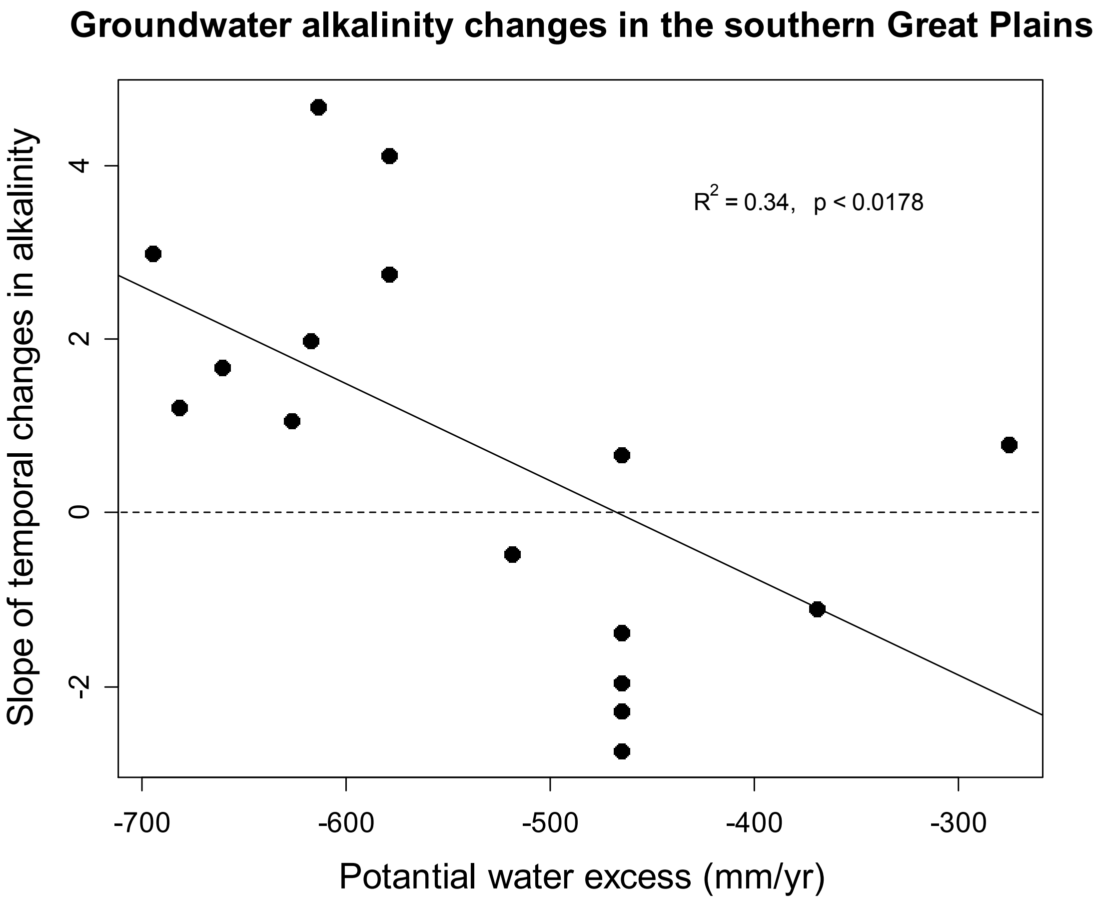

Figure 4.5: Slopes of temporal trends in alkalinity vs. potential water excess, an index of humidity (Precipitation – Potential Evapotranspiration). The data includes shallow ground water wells (>60m) monitored by USGS in Kansas and Texas with enough data collected for minimum of 20-year trend analysis. Only the trends significant to P<0.05 were used in this analysis. While most wells saw increases in alkalinity levels, wells in humid sites reported significant decreases.

5. Concluding remarks

Although recent scientific and policy interests on ecosystem services have expanded the knowledge and conceptual frameworks of quantifying the services, we still lack predictive capacity for determining responses of most ecosystem services to natural and anthropogenic changes. In this dissertation, I quantified relationships between vegetation, abiotic environment, and two coupled ecosystem services—carbon sequestration and groundwater recharge—focusing on linkages between the two. By evaluating the ecosystem services in paired land-use plots in the field and in the literature, I was able to directly compare and distinguish general patterns of carbon sequestration and groundwater recharge under different vegetation types and abiotic conditions.

Significant gains and losses of recharge with conversion of natural vegetation to crops and forests, respectively, indicate that land-use changes will continue to alter recharge dynamics and vadose zone processes globally. Though absolute changes in water yield with land-use changes are likely larger in humid or sandy areas, proportionately large hydrological consequences await land-use changes in arid or clayey regions, indicating greater biological control on soil water fluxes under these conditions. Quantifying and predicting changes to water yield from natural and anthropogenic disturbances are necessary steps for sustainable and holistic management of water resources; our results presented in Chapter 2 highlight importance of land-use change for the soil and groundwater resources. The effects of vegetation on recharge presented here should also help test and improve recharge estimates in large-scale water balance and climate models.

Chapter 3 shows clear tradeoffs between changes in carbon sequestration and groundwater recharge as consequences of land-use changes. I make general recommendations based on the observed relationships between abiotic factors and the changes in carbon and water. For example, woody encroachment may be a more attractive option in more humid climates given the large carbon sequestration potential and low value of water. In clayey soils, small changes in recharge may not be able to offset concomitant large losses of carbon with grassland conversions, indicating that land-use changes in sandy soils may be comparatively less costly. However, negative net values of most land-use changes in the U.S. study system indicate that incorporation of economic values of ecosystem services will likely make these land uses less profitable. Even though woody encroachment was the least costly land-use change in the analysis, its marginally positive net value indicate that establishment of carbon markets may not result in large-scale conversion to woodland. Greater costs were associated with more humid climate and more clayey soils for rain-fed cultivation. Irrigated cultivation had the highest net cost, highlighting negative externalities for both carbon and water associated with the land use. The dominance of carbon over water in the values of their changes suggest that in the Argentinean study system where we witnessed large gains in biomass carbon with woody encroachment, carbon markets may lead to overexploitation of the groundwater resources. Such relationships may serve as references for regional land management decisions, with more holistic approaches furthering efforts at sustainable land management.

In Chapter 4, I make a case for cultivation of natural grassland and the accompanying increase in recharge as causes for loss of soil carbonates through dissolution and leaching. The observed loss of carbonates from the soil profile was deep and rapid, more so than expected from conventional understanding of soil carbonate cycling. Many new questions surfaced in light of these results regarding the fate of carbonates in the ground water, linkages between the hydrological and carbon cycles in the deep soil, and how soil and climate may affect proportions of dissolution and evolution of carbonate minerals in the soil.

This work stemmed from my motivation to quantify ecosystem services for policy decisions that will promote sustainable use of the environment. I focused on the relationships between ecosystem services and vegetation, particularly in context of land-use changes. Vegetation shifts can be accompanied by changes in a suite of ecosystem services, resulting in tradeoffs in multitudes of services and products. My contributions here will further efforts such as predicting response of ecosystem services to land-use changes and market responses to incorporation of ecosystem services and modeling recharge in global climate and vegetation models. They also highlight new exciting opportunities and questions such as identifying interactions in the terrestrial and aqueous carbon cycling and how soil and climate may affect them.

As demand and consumption of environmental goods and product continue to rise, we are under increasing pressure to consider ones' welfare to those next to them and to those of next generations. All the while, we have grown increasingly distanced from the natural world while becoming dependent more than ever on its shrinking remnants. Policy and social incentives can help determine optimal configuration and level of our use of the planet, but it must be based on real measurements of sound scientific nature. It is author's sincerest wish that this work may motivate a reader to reflect on his or her stewardship of the Earth.

Appendix A: Recharge estimates, site information and values used for our multiple regression analyses.

Reference	Lat., Long.[†]	Soil[‡]	Vegetation	Recharge[¶]	Precip.[¶]	Irrig.[¶]	*PET*[§]	*Amp.*[§]	*Phase*[§]	*Methods*[#]
Abdalla (2008)	11.1, 32.6	Sand	Scrubland	0.9	400		1930	154.2	4	Model
	11.1, 29.1	Clay	Scrubland	4	1025		1893	147.4	4	Model
	16.1, 34.6	Sand	Scrubland	7.3	130		2102	72.9	2	Model
Ahmed and Umar (2008)	29.4, 77.3	Clay	Cropland	205	668	550	1454	267.2	2	WTF
	29.4, 77.3	Clay	Cropland	280	668	800	1454	267.2	2	WTF
	29.4, 77.3	Clay	Cropland	300	668	800	1454	267.2	2	WTF
Allen (1981)	-31.8, 115.9	Sand	Scrubland	85	775		1661	155.5	4	WB
Allison and Hughes (1972)	-37.8, 140.8	Sand	Grassland	63	686		1132	86.3	5	T
	-37.8, 140.8	Sand	Woodland	13	686		1132	86.3	5	T
Allison and Hughes (1978)	-37.8, 140.8	Sand	Grassland	106	700		1132	86.3	5	T
	-37.8, 140.8	Sand	Grassland	114	700		1132	86.3	5	T
Allison and Hughes (1983)	-35.1, 142.1	Sand	Cropland	3.5	335		1379	15.5	5	T
	-35.1, 142.1	Sand	Woodland	0.07	335		1379	15.5	5	T
Allison *et al*. (1985)	-34.3, 139.6	Sand	Woodland	0.135	300		1374	14.8	3	T
Allison *et al*. (1990)	-36.3, 140.8	Clay	Cropland	2	500		1245	44	6	T
	-36.3, 140.8	Clay	Cropland	2	500		1245	44	6	T
	-34.3, 139.6	Sand	Cropland	13	300		1374	14.8	3	T
	-35.1, 140.3	Sandy loam	Cropland	25.2	370		1346	22	5	T
	-34.3, 139.6	Sand	Woodland	0.05	300		1374	14.8	3	T
	-35.1, 140.1	Sand	Woodland	0.05	340		1335	23.1	5	T

	-35.1, 141.9	Sand	Woodland	0.06	340		1373	15.7	5	T
	-35.1, 140.3	Sand	Woodland	0.07	370		1346	22	5	T
	-34.4, 140.1	Sandy loam	Woodland	0.07	270		1361	15.2	6	WB
	-35.1, 140.3	Sand	Woodland	0.64	370		1346	22	5	WB
	-34.4, 140.1	Sand	Woodland	1.3	270		1361	15.2	6	T
Al-Sagaby and Moallim (2001)	25.8, 42.9	Sand	NoVeg	1.8	133		2283	41	4	T
Amro *et al*. (2001)	29.8, 35.3	Sand	NoVeg	0.03	65		1768	10	5	T
	32.1, 36.1	Sandy silt	NoVeg	0.2	67		1504	51.6	5	T
	32.1, 36.1	Sandy silt	NoVeg	1.5	67		1504	51.6	5	T
	32.3, 35.9	Sand	NoVeg	8	480		1447	77.3	5	T
	32.3, 35.9	Sand	NoVeg	28	480		1447	77.3	5	T
Anderson *et al*. (1998)	-30.6, 116.1	Loam	Cropland	214	703		1740	75.2	4	WB
Andrews *et al*. (1997)	52.3, 0.4	Sand	Cropland	83	474		481	20.1	4	WTF
	52.3, 0.3	Clay loam	Cropland	104	455		481	20.4	1	WTF
Anuraga *et al*. (2006)	13.1, 78.3	Clay	Cropland	84	902		1530	160	4	Model
	12.9, 78.3	Clay	Cropland	90	902	410	1529	160.9	4	Model
	13.1, 78.3	Clay	Cropland	124	902	150	1530	160	4	Model
	13.1, 78.3	Sandy loam	Cropland	184	902		1530	160	4	Model
	13.1, 78.3	Sandy loam	Cropland	220	902	410	1530	160	4	Model
	13.1, 78.3	Sandy loam	Cropland	232	902	150	1530	160	4	Model
Athavale *et al*. (1980)	16.9, 78.6	Clay	Cropland	67	1100		1669	177.3	4	T
	16.9, 78.6	Clay	Cropland	73	1150		1669	177.3	4	T
	16.9, 78.6	Clay	Cropland	80	970		1669	177.3	4	T

	16.9, 78.6	Sandy clay loam	Cropland	83	1310	1669	177.3	4	T
	16.9, 78.6	Sandy loam	Cropland	83	1150	1669	177.3	4	T
	16.9, 78.6	Clay	Cropland	96.8	1310	1669	177.3	4	T
	16.9, 78.6	Sandy loam	Cropland	98	1200	1669	177.3	4	T
	16.9, 78.6	Sandy loam	Cropland	133	1310	1669	177.3	4	T
	16.9, 78.6	Sandy loam	Cropland	222	1430	1669	177.3	4	T
Babiker *et al*. (2005)	35.4, 136.9	Sand	Cropland	860	1915	895	195.1	2	WB
Beekman *et al*. (1996)	-22.1, 26.3	Sand	Scrubland	12.5	500	1408	83.8	0	T
Bekele *et al*. (2006)	-29.8, 115.6	Sand	Cropland	14.7	440	1869	109.5	4	T
	-29.8, 115.6	Sand	Cropland	35.7	440	1869	109.5	4	WTF
	-29.8, 115.6	Sand	Grassland	16.2	440	1612	109.5	4	T
	-29.8, 115.6	Sand	Grassland	35.9	440	1612	109.5	4	T
	-29.8, 115.6	Sand	Scrubland	9	440	1869	109.5	4	T
Bellot *et al*. (1999)	38.3, -0.6	Loam	Grassland	61.5	454	1136	55	2	Model
	38.3, -0.6	Loam	NoVeg	125	454	1136	55	2	Model
	38.3, -0.6	Loam	Scrubland	18.6	454	1136	55	2	Model
	38.3, -0.6	Loam	Woodland	9.6	454	1136	55	2	Model
Bent (2001)	42.4, -72.3	Fine sandy loam	Woodland	262	1248	855	22.5	4	Model
	42.4, -72.3	Fine sandy loam	Woodland	371	1169	855	22.5	4	Model
Beverly *et al*. (2005)	-37.3, 144.9	Sand	Grassland	113	651	1121	40.3	6	Model
Bird *et al*. (2004)	-37.8, 142.1	Clay loam	Cropland	36	695	1149	57	6	WB
	-37.8, 142.1	Clay loam	Grassland	18	695	1149	57	6	WB

Bredenkamp and Vandoolaeghe (1982)	-33.6, 18.4	Coarse sands	Scrubland	73.5	350	1236	73.2	5	Model
	-33.6, 18.4	Coarse sands	Scrubland	95	350	1236	73.2	5	WB
Butler and Verhagan (2001)	-27.1, 22.8	Sand	Grassland	1.8	337	1572	65.6	2	T
	-27.1, 22.8	Sand	Grassland	13	337	1572	65.6	2	T
Calder *et al*. (2003)	53.3, -1.1	Sand	Grassland	169	800	449	14.6	5	Model
	53.3, -1.1	Sand	Scrubland	156	800	449	14.6	5	Model
	53.3, -1.1	Sand	Woodland	30	643	449	14.6	5	T
	53.3, -1.1	Sand	Woodland	45.8	643	449	14.6	5	Model
	53.3, -1.1	Sand	Woodland	69	643	449	14.6	5	WB
	53.3, -1.1	Sand	Woodland	106	643	449	14.6	5	Model
	53.3, -1.1	Sand	Woodland	120	643	449	14.6	5	WB
Carbon *et al*. (1982)	-31.8, 115.9	Coarse sands	Grassland	173	800	1661	155.5	4	WB
	-31.8, 115.9	Coarse sands	Woodland	121	900	1661	155.5	4	WB, WTF, T
Carlson *et al*. (1988)	33.3, -99.3	Clay loam	Grassland	7	671	1610	70.9	2	Lysimeter
	33.3, -99.3	Clay loam	NoVeg	9.3	671	1610	70.9	2	Lysimeter
	33.3, -99.3	Clay loam	Woodland	3.3	671	1610	70.9	2	Lysimeter
Cherkauer *et al*. (2005)	43.3, -88.3	Sand	Cropland	123	1030	839	71.2	1	Baseflow
Cho *et al*. (2009)	37.3, -80.1	12	Woodland	27	1045	982	33.6	0	Model
Colville and Holmes (1972)	-37.6, 140.8	Sand	Grassland	82	700	1151	86.9	5	WTF
	-37.6, 140.8	Sand	Woodland	44	700	1151	86.9	5	WTF
Conrad *et al*. (2005)	-32.4, 18.8	Coarse sands	Cropland	15	275	1440	58.7	4	T
	-32.4, 18.8	Coarse sands	Scrubland	2	200	1440	58.7	4	T

Cook (1992)	-35.1, 140.1	Loamy sand	Cropland	9.8	340	1335	23.1	5	T
Cook and Kilty (1992)	-35.1, 140.1	Sand	Cropland	9	340	1335	23.1	5	EMI
Cook *et al*. (1989)	-34.6, 142.8	Sandy clay loam	Cropland	7	312	1421	15.9	4	T
	-34.6, 143.6	Sandy clay loam	Cropland	8.3	322	1378	13.8	6	T
	-35.1, 140.1	Sand	Grassland	2.7	340	1335	23.1	5	EMI
	-35.1, 140.1	Sand	Grassland	17.4	340	1335	23.1	5	T
	-35.1, 140.1	Sand	Woodland	0.05	340	1335	23.1	5	T
Cook *et al*. (1992a)	-34.4, 140.1	Sandy loam	Cropland	3	270	1361	15.2	6	T
Cook *et al*. (1992b)	15.6, -16.3	Sand	Cropland	15	356	1853	130.1	2	T
Cook *et al*. (1994)	-34.3, 139.6	Sand	Grassland	11	340	1374	14.8	3	T
	-35.1, 140.1	Sand	Grassland	13	340	1335	23.1	5	T
	-35.1, 140.1	Sand	Grassland	16	340	1335	23.1	5	T
	-35.1, 140.1	Sand	Woodland	0.1	260	1335	23.1	5	T
	-35.1, 140.1	Sand	Woodland	0.9	260	1335	23.1	5	T
Cook *et al*. (1998)	-12.6, 131.1	Clay	Woodland	200	1720	1931	372	2	T
Cook *et al*. (2004)	-34.3, 140.6	Sand	Grassland	2.7	260	1373	14	3	T
	-34.3, 140.6	Sand	Grassland	4.9	260	1373	14	3	Model
	-34.3, 140.6	Sands	Woodland	0.1	260	1373	14	3	T
Crosbie *et al*. (2007)	-34.6, 148.8	Clay	Grassland	5.2	613	1153	25.3	3	WTF
	-34.6, 148.8	Clay	Grassland	48.4	613	1153	25.3	3	WTF
Dams *et al*. (2008)	51.3, 4.8	Sand	Cropland	292	839	577	26.9	4	Model
Daniel (1999)	35.6, -98.1	Loam	Cropland	93.8	743	1424	101	2	WTF

	35.6, -98.1	Loam	Grassland	63.9	743	1424	101	2	WTF
Datta *et al*. (1980)	23.6, 73.3	Sandy loam	Cropland	34	852	1724	307.7	2	T
	23.1, 72.6	Sandy loam	Cropland	35.6	648	1718	276.1	2	T
	23.1, 72.6	Sandy loam	Cropland	58.5	1014	1718	276.1	2	T
	23.4, 72.4	Sandy loam	Cropland	70.9	1357	1754	256.5	2	T
	23.8, 73.1	Sandy loam	Cropland	87	1145	1758	301.1	2	T
	23.4, 72.4	Sandy loam	Cropland	144	1682	1754	256.5	2	T
	23.1, 73.1	Sandy loam	Cropland	184	1411	1731	325.6	2	T
De Vries *et al*. (2000)	-24.8, 25.3	Sand	Scrubland	0.9	325	1376	99.6	0	T
	-24.8, 25.3	Sand	Scrubland	1	350	1376	99.6	0	T
	-24.1, 25.3	Sand	Scrubland	3	420	1372	88.2	0	T
	-23.8, 25.1	Sand	Scrubland	5	450	1396	81.1	0	T
Deans *et al*. (2005)	15.6, -16.3	Sand	Cropland	15	356	1853	130.1	2	T
Di and Cameron (2002)	-43.8, 171.8	Silt loam	Cropland	370	650	681	21.9	5	
Dolling *et al*. (2007)	-29.9, 116.6	Sand	Cropland	30	335	1732	53.1	5	Model
	-33.9, 117.1	Sand	Cropland	115	496	1295	77	5	Model
Dripps and BradBury (2007)	43.1, -89.6	Silt loam	Cropland	256	834	824	75.6	1	WB
	43.1, -89.6	Silt loam	Cropland	290	834	824	75.6	1	WB
	46.1, -89.8	Clay	Grassland	279	790	688	84.9	1	WTF
	46.1, -89.8	Clay	Grassland	287	790	688	84.9	1	WB
	46.1, -89.8	Clay	Woodland	130	790	688	84.9	1	WTF
	46.1, -89.8	Clay	Woodland	175	790	688	84.9	1	WB
	46.1, -89.8	Clay	Woodland	176	790	688	84.9	1	WTF

	46.1, -89.8	Clay	Woodland	268	790		688	84.9	1	WB
Duffkova (2002)	49.3, 14.8	Sandy loam	Grassland	20.6	528		599	52.5	1	Lysimeter
Dunin *et al*. (1999)	-35.4, 147.6	850	Cropland	15	611		1079	38.7	5	WB
	-35.4, 147.6	850	Cropland	84	611		1079	38.7	5	WB
	-35.4, 147.6	850	Cropland	89	611		1079	38.7	5	WB
	-35.4, 147.6	Sandy clay loam	Cropland	185	611		1079	38.7	5	WB
	-35.4, 147.6	Sandy clay loam	Grassland	25	611		1079	38.7	5	WB
	-35.4, 147.6	850	Grassland	2	611		1079	38.7	5	WB
Dyck and de Jong (2003)	51.9, -107.3	Silty loam	Cropland	3	321		719	48.7	0	T
Edmunds (2001)	34.8, 32.9	Sand	Grassland	52.5	420		1364	104.1	4	T
	34.8, 32.9	Sand	Grassland	55.5	420		1364	104.1	4	T
Edmunds and Gaye (1994)	15.9, -16.3	Clay	Cropland	2.69	290		1858	107.3	1	T
	15.8, -16.3	Sand	Cropland	14.9	290		1853	118.4	2	T
Edmunds *et al*. (2002)	13.1, 10.1	Sand	NoVeg	35.3	314		2286	168.4	3	T
Fachi *et al*. (2005)	45.1, 9.6	Coarse sands	Grassland	491	800	512	678	55.9	3	Model
Favreau (2009)	13.6, 2.8	Sand	Cropland	25	557		2160	171.6	3	WTF
	13.6, 2.8	Sand	Scrubland	2	557		2160	171.6	3	Model
Favreau *et al*. (2002)	13.4, 2.8	Sand	Cropland	35	567		2152	175.1	3	WTF
	13.4, 2.8	Sand	Scrubland	3	567		2152	175.1	3	T
Fayer *et al*. (1996)	46.6, -119.4	Loamy sand	Grassland	1.2	159		1083	18.5	5	T
	46.6, -119.4	Sandy loam	Grassland	5.1	159		1083	18.5	5	T

	46.6, -119.4	Sandy loam	Grassland	25.4	159		1083	18.5	5	WB
	46.6, -119.4	Coarse sands	NoVeg	55.4	159		1083	18.5	5	Lysimeter
	46.6, -119.4	Gravel	NoVeg	86.7	184		1083	18.5	5	Lysimeter
	46.6, -119.4	Gravel	NoVeg	300	480		1083	18.5	5	Lysimeter
	46.6, -119.4	Loamy sand	Scrubland	0.02	159		1083	18.5	5	T
	46.6, -119.4	Silt loam	Scrubland	0.05	159		1083	18.5	5	T
	46.6, -119.4	Loamy sand	Scrubland	2	159		1083	18.5	5	T
	46.6, -119.4	Silt loam	Scrubland	2.75	159		1083	18.5	5	T
Fillery and Poulter (2006)	-30.8, 116.6	Loamy sand	Cropland	53	495		1643	62.7	5	WB
Finch (1998)	51.6, -1.1	Sandy clay loam	Cropland	290	587		473	25.3	5	WB
	51.6, -1.1	Sandy clay loam	Grassland	176	587		473	25.3	5	WB
	51.6, -1.1	Sandy clay loam	Woodland	96	587		473	25.3	5	WB
Fisher and Healy (2008)	46.3, -119.9	Silty clay	Cropland	119	187	744	1068	22.5	5	Lysimeter, WB
	39.3, -76.1	Fine sandy loam	Cropland	315	981		1045	27.6	1	Lysimeter, WB
	37.3, -120.8	Sand	Cropland	423	270	1200	1384	48.4	6	Lysimeter, WB
	39.8, -85.8	Silty clay loam	Cropland	475	906		955	68.4	0	Lysimeter, WB
Fouty (1989)	36.9, -116.8	Loamy sand	Scrubland	0.23	104		1754	12.2	5	T
Gates *et al*. (2008)	39.9, 101.9	Sand	Grassland	1.5	84		1010	17.5	1	T

Gaye Edmunds (1996)	15.8, -16.4	Sand	Cropland	24	290		1843	118.3	2	T
	15.8, -16.4	Sand	Cropland	31.5	290		1843	118.3	2	T
Gee and Fayer (1994)	32.6, -106.4	Loamy fine sand	NoVeg	87	338		1704	50.8	1	Lysimeter, WB
Gee *et al*. (1993)	46.6, -119.4	Sand	NoVeg	71.1	172		1083	18.5	5	Lysimeter, WB
	46.6, -119.4	Sand	NoVeg	300	480		1083	18.5	5	Lysimeter, WB
George and Frantom (1988)	-31.6, 118.3	Sandy clay	Woodland	0.1	328		1501	42	5	T
	-31.6, 118.3	Sandy clay	Woodland	1.5	328		1501	42	5	T
Gieske (1992)	-24.4, 25.6	Sand	Scrubland	10	492		1357	94.1	0	T
Gieske *et al*. (1995)	-24.3, 25.3	Sand	Scrubland	9	425		1372	91.6	0	T
	-24.3, 25.3	Sand	Scrubland	15	425		1372	91.6	0	T
Goni and Edmunds (2001)	13.6, 13.4	Fine sands	Scrubland	7	389		2300	132.7	3	T
	12.1, 12.8	Fine sands	Scrubland	22.5	389		2184	191.2	3	T
Goodrich *et al*. (2004)	31.8, -110.8	Silty clay	Scrubland	3	324		1499	99.9	0	T
Green *et al*. (2008)	41.6, -96.6	Silt loam	Cropland	159	720	203	1024	94.4	1	WTF
	41.6, -96.6	Loamy sand	Grassland	48	720		1024	94.4	1	WTF
Gregory *et al*. (1992)	-32.1, 117.1	Sandy loam	Cropland	6.5	380		1469	69.5	5	WB
Gupta and Sharma (1984)	22.9, 76.6	Sand	Cropland	67	750		1690	348.9	3	T
	22.9, 76.6	Sand	Cropland	81	894		1690	348.9	3	T
	22.9, 76.6	Sand	Cropland	94	821		1690	348.9	3	T
Hadas *et al*. (1999)	31.3, 34.6	360	Cropland	70	210	525	1452	57.6	5	WB, T
	31.9, 34.8	680	Cropland	73.7	567		1311	136.8	5	WB, T

	32.1, 34.8	330	Cropland	81.6	544	266	1290	141.1	4	WB, T
	32.3, 34.9	680	Cropland	95.9	588	150	1307	154.6	4	WB, T
Halm *et al*. (2002)	-7.1, -41.8	Sand	Cropland	14.5	700		1835	163.6	5	WB
	-7.1, -41.8	Sand	Scrubland	6.5	700		1835	163.6	5	WB
Hatton and Nulsen (1999)	-35.4, 147.6	Sandy clay loam	Grassland	3	611		1079	38.7	5	Model
	-35.4, 147.6	Sandy clay loam	Grassland	134	611		1079	38.7	5	Model
	-35.4, 147.6	Sandy clay loam	Woodland	0	611		1079	38.7	5	Model
Heilweil *et al*. (2006)	37.1, -113.3	Loam	Scrubland	0.3	210		1639	34.7	4	T
	37.1, -113.3	Loam	Scrubland	4	210		1639	34.7	4	T
	37.1, -113.3	Loam	Scrubland	6.8	210		1639	34.7	4	T
	37.1, -113.3	Loam	Scrubland	10	210		1639	34.7	4	T
Heng *et al*. (2001)	-35.4, 147.6	Clay	Grassland	47.6	650		1079	38.7	5	WB
Holmes and Colville (1968)	-37.8, 140.8	Sand	Grassland	120	700		1132	86.3	5	Lysimeter
Holmes and Colville (1970a)	-37.8, 140.8	Sand	Grassland	63	600		1132	86.3	5	WB
Holmes and Colville (1970b)	-37.9, 140.9	Sand	Woodland	0	600		1147	84.2	5	WB
Holmstead *et al*. (1988)	29.1, -99.9	Loam	Grassland	0	273		1565	58.9	2	Lysimeter
	29.1, -99.9	Loam	Grassland	1.2	736		1565	58.9	2	Lysimeter
	29.1, -99.9	Loam	NoVeg	10.7	273		1565	58.9	2	Lysimeter
	29.1, -99.9	Loam	NoVeg	29.9	736		1565	58.9	2	Lysimeter
Houston (1982)	-14.4, 28.4	1800	NoVeg	281	937		1448	240.3	3	Baseflow

	-14.4, 28.4	1800	Woodland	80	937	1448	240.3	3	Baseflow
Howard and Karundu (1992)	0.1, 30.8	Loam	Cropland	66	869	1235	105.4	4	WB
	0.1, 30.8	Loam	Grassland	33.5	869	1235	105.4	4	WB
	0.1, 30.8	Loam	NoVeg	81	869	1235	105.4	4	WB
	0.1, 30.8	Loam	Woodland	0	869	1235	105.4	4	WB
Huang and Gallichand (2006)	35.3, 107.8	Silty clay loam	Cropland	18.3	545	818	109.9	0	Model
Hughes *et al*. (1988)	-35.1, 140.1	Sandy loam	Cropland	16.5	340	1335	23.1	5	T
Hume (1997)	-35.8, 150.1	Coarse sands	Woodland	200	800	1163	63.8	1	
Hussein (2001)	31.1, 33.8	Sand	NoVeg	18	300	1405	25.6	4	T
	31.1, 33.8	Sand	NoVeg	24	300	1405	25.6	4	T
Jackson and Rushton (1987)	50.1, 10.1	Boulder clay	Cropland	24	521	540	38.5	1	WB
Jipp *et al*. (1998)	-2.9, -47.6	Clay	Grassland	287	1672	1321	323	4	WB
	-2.9, -47.6	Clay	Woodland	141	1672	1321	323	4	WB
	-2.9, -47.6	Clay	Woodland	187	1672	1321	323	4	WB
Johnston (1987a)	-33.4, 115.9	Clay	Woodland	28.1	1220	1504	178	5	T
	-33.4, 115.9	Clay	Woodland	75	1220	1504	178	5	T
Johnston (1987b)	-33.3, 116.4	Clay	Woodland	2.45	800	1423	138.9	5	T
	-33.4, 115.9	Sand	Woodland	26.5	1250	1504	178	5	T
Jolly (1992)	-32.3, 18.4	Coarse sands	Scrubland	23.5	196	1398	40.7	4	WTF
Jolly *et al*. (1989)	-35.1, 140.3	Sand	Cropland	45	370	1346	22	5	T
	-35.1, 140.3	Sand	Woodland	0.8	370	1346	22	5	T
Joshi (1997)	52.1, -106.1	Silt	Cropland	12	371	699	51.3	0	T
	52.1, -106.1	Silt	Grassland	1	371	699	51.3	0	T

Julien *et al*. (1988)	33.3, -99.3	Fine sandy loam	Grassland	0	723		1610	70.9	2	Lysimeter
	33.3, -99.3	Fine sandy loam	Grassland	0	811		1610	70.9	2	Lysimeter
	33.3, -99.3	Fine sandy loam	Grassland	0	852		1610	70.9	2	Lysimeter
	33.3, -99.3	Fine sandy loam	NoVeg	10.8	837		1610	70.9	2	Lysimeter
	33.3, -99.3	Fine sandy loam	Woodland	0	678		1610	70.9	2	Lysimeter
Kendy *et al*. (2003)	37.9, 114.8	Loam	Cropland	66.3	367	81	1031	150.6	1	WB
	37.9, 114.8	Loam	Cropland	105	367	301	1031	150.6	1	WB
	37.9, 114.8	Loam	Cropland	140	367	371	1031	150.6	1	WB
	37.9, 114.8	Loam	Cropland	174	367	460	1031	150.6	1	WB
Kendy *et al*. (2004)	37.9, 114.8	Loam	Cropland	200	461		1031	150.6	1	WB
	37.9, 114.8	Loam	Cropland	690	461	900	1031	150.6	1	WB
	37.9, 114.8	Loam	Cropland	1300	461	1500	1031	150.6	1	WB
Kennett-Smith *et al*. (1990)	-34.3, 141.3	Sandy clay loam	Cropland	4	310		1387	11.5	3	T, WB
	-34.3, 141.3	Loamy sand	Cropland	7.5	310		1387	11.5	3	T, WB
	-34.6, 142.8	Loamy sand	Cropland	13.6	312		1421	15.9	4	T, WB
	-34.6, 143.6	Sandy clay loam	Cropland	18	322		1378	13.8	6	T, WB
Kennett-Smith *et al*. (1992a)	-37.6, 143.9	Clay	Cropland	3	430		1108	56.7	6	T, WB
Kennett-Smith *et al*. (1992b)	-33.4, 142.6	Loamy sand	Grassland	0.4	255		1493	9.4	3	T, WB

Kennett-Smith *et al*. (1993)	-35.8, 141.4	Clay	Grassland	3.5	530		1294	27.8	5	T, WB
Kennett-Smith *et al*. (1994)	-35.1, 141.9	Sandy clay loam	Cropland	9	340		1373	15.7	5	T
Kienzle and Schulze (1992)	-27.4, 32.6	Sand	Woodland	179	850		1337	103.2	0	WB
Knoche *et al*. (2002)	51.8, 13.6	Sand	Woodland	82	652		616	36.7	1	Model
Krajenbrink *et al*. (1988)	52.3, 5.6	Coarse sands	Cropland	305	854		518	34.3	4	T
	52.3, 5.6	Coarse sands	Grassland	305	854		518	34.3	4	T
	52.3, 5.6	Coarse sands	Woodland	101	854		518	34.3	4	T
Külls (2000)	-24.3, 29.9	Sand	Scrubland	11.5	465		1341	98.6	1	T
Ladekarl *et al*. (2005)	56.4, 8.9	Sand	Scrubland	733	1077		450	49.4	3	T
	56.4, 9.4	Sand	Woodland	390	875		445	40.2	4	T
Larsen *et al*. (2002)	-19.9, 28.3	Sand	Scrubland	25	550		1528	126	2	T
Leaney and Allison (1986)	-34.1, 139.9	Sand	Woodland	0.15	275		1394	13.9	3	T
	-34.1, 139.9	Sand	Woodland	0.25	275		1394	13.9	3	T
Leaney and Herczeg (1995)	-36.3, 140.8	Clay	Cropland	1.1	545		1245	44	6	T
	-36.3, 140.8	Clay	Cropland	10	545	450	1245	44	6	T
	-36.3, 140.8	Sand	Cropland	60	545		1245	44	6	T
	-36.3, 140.8	Clay	Woodland	0.5	545		1245	44	6	T
	-36.3, 140.8	Sand	Woodland	0.5	545		1245	44	6	T
Leaney and Herczeg (1999)	-35.3, 140.8	Clay	Grassland	12	375	640	1346	23.3	5	T
	-35.3, 140.9	Clay	Woodland	0.3	440		1345	21.5	5	T
	-36.6, 141.3	Sand	Woodland	1.5	450		1216	46.3	6	T
Leduc *et al*. (2001)	13.6, 2.6	Sand	Scrubland	3	565		2162	174.9	3	T

	13.6, 2.6	Sand	Scrubland	6	565		2162	174.9	3	T
	13.6, 2.6	Sand	Scrubland	20	565		2162	174.9	3	WTF
Li *et al*. (2005)	36.1, 140.1	Loam	Grassland	392	1194		781	130	2	WB
Lin and Wei (2001)	42.9, 118.9	Silt loam	NoVeg	47	360		899	118.3	0	T
	37.8, 113.8	Silt loam	NoVeg	68	550		931	146.8	0	T
	42.9, 118.9	Silt loam	NoVeg	85	360		899	118.3	0	T
	37.8, 113.8	Silt loam	NoVeg	288	550		931	146.8	0	T
Loh and Stokes (1981)	-32.9, 121.6	Sand	Cropland	15	390		1462	23.5	5	T
	-32.9, 117.6	Sand	Cropland	19	410		1331	61.4	5	WTF
	-31.8, 116.4	Sand	Cropland	30	590		1570	125.4	4	WTF
	-33.3, 116.4	Sand	Cropland	40	750		1423	138.9	5	WTF
	-33.3, 116.6	Sand	Cropland	55	650		1396	115.7	5	WTF
	-33.3, 116.4	Sand	Cropland	60	725		1423	138.9	5	WTF
	-33.4, 115.9	Sand	Cropland	100	1150		1504	178	5	WTF
	-31.8, 116.4	Clay	Grassland	24	590		1570	125.4	4	WTF
	-33.4, 115.9	Sand	Woodland	10	1250		1504	178	5	WTF
Marechal *et al*. (2006)	17.4, 78.4	Clay	Cropland	114	613	165	1704	180.5	4	WTF
Marechal *et al*. (2009)	11.8, 76.4	Clay	Woodland	75	1273		1386	501.9	3	WTF, T
McDowall *et al*. (2003)	-33.4, 121.9	Sand	Grassland	55.3	522		1448	43.8	5	WB
McMahon *et al*. (2003)	37.8, -100.8	Sand	Cropland	53	487	675	1419	68.4	0	T
	37.3, -101.8	Loamy fine sand	Grassland	5.1	453		1464	57.7	2	T
McMahon *et al*. (2006)	33.6, -102.8	Loam	Cropland	17	420	585	1627	57.6	1	T
	33.8, -102.8	Loam	Cropland	24.5	440	450	1622	58	1	T

	33.6, -102.8	Loam	Cropland	32	420	433	1627	57.6	1	T
	33.8, -102.8	Sandy loam	Cropland	39	420	593	1622	58	1	T
	33.8, -102.8	Loamy sand	Cropland	54	420	638	1622	58	1	T
	33.8, -102.8	Sandy loam	Cropland	102	420	330	1622	58	1	T
	33.8, -102.8	Sandy loam	Cropland	111	420	540	1622	58	1	T
	34.1, -102.8	Loamy sand	Grassland	0.2	420		1595	60	1	T
	37.3, -101.8	Loamy sand	Grassland	5	453		1464	57.7	2	T
	40.6, -101.8	Sand	Grassland	70	500		1191	75	2	T
Mileham *et al*. (2008)	-0.9, 30.1	Sandy loam	Cropland	104	1190		1126	99.5	4	WB
Milroy *et al*. (2008)	-29.6, 115.8	Sand	Cropland	25.1	324		1969	74.1	4	Model
	-29.6, 115.8	Sand	Cropland	37.9	356		1900	74.1	4	Model
	-29.6, 115.8	Sand	Cropland	40.6	387		1800	74.1	4	Model
	-29.6, 115.8	Sand	Cropland	45	339		1969	74.1	4	Model
	-29.6, 115.8	Sand	Cropland	54.3	409		1700	74.1	4	Model
	-29.6, 115.8	Sand	Cropland	83.1	461		1622	74.1	4	Model
Monirul Islam (2005)	24.8, 88.6	Clay	Cropland	153	1442	207	1195	360.3	2	WB
Muller and Bolte (2009)	52.6, 13.4	Sand	Grassland	285	620		593	36.6	1	Lysimeter
	52.6, 13.4	Sand	Woodland	74.4	620		593	36.6	1	Lysimeter
	52.6, 13.4	Sand	Woodland	80.6	620		593	36.6	1	Lysimeter
	52.6, 13.4	Sand	Woodland	124	620		593	36.6	1	Lysimeter
Navada *et al*. (2001)	24.9, 71.1	Fine sands	Cropland	12	240		1872	106.4	2	T
	24.9, 71.1	Fine sands	Cropland	14.5	240		1872	106.4	2	T
	24.9, 71.1	Fine sands	Cropland	18	240		1872	106.4	2	T

	25.4, 71.1	Fine sands	Cropland	20	240	1836	81.3	2	T
Newman *et al*. (1997)	35.8, -106.3	Loam	Grassland	1	470	1444	48.8	1	T
	35.8, -106.3	Fine sandy loam	Woodland	0.45	510	1444	48.8	1	T
	35.8, -106.3	Loam	Woodland	0.8	470	1444	48.8	1	T
Nichols and Verry (2001)	47.6, -93.4	Fine sandy loam	Woodland	109	784	725	89.9	0	WB
O'connell *et al*. (2003)	-35.1, 141.9	Sandy loam	Cropland	5.3	356	1373	15.7	5	Lysimeter
Ojeda (2001)	28.4, -110.8	Sand	Scrubland	0.11	320	1737	92.5	1	T
	28.4, -110.8	Sand	Scrubland	0.16	320	1737	92.5	1	T
	31.6, -106.9	Sand	Scrubland	0.24	230	1780	39.4	0	T
Pakrou and Dillon (2000)	-37.8, 140.8	Silt loam	Cropland	129	750	1132	86.3	5	Lysimeter
	-37.8, 140.8	Silt loam	Cropland	163	750	1132	86.3	5	Lysimeter
Paydar and Gallant (2008)	-35.8, 146.8	360	Cropland	93.8	546	1071	33.1	5	Model
	-35.8, 146.8	360	Grassland	17.6	546	1071	33.1	5	Model
Peck and Hurle (1973)	-32.4, 116.8	Sand	Grassland	24	490	1468	98.7	5	Baseflow
	-32.8, 116.8	Sand	Grassland	26	730	1434	95.5	5	Baseflow
	-33.1, 116.9	Sand	Grassland	37	500	1373	84.4	5	Baseflow
	-33.3, 116.6	Sand	Grassland	60	820	1396	115.7	5	Baseflow
	-31.8, 116.3	Sand	Grassland	61	880	1597	149.7	4	Baseflow
	-31.4, 116.1	Sand	Grassland	78	910	1647	125.7	4	Baseflow
	-32.4, 116.8	Sand	Woodland	0.82	490	1468	98.7	5	WB
	-33.1, 116.9	Sand	Woodland	1.2	500	1373	84.4	5	WB
	-33.3, 116.6	Sand	Woodland	1.7	820	1396	115.7	5	WB

	-32.8, 116.8	Sand	Woodland	1.9	730	1434	95.5	5	WB
	-31.8, 116.3	Sand	Woodland	3.9	880	1597	149.7	4	WB
	-31.4, 116.1	Sand	Woodland	8	910	1647	125.7	4	WB
	-31.6, 116.3	Sand	Woodland	13.4	660	1620	126.6	4	T
	-32.9, 116.3	Sand	Woodland	24.2	1100	1470	181.6	5	T
	-33.3, 116.3	Sand	Woodland	33.4	870	1456	162.2	5	T
	-32.8, 116.1	Sand	Woodland	106	1350	1517	229.1	5	T
	-32.3, 116.1	Sand	Woodland	134	1147	1546	217.7	4	T
	-32.3, 116.1	Sand	Woodland	157	1100	1559	217.7	4	T
Peck *et al*. (1981)	-33.4, 116.1	Sand	Woodland	0.69	1150	1457	169.7	5	T, Model
	-33.4, 116.1	Sand	Woodland	8	800	1457	169.7	5	T, Model
	-33.4, 116.1	Sand	Woodland	104	1300	1457	169.7	5	T, Model
	-33.4, 116.1	Sand	Woodland	150	1150	1457	169.7	5	T, Model
Pracilio *et al*. (2003)	-31.3, 117.6	Loamy sand	Cropland	12	336	1541	45	5	Model
	-31.3, 117.6	Loamy sand	Cropland	32	336	1541	45	5	Model
	-31.3, 117.6	Sand	Cropland	53	336	1541	45	5	Model
Prych (1998)	46.6, -119.4	Loam	Grassland	1.2	160	1083	18.5	5	T
	46.6, -119.4	Loam	Grassland	5.1	160	1083	18.5	5	T
	46.6, -119.4	Silt loam	Scrubland	0.06	160	1083	18.5	5	T
	46.6, -119.4	Loam	Scrubland	0.15	160	1083	18.5	5	T
	46.6, -119.4	Loam	Scrubland	2.6	160	1083	18.5	5	T
Radford *et al*. (2009)	-24.8, 150.1	Clay	Cropland	1.6	700	1502	72.1	1	T
	-23.9, 150.3	Clay	Cropland	2	659	1600	93.9	0	T
	-23.1, 148.1	Clay	Cropland	7.4	597	1638	93.8	1	T

	-24.3, 149.8	Clay	Cropland	8.9	632	1548	82.8	1	T
	-23.9, 148.4	Clay	Cropland	16.1	600	1596	94.7	1	T
	-22.9, 148.9	Clay	Cropland	18	580	1621	96.8	0	T
	-24.3, 150.4	Clay	Cropland	27.5	639	1579	88.2	1	T
	-22.9, 148.9	Clay	Woodland	0.2	580	1621	96.8	0	T
	-23.9, 148.4	Clay	Woodland	0.2	600	1596	94.7	1	T
	-23.9, 150.3	Clay	Woodland	0.2	659	1600	93.9	0	T
	-24.3, 149.8	Crackling clay	Woodland	0.3	632	1548	82.8	1	T
	-24.3, 149.8	Crackling clay	Woodland	0.3	638	1548	82.8	1	T
	-24.8, 150.1	Crackling clay	Woodland	0.3	700	1502	72.1	1	T
	-24.3, 150.4	Clay	Woodland	0.3	639	1579	88.2	1	T
	-23.1, 148.1	Crackling clay	Woodland	1.7	597	1638	93.8	1	T
Ragab *et al*. (1997)	52.3, -2.6	Loamy sand	Grassland	68	625	444	25.9	5	WB
	52.3, 0.3	4	Grassland	91	550	481	20.4	1	WB
	50.8, -3.3	Loamy sand	Grassland	153	738	477	56	6	WB
	52.3, 0.3	4	Grassland	165	550	481	20.4	1	Lysimeter
	51.1, -1.3	Silty clay loam	Grassland	213	771	469	38.9	5	WB
Rangarajan *et al*. (2009)	8.8, 78.1	Sandy loam	Cropland	16.3	582	1514	186.6	5	T
	8.8, 78.1	Sand	Cropland	47.6	582	1514	186.6	5	T
	8.8, 78.1	Sandy loam	Cropland	60	582	1514	186.6	5	T
	8.8, 78.1	Clay	Cropland	70.2	582	1514	186.6	5	T

	8.8, 78.1	Sand	Cropland	82.3	582	1514	186.6	5	T
Renard *et al*. (1993)	31.8, -110.8	Loam	Scrubland	0.2	303	1499	99.9	0	Model
Renger and Wessolek (1990)	53.1, 10.8	Sand	Cropland	230	615	519	32.5	0	
	51.4, 9.3	deposits of glacial till	Cropland	232	687	516	34.9	1	
Renger *et al*. (1986)	52.3, 9.8	Fine sands	Cropland	225	655	518	34	1	WB, Model
	52.3, 9.8	Fine sands	Grassland	190	655	518	34	1	WB, Model
	52.3, 9.8	Fine sands	Woodland	110	655	518	34	1	WB, Model
Richardson and Narayan (1995)	-34.4, 135.9	Sand	Cropland	40	550	1408	66.9	5	WTF, WB
	-34.4, 135.9	Sand	Grassland	10	550	1408	66.9	5	Model
Ridley *et al*. (1997)	-36.1, 146.6	Sandy clay loam	Grassland	74.5	693	1056	42.2	5	WB
	-36.1, 146.6	Sandy clay loam	Grassland	83	693	1056	42.2	5	WB
	-36.1, 146.6	Sandy clay loam	NoVeg	83	693	1056	42.2	5	WB
	-36.1, 146.6	Sandy clay loam	NoVeg	142	693	1056	42.2	5	WB
	-36.1, 146.6	fine sandy clay loam	Cropland	36.5	600	1056	42.2	5	WB
	-36.1, 146.6	fine sandy clay loam	Cropland	51.5	600	1056	42.2	5	Model
	-36.1, 146.6	fine sandy clay loam	Grassland	5.5	600	1056	42.2	5	WB

	-36.1, 146.6	fine sandy clay loam	Grassland	6.8	600		1056	42.2	5	Model
Roberts and Rosier (2006)	51.1, -1.3	Silty clay	Grassland	207	986		469	38.9	5	WB
	51.1, -1.3	Silty clay	Woodland	300	1004		469	38.9	5	WB
Rodvang *et al*. (2004)	49.9, -112.8	fine sandy clay loam	Cropland	11.6	400	300	851	50.4	1	T
	49.9, -112.8	Coarse sands	Cropland	29.7	400	350	851	50.4	1	T
	49.9, -112.8	fine sandy clay loam	Cropland	34.7	400	300	851	50.4	1	WTF
	49.9, -112.8	Coarse sands	Cropland	59.7	400	350	851	50.4	1	WTF
	49.9, -112.8	Coarse sands	Cropland	117	400	350	851	50.4	1	T
	49.9, -112.8	Coarse sands	Cropland	170	400	440	851	50.4	1	T
	49.9, -112.8	Coarse sands	Grassland	42	400	440	851	50.4	1	T
Sami and Hughes (1996)	-32.8, 26.1	Loam	Grassland	5.2	460		1342	50.7	2	T
	-32.8, 26.1	Loam	Grassland	5.8	460		1342	50.7	2	Model
Santoni *et al*. (2010)	-33.6, -65.8	Sandy loam	Cropland	5.3	518		1317	86	0	T
	-33.6, -65.8	Sandy loam	Cropland	6.9	502		1317	86	0	T
	-33.6, -65.8	Sandy loam	Cropland	7.9	502		1317	86	0	T
	-33.8, -65.8	Sandy loam	Cropland	9.6	542		1294	82.7	0	T
	-33.4, -65.9	Sandy loam	Cropland	10.4	538		1383	90	0	T
	-33.6, -65.8	Sandy loam	Cropland	10.8	518		1317	86	0	T
	-33.4, -65.9	Sandy loam	Cropland	13.2	538		1383	90	0	T
	-33.8, -65.8	Sandy loam	Cropland	128	542		1294	82.7	0	T
	-33.4, -66.6	Sandy loam	Woodland	0.02	447		1476	84.6	0	T
	-33.4, -65.9	Sandy loam	Woodland	0.04	538		1383	90	0	T

	-33.6, -65.8	Sandy loam	Woodland	0.05	502	1317	86	0	T
	-33.6, -65.8	Sandy loam	Woodland	0.14	518	1317	86	0	T
	-33.8, -65.8	Sandy loam	Woodland	0.33	542	1294	82.7	0	T
Scanlon (1991)	31.4, -105.8	Silt loam	Scrubland	0.07	280	1766	43.7	1	T
Scanlon and Goldsmith, 1997 (1997)	35.3, -101.8	Silty clay loam	Grassland	0.62	500	1571	71.5	1	T
Scanlon *et al*. (1999)	31.1, -105.3	Clay	Grassland	0.02	320	1737	48.4	1	T
	31.1, -105.3	Clay loam	Grassland	0.05	320	1737	48.4	1	T
Scanlon *et al*. (2005)	32.9, -102.1	Sandy loam	Cropland	19.5	457	1670	53.6	2	T
	32.9, -102.1	Sand	Cropland	24	457	1670	53.6	2	T
	32.9, -102.1	Sand	Grassland	2	457	1670	53.6	2	T
	36.8, -116.8	Sand	Scrubland	0.5	113	1870	11.3	5	T
Scanlon *et al*. (2007)	32.8, -101.9	Loamy sand	Cropland	19	452	1677	53.3	2	T
	32.8, -101.9	Loamy sand	Cropland	31	449	1677	53.3	2	T
	32.8, -101.9	Loamy sand	Cropland	39	446	1677	53.3	2	T
	32.8, -101.9	Loamy sand	Grassland	0	426	1677	53.3	2	T
Selaolo (1998)	-24.1, 25.3	Sand	Scrubland	8	400	1372	88.2	0	T
Selaolo *et al*. (2003)	-23.6, 24.3	Sand	Scrubland	0.5	400	1433	78	0	T
	-23.6, 24.3	Sand	Scrubland	1.1	400	1433	78	0	T
	-23.6, 24.3	Sand	Scrubland	3.8	400	1433	78	0	T
	-24.1, 25.1	Sand	Scrubland	4	420	1384	86.1	0	T
	-24.1, 25.1	Sand	Scrubland	9.8	420	1384	86.1	0	T
	-25.3, 25.6	Sand	Scrubland	11	500	1394	102.2	0	T
	-25.3, 25.6	Sand	Scrubland	16	500	1394	102.2	0	T

Sharda *et al*. (2006)	23.1, 73.3	Sandy clay loam	Cropland	62.7	835	1734	322.4	2	T
	23.1, 73.3	Sandy clay loam	Cropland	71	835	1734	322.4	2	WTF
Sharma and Gupta (1987)	26.3, 73.1	Sand	Cropland	16.6	219	1963	122.8	2	T
	26.3, 73.1	Sand	Cropland	17.4	219	1963	122.8	2	T
	26.6, 72.8	Sand	NoVeg	21.8	389	1907	108.6	1	T
	26.3, 73.1	Sand	NoVeg	22.1	219	1963	122.8	2	T
	26.8, 71.3	Sand	NoVeg	22.3	165	1770	65.2	2	T
	26.3, 73.1	Sand	NoVeg	25.7	219	1963	122.8	2	T
	26.6, 72.8	Sand	NoVeg	46.8	389	1907	108.6	1	T
Silburn *et al*. (2009)	-24.8, 149.8	Crackling clay	Cropland	19.8	720	1510	75.6	1	T
	-24.8, 149.8	Crackling clay	Grassland	0.16	720	1510	75.6	1	T
	-24.8, 149.8	Crackling clay	NoVeg	32.4	720	1510	75.6	1	T
	-24.8, 149.8	Crackling clay	Woodland	0.17	720	1510	75.6	1	T
	-24.8, 149.8	Clay	Woodland	0.26	720	1510	75.6	1	T
Singh *et al*. (1984)	-0.1, 34.8	Sandy clay loam	Grassland	55	1278	1702	163.9	2	WB
Sloots and Wijnen (1990)	-24.4, 25.6	Sand	Scrubland	9	492	1357	94.1	0	
Smettem (1998)	-33.9, 121.8	Sand	Grassland	35	500	1410	85.5	5	WB
Smith *et al*. (1998)	-35.4, 147.6	Sandy clay loam	Cropland	33.3	343	1079	38.7	5	WB, Lysimeter

	-35.4, 147.6	Sandy clay loam	Cropland	97	628		1079	38.7	5	WB, Lysimeter
Snow *et al*. (1999)	-35.4, 147.6	850	Woodland	216	674	896	1079	38.7	5	Model
Sophocleous (2005)	34.3, -102.8	Clay loam	Cropland	7	408		1596	59.5	1	Model
	38.9, -101.8	Clay loam	Cropland	15	465		1274	69.8	2	Model
	40.6, -102.3	Clay loam	Cropland	29.5	448		1201	74.6	2	Model
	40.6, -102.3	Silt loam	Cropland	49.5	448		1201	74.6	2	Model
	38.1, -101.3	Clay loam	Cropland	91	623		1376	65.9	1	Model
	47.9, -97.1	Clay loam	Cropland	102	464		809	67.1	1	Model
	40.6, -98.1	Clay loam	Cropland	109	668		1138	95.5	2	Model
	34.3, -102.8	Clay loam	Grassland	0	408		1596	59.5	1	Model
	40.6, -102.3	Clay loam	Grassland	1	448		1201	74.6	2	Model
	40.6, -102.3	Silt loam	Grassland	2	448		1201	74.6	2	Model
	38.9, -101.8	Clay loam	Grassland	8	465		1274	69.8	2	Model
	38.1, -101.3	Clay loam	Grassland	19	623		1376	65.9	1	Model
	47.9, -97.1	Clay loam	Grassland	40	464		809	67.1	1	Model
	40.6, -98.1	Clay loam	Grassland	92	668		1138	95.5	2	Model
Sophocleous and McAllister (1987)	38.1, -98.8	Silty clay loam	Cropland	65	600		1330	87.9	1	WB
	38.1, -98.8	Coarse sands	Cropland	103	600		1330	87.9	1	WB
	38.1, -98.8	Silty clay loam	Grassland	1.6	600		1330	87.9	1	WB
	38.1, -98.8	Coarse sands	Grassland	42	600		1330	87.9	1	WB
Stone *et al*. (1983)	46.6, -119.4	Sand	NoVeg	127	240		1083	18.5	5	
Stonestrom *et al*.	38.6, -116.1	Sand	Scrubland	0	113		1358	9.3	4	T

(2003)									
Sukhija *et al*. (1988)	11.9, 79.8	Coarse sands	Cropland	80	1200	1702	290.7	6	T
	11.9, 79.8	Sand	Cropland	110	1200	1702	290.7	6	T
	11.9, 79.8	Coarse sands	Cropland	130	1200	1702	290.7	6	T
	11.9, 79.8	Sand	Cropland	160	1200	1702	290.7	6	T
	11.9, 79.8	Sand	Cropland	180	1200	1702	290.7	6	T
	11.9, 79.8	Coarse sands	Cropland	200	1200	1702	290.7	6	T
Sumioka and Bauer (2004)	48.3, -122.6	Sandy loam	Woodland	89.8	618	643	69.1	4	T
	48.3, -122.6	Coarse sands	Woodland	116	618	643	69.1	4	WB
Sun and Cornish (2005)	-31.8, 150.6	180	Grassland	4.9	738	1226	65.3	0	Model
Talsma and Gardner (1986)	-35.4, 148.8	10	Woodland	120	1230	1020	59.2	5	Baseflow, WTF
Taylor and Howard (1996)	2.6, 32.6	Clay	Cropland	200	1400	1558	174.6	6	T, Model
Thorburn *et al*. (1991)	-24.8, 149.8	Clay	Cropland	17.6	650	1510	75.6	1	T
	-24.8, 149.8	Clay	Grassland	2.3	650	1510	75.6	1	T
	-24.8, 149.8	Clay	Woodland	0	650	1510	75.6	1	T
Thorpe (1989)	-31.8, 115.9	Sand	Scrubland	174	830	1661	155.5	4	T
Timmerman (1985)	-32.3, 18.4	Coarse sands	Scrubland	38.7	216	1398	40.7	4	
	-32.3, 18.4	Coarse sands	Scrubland	43.5	290	1398	40.7	4	
Timmerman (1986)	-32.1, 18.6	Coarse sands	Scrubland	20	250	1434	44.3	4	
Tomasella *et al*. (2007)	-3.1, -60.1	Clay	Woodland	438	2627	1299	260.8	5	Baseflow
Unkovich *et al*.	-35.1, 139.3	Sand	Cropland	1	300	1306	22.6	5	WB

(2003)	-35.1, 141.9	Sandy loam	Cropland	2.5	330	1373	15.7	5	WB
	-36.6, 143.9	Silty clay loam	Cropland	58.8	425	1241	23	6	WB
	-36.6, 143.9	Silty clay loam	Woodland	0.02	425	1241	23	6	WB
van Lanen and Dijksma (1999)	51.1, 5.8	Sand	Grassland	293	905	547	24.9	1	Model
Vandoolaeghe and Bertram (1982)	-33.6, 18.4	Coarse sands	Scrubland	98.8	380	1236	73.2	5	WB
Vegter (1995)	-32.3, 18.4	Coarse sands	Scrubland	15.7	196	1398	40.7	4	
	-32.1, 18.4	Coarse sands	Scrubland	23.5	196	1398	39.4	4	
Verhagen (1994)	-22.1, 26.3	Sand	Scrubland	6	500	1408	83.8	0	T
	-23.8, 25.1	Sand	Scrubland	6	450	1396	81.1	0	T
	-23.8, 25.1	Sand	Scrubland	11.5	450	1396	81.1	0	T
Walker *et al*. (1990a)	-34.3, 141.3	Sandy clay loam	Grassland	4.7	295	1387	11.5	3	T
Walker *et al*. (1990b)	-36.8, 140.9	Clay	Grassland	1	520	1210	52.3	5	T
	-36.3, 140.8	Clay	Grassland	5	500	1245	44	6	T
	-36.9, 140.8	Clay	Grassland	8.5	580	1207	63.4	5	T
Walker *et al*. (1992a)	-35.1, 139.4	Sand	Grassland	60	580	1299	23.7	5	T
Walker *et al*. (1992b)	-35.4, 139.6	Sand	Cropland	25.5	380	1278	35	5	T
	-35.4, 139.6	Sandy loam	Grassland	13	380	1278	35	5	T
Walvoord and Phillips (2004)	31.4, -104.4	Clay loam	Grassland	0.1	365	1699	58.2	2	T
	31.4, -104.4	Clay loam	Scrubland	0	275	1699	58.2	2	T
	31.4, -104.4	Clay loam	Scrubland	0.05	365	1699	58.2	2	T
Wang *et al*. (2004)	37.4, 104.9	Fine sand	NoVeg	48	191	879	54	1	WB,

										Lysimeter
Wang *et al*. (2008)	37.8, 115.8	Clay	Cropland	15.3	423	251	1044	154.7	0	T
	37.9, 115.8	Clay	Cropland	131	667	281	1041	158.9	0	T
	38.1, 114.4	Silty clay	Cropland	168	626	178	1019	155.1	1	T
	37.4, 116.3	Silty clay	Cropland	198	650	63	1079	184	0	T
	38.3, 116.8	Silt	Cropland	256	670	787	1059	206.3	0	T
	37.8, 115.8	Clay	Grassland	0	544		1044	154.7	0	T
	37.4, 116.3	Silty clay	Grassland	84.6	643		1079	184	0	T
Wanke *et al*. (2008)	-22.6, 18.3	35	Grassland	7.6	409		1542	76	1	Model
	-22.6, 18.3	43	NoVeg	75.3	409		1542	76	1	Model
	-22.6, 18.3	Sand	Scrubland	7	409		1542	76	1	Model
Ward *et al*. (2002)	-33.8, 117.4	Loamy sand	Grassland	17	483		1286	65.8	5	WB
	-33.8, 117.4	Loamy sand	Grassland	45	483		1286	65.8	5	WB
Watson *et al*. (2004)	-43.6, 172.1	Silt loam	Woodland	17	625		686	28	5	Lysimeter
Weaver *et al*. (2005)	-30.3, 149.3	Clay	Cropland	31.3	514	350	1495	63.3	0	T
	-30.3, 149.4	Clay	Cropland	56.5	460	300	1479	65.4	0	T
	-30.3, 149.6	Clay	Cropland	72.5	417	150	1466	71.9	0	T
	-30.3, 149.3	Clay	Cropland	87.3	514	500	1495	63.3	0	T
	-30.3, 149.3	Clay	Cropland	121	514	650	1495	63.3	0	T
Webb *et al*. (2008)	39.1, -75.4	Silty loam	Cropland	159	1150		1020	32.2	1	Model
Wechsung *et al*. (2000)	52.6, 13.4	Sand	Cropland	114	534		593	36.6	1	Model
	52.6, 13.4	Sand	Woodland	28.9	534		593	36.6	1	Model
Wegenhenkel *et al*. (2008)	52.4, 13.3	Sand	Grassland	269	545		601	38.2	1	Lysimeter

Weltz and Blackburn (1995)	27.6, -98.3	Fine sandy loam	Grassland	22	887	1493	101.1	1	Lysimeter
	27.6, -98.3	Fine sandy loam	NoVeg	78	887	1493	101.1	1	Lysimeter
	27.6, -98.3	Fine sandy loam	Woodland	0	887	1493	101.1	1	WB
White (1997)	-35.4, 147.6	Sand	Grassland	22	650	1079	38.7	5	WB
	-35.4, 147.6	Sand	Grassland	62	697	1079	38.7	5	WB
White *et al*. (2003)	-35.1, 147.4	Sandy clay loam	Grassland	44.5	593	1134	22.1	4	WB
	-30.6, 150.6	Clay loam	Grassland	47.5	662	1275	71.5	0	WB
	-37.4, 141.9	Sandy loam	Grassland	142	642	1160	57.8	6	WB
	-33.6, 149.1	Sandy loam	Grassland	159	885	1136	32.8	5	WB
	-34.9, 117.8	Sand	Grassland	161	758	1190	103	5	WB
	-37.1, 145.9	Loamy sand	Grassland	161	813	1066	84.7	6	WB
Williamson *et al*. (2004)	34.3, -117.8	Sandy loam	Grassland	55	678	1267	68.2	6	T
	34.3, -117.8	Sandy loam	Scrubland	39	678	1267	68.2	6	T
Wright *et al*. (1988)	33.3, -99.3	Clay loam	Grassland	0.13	679	1610	70.9	2	Lysimeter
Zeppel *et al*. (2006)	-31.4, 150.8	Sand	Woodland	21	752	1226	55.9	0	WB
Zhang *et al*. (1999)	-33.4, 145.6	Sandy clay	Cropland	8.5	564	1400	14.7	3	Model
	-35.1, 142.1	Sandy clay loam	Grassland	9.5	351	1379	15.5	5	Model
Zhu (2000)	36.1, -111.3	Fine sands	Grassland	16	305	1545	19	1	T
	36.1, -111.3	Fine sands	Grassland	16	305	1545	19	1	T
Zouari *et al*. (1999)	34.9, 8.1	Sand	Grassland	0.9	94	1226	30.8	4	T

[†]Approximate latitude and longitude of the studies
[‡]Soil texture or *Ks* (mm/day) given in the studies
[§]Variables estimated from CRU dataset: *PET* (mm/yr), *Amplitude* (mm/month), *Phase* (months)
[¶]Values as reported in the studies: Recharge, Precipitation, and Irrigation (mm/yr)
[#]Recharge estimation methods based on natural and injected tracers such as Cl^-, stable and radio-isotopes of water (T), water table fluctuations (WTF), water balance from monitoring of soil moisture or evapotranspiration (WB), baseflow of surface water bodies (baseflow), simulations of soil water movement, water balance, GIS or spatially explicit models (Model), electromagnetic induction (EMI).

References

Abdalla, O. A. E. 2008. Groundwater discharge mechanism in semi-arid regions and the role of evapotranspiration. Hydrological Processes **22**:2993-3009.

Ahmed, I. and R. Umar. 2008. Hydrogeological framework and water balance studies in parts of Krishni–Yamuna interstream area, Western Uttar Pradesh, India. Environmental Geology **53**:1723-1730.

Al-Sagaby, A. and A. Moallim. 2001. Isotopes based assessment of groundwater renewal and related anthropogenic effects in water scarce areas: Sand dunes study in Qasim area, Saudi Arabia. International Atomic Energy Agency, Vienna, Austria.

Allen, A. J. 1981. Groundwater proceses of the Swan Coastal Plain, near Perth, Western Australia. CSIRO Division of Land Management, Wembley, Perth, W.A.

Allison, G., P. Cook, S. Barnett, G. Walker, I. Jolly, and M. Hughes. 1990. Land clearance and river salinisation in the western Murray Basin, Australia. Journal of Hydrology **119**:1-20.

Allison, G. and M. Hughes. 1972. Comparison of recharge to groundwater under pasture and forest using environmental tritium. Journal of Hydrology **17**:81-95.

Allison, G. and M. Hughes. 1978. The use of environmental tritium and chloride to estimate total local recharge to an unconfined aquifer. Australian Journal of Soil Research **16**:181-195.

Allison, G. and M. Hughes. 1983a. The use of natural tracers as indicators of soil-water movement in a temperate semi-arid region. Journal of Hydrology **60**:157-173.

Allison, G., W. Stone, and M. Hughes. 1985. Recharge in karst and dune elements of a semi-arid landscape as indicated by natural isotopes and chloride. Journal of Hydrology **76**:1-25.

Allison, G. B. and M. W. Hughes. 1983b. The Use of Natural Tracers as Indicators of Soil-Water Movement in a Temperate Semi-Arid Region. Journal of Hydrology **60**:157-173.

Amro, H., S. Kilani, J. Jawawdeh, and I. Abd El-Din. 2001. Isotope based assessment of groundwater recharge and pollution in water scarce areas: A case study in Jordan. International Atomic Energy Agency, Vienna, Austria.

Anderson, G. C., I. R. P. Fillery, P. J. Dolling, and S. Asseng. 1998. Nitrogen and water flows under pasture-wheat and lupin-wheat rotations in deep sands in Western Australia. 1. Nitrogen fixation in legumes, net N mineralisation, and utilisation of soil-derived nitrogen. Australian Journal of Agricultural Research **49**:329-344.

Anderson, H. W. 1947. Soil freezing and thawing as related to some vegetation, climatic, and soil variables. Journal of Forestry **45**:94-101.

Andrews, R., J. Lloyd, and D. Lerner. 1997. Modelling of nitrate leaching from arable land into unsaturated soil and chalk 2. Model confirmation and application to agricultural and sewage sludge management. Journal of Hydrology **200**:198-221.

Ansley, R. J., M. Mirik, and M. J. Castellano. 2010. Structural biomass partitioning in regrowth and undisturbed mesquite (Prosopis glandulosa): implications for bioenergy uses. Global Change Biology Bioenergy **2**:26-36.

Anuraga, T. S., L. Ruiz, M. S. M. Kumar, M. Sekhar, and A. Leijnse. 2006. Estimating groundwater recharge using land use and soil data: A case study in South India. Agricultural Water Management **84**:65-76.

Aragón, R., E. Jobbágy, and E. Viglizzo. 2010. Surface and groundwater dynamics in the sedimentary plains of the Western Pampas (Argentina). Ecohydrology.

Archer, S., C. Scifres, C. R. Bassham, and R. Maggio. 1988. Autogenic Succession in a Sub-Tropical Savanna - Conversion of Grassland to Thorn Woodland. Ecological Monographs **58**:111-127.

Archer, S., M. Vavra, W. Laycock, and R. Pieper. 1994. Woody plant encroachment into southwestern grasslands and savannas: rates, patterns and proximate causes. Pages 13-68. Society for Range Management.

Arora, V. 2002. Modeling vegetation as a dynamic component in soil-vegetation-atmosphere transfer schemes and hydrological models. Reviews of Geophysics **40**:26.

Athavale, R., C. Murti, and R. Chand. 1980a. Estimation of recharge to the phreatic aquifers of the Lower Maner Basin, India, by using the titrium injection method. Journal of Hydrology **45**:185-202.

Athavale, R. N., C. S. Murti, and R. Chand. 1980b. Estimation of Recharge to the phreatic Aquifers of the Lower Maner Basin, India, by Using the Tritium Injection Method. Journal of Hydrology **45**:185-202.

Babiker, I. S., M. A. A. Mohamed, T. Hiyama, and K. Kato. 2005. A GIS-based DRASTIC model for assessing aquifer vulnerability in Kakamigahara Heights, Gifu Prefecture, central Japan. Science of the Total Environment **345**:127-140.

Batjes, N. H. 1996. Total carbon and nitrogen in the soils of the world. European Journal of Soil Science **47**:151-164.

Beekman, H. E., E. T. Selaolo, and N. G-J. 1996. Groundwater recharge at the fringe of the Kalahari – the Letlhakeng–Botlhapatou area. Botswana J Earth Sci **3**:19–23.

Bekele, E. B., R. B. Salama, and D. P. Commander. 2006. Impact of change in vegetation cover on groundwater recharge to a phreatic aquifer in Western Australia: assessment of several recharge estimation techniques. Australian Journal of Earth Sciences **53**:905-917.

Bellot, J., J. Sanchez, E. Chirino, N. Hernandez, F. Abdelli, and J. Martinez. 1999. Effect of different vegetation type cover on the soil water balance in semi-arid areas of south eastern Spain. Physics and Chemistry of the Earth, Part B: Hydrology, Oceans and Atmosphere **24**:353-357.

Bent, G. C. 2001. Effects of forest-management activities on runoff components and ground-water recharge to Quabbin Reservoir, central Massachusetts. Forest Ecology and Management **143**:115-129.

Berner, R. A., A. C. Lasaga, and R. Garrels. 1983. The carbonate-silicate geochemical cycle and its effect on atmospheric carbon dioxide over the past 100 million years. Am. J. Sci **283**:641–683.

Beverly, C., M. Bari, B. Christy, M. Hocking, and K. Smettem. 2005. Predicted salinity impacts from land use change: comparison between rapid assessment approaches and a detailed modelling framework. Australian Journal of Experimental Agriculture **45**:1453-1469.

Bhatti, J., M. Apps, and H. Jiang. 2002. Influence of nutrients, disturbances and site conditions on carbon stocks along a boreal forest transect in central Canada. Plant and Soil **242**:1-14.

Bird, P., T. Jackson, G. Kearney, G. Saul, R. Waller, and G. Whipp. 2004. The effect of improved pastures and grazing management on soil water storage on a basaltic plains site in south-west Victoria. Australian Journal of Experimental Agriculture **44**:559-569.

Birkeland, P. W. 1984. Soils and Geomophology. Oxford University Press. New York, New York.

Boumans, L. J. M., D. Fraters, and G. Van Drecht. 2005. Nitrate leaching in agriculture to upper groundwater in the sandy regions of the Netherlands during the 1992-1995 period. Environmental Monitoring and Assessment **102**:225-241.

Bradford, J. B., W. K. Lauenroth, and I. C. Burke. 2005. The impact of cropping on primary production in the US great plains. Ecology **86**:1863-1872.

Bredenkamp, D. B. 1988. Quantitative estimation of ground-water recharge in dolomite. Pages 449-460 *in* I. Simmers, editor. Estimation of Natural Groundwater Recharge. Proc. NATO Advance Research Workshop on Estimation of Natural Recharge of Groundwater, Antalya, Turkey, 1987, Reidel, Dordrecht.

Bredenkamp, D. B. and M. A. C. Vandoolaeghe. 1982. Die Ontginbare Grondwater Potensiaal van die Atlantis Gebied. Geohydrology, Cape Town.

Briggs, J. M., A. K. Knapp, J. M. Blair, J. L. Heisler, G. A. Hoch, M. S. Lett, and J. K. McCARRON. 2005. An ecosystem in transition: causes and consequences of the conversion of mesic grassland to shrubland. BioScience **55**:243-254.

Brookshire, D. S., B. Colby, M. Ewers, and P. T. Ganderton. 2004. Market prices for water in the semiarid West of the United States. Water Resources Research **40**:doi:10.1029/2003WR002846.

Butler, M. J. and B. T. Verhagen. 2001. Isotope studies of a thick unsaturated zone in a semi-arid area of Southern Africa. International Atomic Energy Agency, Vienna, Austria.

Calder, I. R., I. Reid, T. R. Nisbet, and J. C. Green. 2003. Impact of lowland forests in England on water resources: Application of the Hydrological Land Use Change (HYLUC) model. Water Resources Research **39**:1319.

Canadell, J., R. B. Jackson, J. R. Ehleringer, H. A. Mooney, O. E. Sala, and E. D. Schulze. 1996. Maximum rooting depth of vegetation types at the global scale. Oecologia **108**:583-595.

Carbon, B., F. Roberts, P. Farrington, and J. Beresford. 1982. Deep drainage and water use of forests and pastures grown on deep sands in a Mediterranean environment. Journal of Hydrology **55**:53-63.

Carlson, D., T. Thurow, R. Knight, and R. K. Heitschmidt. 1988. Effect of Honey Mesquite on the Water Balance of Texas Rolling Plains Rangeland. Texas Water Development Board.

CCX. Chicago Climate Exchange. CCX CFI Monthly Summary.

Cerling, T. E. 1984. The stable isotopic composition of modern soil carbonate and its relationship to climate. Earth and Planetary Science Letters **71**:229-240.

Cherkauer, D. S. and S. A. Ansari. 2005. Estimating ground water recharge from topography, hydrogeology, and land cover. Ground water **43**:102-112.

Chichester, F. W. and R. F. Chaison. 1992. Analysis of Carbon in Calcareous Soils Using a 2 Temperature Dry Combustion Infrared Instrumental Procedure. Soil Science **153**:237-241.

Cho, J., V. Barone, and S. Mostaghimi. 2009. Simulation of land use impacts on groundwater levels and streamflow in a Virginia watershed. Agricultural Water Management **96**:1-11.

Christensen, N. L., A. M. Bartuska, J. H. Brown, S. Carpenter, C. Dantonio, R. Francis, J. F. Franklin, J. A. MacMahon, R. F. Noss, D. J. Parsons, C. H. Peterson, M. G. Turner, and R. G. Woodmansee. 1996. The report of the ecological society of America committee on the scientific basis for ecosystem management. Ecological Applications **6**:665-691.

Cihacek, L. J. and M. G. Ulmer. 2002. Effects of tillage on inorganic carbon storage in soils of Norther Great Plains of the US. Pages 63-69 *in* J. M. Kimble, R. Lal, and R. F. Follett, editors. Agricultural Practices and Policies for Carbon Sequestration in Soil. CRC Press, Boca Raton, Florida.

Cochard, R. and P. J. Edwards. 2011. Structure and biomass along an Acacia zanzibarica woodland–savanna gradient in a former ranching area in coastal Tanzania. Journal of Vegetation Science.

Coleman, K., D. Jenkinson, G. Crocker, P. Grace, J. Klir, M. Korschens, P. Poulton, and D. Richter. 1997. Simulating trends in soil organic carbon in long-term experiments using RothC-26.3. Geoderma **81**:29-44.

Colville, J. and J. Holmes. 1972. Water table fluctuations under forest and pasture in a karstic region of southern Australia. Journal of Hydrology **17**:61-80.

Cook, P., T. Hatton, D. Pidsley, A. Herczeg, A. Held, A. O'Grady, and D. Eamus. 1998. Water balance of a tropical woodland ecosystem, Northern Australia: a combination of micro-meteorological, soil physical and groundwater chemical approaches. Journal of Hydrology **210**:161-177.

Cook, P., I. Jolly, F. Leaney, G. Walker, G. Allan, L. Fifield, and G. Allison. 1994. Unsaturated zone tritium and chlorine 36 profiles from southern Australia: Their use as tracers of soil water movement. Water Resources Research **30**:1709-1719.

Cook, P. and S. Kilty. 1992. A helicopter-borne electromagnetic survey to delineate groundwater recharge rates. Water Resources Research **28**:2953-2961.

Cook, P., F. Leaney, and M. Miles. 2004. Groundwater recharge in the north-east Mallee Region, South Australia. 1446-6171, Commonwealth Scientific & Industrial Research Organisation.

Cook, P., G. Walker, and I. Jolly. 1989. Spatial variability of groundwater recharge in a semiarid region. Journal of Hydrology **111**:195-212.

Cook, P. G. 1992. The spatial and temporal variability of groundwater recharge. Flinders University of South Australia.

Cook, P. G., W. M. Edmunds, and C. B. Gaye. 1992. Estimating paleorecharge and paleoclimate from unsaturated zone profiles. Water Resources Research **28**:2721-2731.

Cowan, I. 1982. Regulation of water use in relation to carbon gain in higher plants. Physiological Plant Ecology II **12**:589–615.

Crosbie, R. S., J. D. Hughes, J. Friend, and B. J. Baldwin. 2007. Monitoring the hydrological impact of land use change in a small agricultural catchment affected by dryland salinity in central NSW, Australia. Agricultural Water Management **88**:43-53.

Dams, J., S. Woldeamlak, and O. Batelaan. 2008. Predicting land-use change and its impact on the groundwater system of the Kleine Nete catchment, Belgium. Hydrology of Earth Systems Science **12**:1369-1385.

Daniel, J. A. 1999. Influence of wheat tillage practices on shallow groundwater recharge. Journal of Soil and Water Conservation **54**:560-564.

Datta, P. S., B. I. Desai, and S. K. Gupta. 1980. Hydrological investigations in Sabarmati basin 1. Groundwater recharge estimation using tritium taggin method. Proc. Indian natn. Sci. Acad., Part A. Phys. Sci. **46**:84-98.

De Vries, J. J., E. T. Selaolo, and H. E. Beekman. 2000. Groundwater recharge in the Kalahari, with reference to paleo-hydrologic conditions. Journal of Hydrology **238**:110-123.

Deans, J. D., W. M. Edmunds, D. K. Lindley, C. B. Gaye, B. Dreyfus, J. J. Nizinski, M. Neyra, K. Ingleby, and R. C. Munro. 2005. Nitrogen in interstitial waters in the sahel; Natural baseline, pollutant or resource? Plant and Soil **271**:47-62.

DeFries, R. S., J. A. Foley, and G. P. Asner. 2004. Land-use choices: balancing human needs and ecosystem function. Frontiers in Ecology and the Environment **2**:249-257.

Denef, K., C. Stewart, J. Brenner, and K. Paustian. 2008. Does long-term center-pivot irrigation increase soil carbon stocks in semi-arid agro-ecosystems? Geoderma **145**:121-129.

Di, H. and K. Cameron. 2002. Nitrate leaching and pasture production from different nitrogen sources on a shallow stoney soil under flood-irrigated dairy pasture. Australian Journal of Soil Research **40**:317-334.

Dodds, W. K., K. Gido, M. R. Whiles, K. M. Fritz, and W. J. Matthews. 2004. Life on the edge: the ecology of Great Plains prairie streams. BioScience **54**:205-216.

Doll, P. and K. Fiedler. 2008. Global-scale modeling of groundwater recharge. Hydrology and Earth System Sciences **12**:863-885.

Doll, P., F. Kaspar, and B. Lehner. 2003. A global hydrological model for deriving water availability indicators: model tuning and validation. Journal of Hydrology **270**:105-134.

Dolling, P., S. Asseng, M. Robertson, and M. Ewing. 2007. Water excess under simulated lucerne–wheat phased systems in Western Australia. Australian Journal of Agricultural Research **58**:826-838.

Dripps, W. and K. Bradbury. 2007. A simple daily soil–water balance model for estimating the spatial and temporal distribution of groundwater recharge in temperate humid areas. Hydrogeology Journal **15**:433-444.

Duffkova, R. 2002. The effect of rainfall and extensive use of grasslands on water regime. Rostlinna Vyroba **48**:89-95.

Dunin, F., J. Williams, K. Verburg, and B. Keating. 1999. Can agricultural management emulate natural ecosystems in recharge control in south eastern Australia? Agroforestry Systems **45**:343-364.

Dyck, M. and R. de Jong. 2003. Long-term movement of a chloride tracer under transient, semi-arid conditions. Soil Science Society of America Journal **67**:471-477.

Edmunds, W., E. Fellman, I. Goni, and C. Prudhomme. 2002. Spatial and temporal distribution of groundwater recharge in northern Nigeria. Hydrogeology Journal **10**:205-215.

Edmunds, W. and C. Gaye. 1994. Estimating the spatial variability of groundwater recharge in the Sahel using chloride. Journal of Hydrology **156**:47-59.

Edmunds, W. M. 2001. Investigation of the unsaturated zone in semi-arid regions using isotopic and chemical methods and applications to water resource problems. International Atomic Energy Agency, Vienna, Austria.

Eldridge, D. J., M. A. Bowker, F. T. Maestre, E. Roger, J. F. Reynolds, and W. G. Whitford. Impacts of shrub encroachment on ecosystem structure and functioning: towards a global synthesis. Ecology Letters.

Emmerich, W. E. 2003. Carbon dioxide fluxes in a semiarid environment with high carbonate soils. Agricultural and Forest Meteorology **116**:91-102.

Eswaran, H., P. F. Reich, and J. M. Kimble. 2000. Global carbon stocks. Pages 15–25 *in* R. Lal, J. M. Kimble, H. Eswaran, and B. A. Stewart, editors. Global climate change and pedogenic carbonates. CRC Press, Boca Raton, USA.

Facchi, A., C. Gandolfi, B. Ortuani, D. Maggi, and M. Bruen. 2005. Simulation supported scenario analysis for water resources planning: a case study in Northern Italy. Water science & technology **51**:11-18.

Farley, K. A., E. G. Jobbagy, 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**.

Favreau, G., C. Leduc, C. Marlin, M. Dray, J. D. Taupin, M. Massault, C. L. G. La Salle, and M. Babic. 2002. Estimate of recharge of a rising water table in semiarid Niger from 3H and 14C modeling. Ground water **40**:144-151.

Fayer, M., G. Gee, M. Rockhold, and M. Freshley. 1996. Estimating recharge rates for a groundwater model using a GIS. Journal of Environmental quality **25**.

Fillery, I. and R. Poulter. 2006. Use of long-season annual legumes and herbaceous perennials in pastures to manage deep drainage in acidic sandy soils in Western Australia. Australian Journal of Agricultural Research **57**:297-308.

Finch, J. 1998a. Estimating direct groundwater recharge using a simple water balance model-sensitivity to land surface parameters. Journal of Hydrology **211**:112-125.

Finch, J. W. 1998b. Estimating direct groundwater recharge using a simple water balance model - sensitivity to land surface parameters. Journal of Hydrology **211**:112-125.

Fisher, L. H. and R. W. Healy. 2008. Water movement within the unsaturated zone in four agricultural areas of the United States. Journal of Environmental quality **37**:1051-1063.

Foley, J. A., R. DeFries, G. P. Asner, C. Barford, G. Bonan, S. R. Carpenter, F. S. Chapin, M. T. Coe, G. C. Daily, and H. K. Gibbs. 2005. Global consequences of land use. Science **309**:570.

Ford, D. C. and P. W. Williams. 1989. Karst geomorphology and hydrology. Unwin Hyman, London, UK.

Fouty, S. C. 1989. Chloride Mass Balance as a Method for Determining Long-term Groundwater Recharge Rates and Geomorphic-surface Stability in Arid and Semi-arid Regions, Whisky Flat and Beatty, Nevada. University of Arizona., Tucson.

Fredeen, A. L., C. H. Bois, D. T. Janzen, and P. T. Sanborn. 2005. Comparison of coniferous forest carbon stocks between old-growth and young second-growth forests on two soil types in central British Columbia, Canada. Canadian Journal of Forest Research **35**:1411-1421.

Freeze, A. R. and J. A. Cherry. 1979. Groundwater. Prentice Hall, Inc., Upper Saddle River, NJ.

Galloway, D. L., D. R. Jones, S. Ingebritsen, and G. Survey. 1999. Land subsidence in the United States. US Geological Survey.

Garbrecht, J., M. Van Liew, and G. O. Brown. 2004. Trends in precipitation, streamflow, and evapotranspiration in the Great Plains of the United States.

Gates, J. B., W. 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.

Gaye, C. and W. Edmunds. 1996. Groundwater recharge estimation using chloride, stable isotopes and tritium profiles in the sands of northwestern Senegal. Environmental Geology **27**:246-251.

Gee, G., M. Fayer, M. Rockhold, P. Wierenga, M. Young, and B. Andraski. 1994. Variations in water balance and recharge potential at three western desert sites. Soil Science Society of America Journal;(United States) **58**:63-72.

Gee, G. W., D. Felmy, J. C. Ritter, M. D. Campbell, J. L. Downs, M. J. Fayer, R. R. Kirkham, and S. O. Link. 1993. Field Lysimeter Test Facility. Pacific Northwest Laboratory, Richland, Washington.

George, R. J. and P. W. C. Frantom. 1988. Preliminary Groundwater and Salinity Investigations in the Eastern Wheatbelt: Merredin Catchment. Western Australian Dept. of Agriculture.

Gerten, D., S. Schaphoff, U. Haberlandt, W. Lucht, and S. Sitch. 2004. Terrestrial vegetation and water balance - hydrological evaluation of a dynamic global vegetation model. Journal of Hydrology **286**:249-270.

Ghadiri, H. and C. Rose. 1992. Modeling Chemical Transport in Soils: Natural and Applied Contaminants. Lewis Publishers, Boca Raton.

Gieske, A. 1992. Dynamics of groundwater recharge – a case study in semi-arid eastern Botswana. Vrije Universiteit Amsterdam.

Gieske, A., E. Selaolo, and H. Beekman. 1995. Tracer interpretation of moisture transport in a Kalahari sand profile. Application of tracers in arid zone hydrology (United Kingdom).

Gilliland, C. F., J. Robertson, and H. Cover. 2004. Water Power. Real Estate Center, Texas A&M University.

Gilson, P., B. Zollinger, J. A. Aistrup, J. Heinrichs, F. H. S. University, D. I. o. P. A. C. f. S. Research, and S. K. G. M. D. N. 3. 2001. The Value of Ogallala Aquifer Water in Southwest Kansas. Fort Hays State University, the Docking Institute of Public Affairs, Center for Survey Research.

Goni, I. B. and W. M. Edmunds. 2001. The use of unsaturated zone solutes and deuterium profiles in the study of groundwater recharge in the semi-arid zone of Nigeria., International Atomic Energy Agency, Vienna, Austria.

Goodrich, D. C., D. G. Williams, C. L. Unkrich, J. F. Hogan, R. L. Scott, K. R. Hultine, D. Pool, A. L. Coes, and S. Miller. 2004. Comparison of methods to estimate ephemeral channel recharge, Walnut Gulch, San Pedro River basin, Arizona. Groundwater recharge in a desert environment: the southwestern United States:77–99.

Gourlet-Fleury, S., V. Rossi, M. Rejou Mechain, V. Freycon, A. Fayolle, L. Saint André, G. Cornu, J. Gérard, J. M. Sarrailh, and O. Flores. 2011. Environmental filtering of dense wooded species controls above ground biomass stored in African moist forests. Journal of Ecology **99**:981-990.

Green, C. T., L. H. Fisher, and B. A. Bekins. 2008. Nitrogen fluxes through unsaturated zones in five agricultural settings across the United States. Journal of Environmental quality **37**:1073-1085.

Gregory, P. J., D. Tennant, A. P. Hamblin, and J. Eastham. 1992. Components of the water balance on duplex soils in Western Australia. Australian Journal of Experimental Agriculture **32**:845-855.

Griffin, R. C. 2006. Water resource economics: the analysis of scarcity, policies, and projects. MIT Press, Cambridge, MA.

Guo, L. B. and R. M. Gifford. 2002. Soil carbon stocks and land use change: a meta analysis. Global Change Biology **8**:345-360.

Gupta, S. and P. Sharma. 1984. Soil moisture transport through the unsaturated zone: tritium tagging studies in Sabarmati basin, western India/Déplacement de l'humidité du sol à travers la zone non saturée: études avec marquage au tritium dans le bassin du Sabarmati, Inde occidentale. Hydrological Sciences Journal **29**:177-189.

Gutmann, M. P., W. J. Parton, G. Cunfer, and I. Burke. 2005. Population and environment in the US Great Plains. Pages 84–105 *in* B. Entwisle and P. C. Stern, editors. Population, land use, and environment: research directions. Natinal Academies Press, Washington, DC.

Hadas, A., A. Hadas, B. Sagiv, and N. Haruvy. 1999. Agricultural practices, soil fertility management modes and resultant nitrogen leaching rates under semi-arid conditions. Agricultural Water Management **42**:81-95.

Halm, D., T. Gaiser, and K. Stahr. 2002. Seepage and groundwater recharge in sandy soils of the semi-arid region of Picos, Northeast Brazil. Neues Jahrbuch Fur Geologie Und Palaontologie-Abhandlungen **225**:85-101.

Harmel, R., C. Richardson, K. King, and P. Allen. 2006. Runoff and soil loss relationships for the Texas Blackland Prairies ecoregion. Journal of Hydrology **331**:471-483.

Harris, D. H., W. R. Horwath, and C. van Kessel. 2001. Acid fumigation of soils to remove carbonates prior to total organic carbon or carbon-13 isotopic analysis. Soil Science Society of America Journal **65**:1853.

Hatton, T. and R. Nulsen. 1999. Towards achieving functional ecosystem mimicry with respect to water cycling in southern Australian agriculture. Agroforestry Systems **45**:203-214.

Heilweil, V. M., D. K. Solomon, and P. M. Gardner. 2006. Borehole environmental tracers for evaluating net infiltration and recharge through desert bedrock. Vadose Zone Journal **5**:98.

Helyar, K., W. Porter, and A. Robson. 1989. Soil acidification, its measurement and the processes involved. Pages 61-101 *in* A. D. Robson, editor. Soil acidity and plant growth. Academic Press, Sydney, Australia.

Heng, L., R. White, K. Helyar, R. Fisher, and D. Chen. 2001. Seasonal differences in the soil water balance under perennial and annual pastures on an acid Sodosol in southeastern Australia. European Journal of Soil Science **52**:227-236.

Hierro, J. L., L. C. Branch, D. Villarreal, and K. L. Clark. 2000. Predictive equations for biomass and fuel characteristics of Argentine shrubs. Journal of Range Management **53**:617-621.

Holmes, J. and J. Colville. 1970a. Forest hydrology in a karstic region of southern Australia. Journal of Hydrology **10**:59-74.

Holmes, J. and J. Colville. 1970b. Grassland hydrology in a karstic region of southern Australia. Journal of Hydrology **10**:38-58.

Holmes, J. W. and J. S. Colville. 1968. On the water balance of grassland and forest. Pages 39–46 *in* Transactions of the 9th Congress of the International Soil Science Society, Adelaide, Australia.

Holmstead, G. L., R. W. Knight, and M. A. Hussey. 1988. Water-use and Water Yield of Three C4 Bunchgrasses in the South Texas Plains. Texas Water Development Board.

Horton, R. E. 1933. The role of infiltration in the hydrologic cycle. Trans. Am. Geophys. Union **14**:446-460.

Houghton, R. 2000. Interannual variability in the global carbon cycle. Journal of Geophysical Research **105**:20121-20120,20130.

Houghton, R., J. Hackler, and K. Lawrence. 2000. Changes in terrestrial carbon storage in the United States. 2: The role of fire and fire management. Global Ecology and Biogeography **9**:145-170.

Houston, J. F. T. 1982. Rainfall and recharge to a dolomite aquifer in a semi-arid climate at Kabwe, Zambia. Journal of Hydrology **59**:173-187.

Howard, K. W. F. and J. Karundu. 1992. Constraints on the exploitation of basement aquifers in East Africa--water balance implications and the role of the regolith. Journal of Hydrology **139**:183-196.

Huang, M. and J. Gallichand. 2006. Use of the SHAW model to assess soil water recovery after apple trees in the gully region of the Loess Plateau, China. Agricultural Water Management **85**:67-76.

Hughes, M. W., P. G. Cook, I. D. Jolly, T. A. Beech, and C. T. Fiebiger. 1988. Recharge Studies in the Western Murray Basin: 1. Results of a Drilling Program at Borrika. CSIRO Division of Water Resources

Hume, I. H. 1997. Episodic deep drainage under crops and shrubs in the mallee zone. Pages 19-26 *in* Proceedings Dryland Forum. Murray-Darling Basin Commission, North Adelaide, South Australia.

Hussein, M. F. 2001. Water flow and solute transport using environmental isotopes and modeling. International Atomic Energy Agency, Vienna, Austria.

Hutjes, R. W. A., P. Kabat, S. W. Running, W. J. Shuttleworth, C. Field, B. Bass, M. A. F. D. S. Dias, R. Avissar, A. Becker, M. Claussen, A. J. Dolman, R. A. Feddes, M. Fosberg, Y. Fukushima, J. H. C. Gash, L. Guenni, H. Hoff, P. G. Jarvis, I. Kayane, A. N. Krenke, C. Liu, M. Meybeck, C. A. Nobre, L. Oyebande, A. Pitman, R. A. Pielke, Sr., M. Raupach, B. Saugier, E. D. Schulze, P. J. Sellers, T. D. Tenhonen, R. Valentini, R. L. Victoria, and C. J. Vorosmarty. 1998. Biospheric aspects of the hydrological cycle. Journal of Hydrology (Amsterdam) **212**:1-21.

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

IAEA. 2001. Isotope Based Assessment of Groundwater Renewal in Water Scarce Regions, Vienna.

INTA. Agrometeorologia. http://www.inta.gov.ar/index.asp. Instituto Nacional de Tecnología Agropecuaria.

IPCC. 2007. Intergovernmental Panel on Climate Change. Climate Change 2007: Synthesis Report. Summary for Policymakers. Cambridge University Press, Cambridge, United Kingdom.

Jackson, D. and K. Rushton. 1987. Assessment of recharge components for a chalk aquifer unit. Journal of Hydrology **92**:1-15.

Jackson, R. B., J. L. Banner, E. G. Jobbágy, W. T. Pockman, and D. H. Wall. 2002. Ecosystem carbon loss with woody plant invasion of grasslands. Nature **418**:623-626.

Jackson, R. B., S. R. Carpenter, C. N. Dahm, D. M. McKnight, R. J. Naiman, S. L. Postel, and S. W. Running. 2001. Water in a changing world. Ecological Applications **11**:1027-1045.

Jackson, R. B., E. G. Jobbagy, R. Avissar, S. B. Roy, D. J. Barrett, C. W. Cook, K. A. Farley, D. C. le Maitre, B. A. McCarl, and B. C. Murray. 2005. Trading water for carbon with biological sequestration. Science **310**:1944-1947.

Jackson, R. B., J. T. Randerson, J. G. Canadell, R. G. Anderson, R. Avissar, D. D. Baldocchi, G. B. Bonan, K. Caldeira, N. S. Diffenbaugh, and C. B. Field. 2008. Protecting climate with forests. Environmental Research Letters **3**:044006.

Jackson, R. B., H. J. Schenk, E. G. Jobbágy, J. Canadell, G. D. Colello, R. E. Dickinson, C. B. Field, P. Friedlingstein, M. Heimann, and K. Hibbard. 2000. Belowground consequences of vegetation change and their treatment in models. Ecological Applications **10**:470-483.

Jan, C. D., T. H. Chen, and W. C. Lo. 2007. Effect of rainfall intensity and distribution on groundwater level fluctuations. Journal of Hydrology **332**:348-360.

Jipp, P. H., D. C. Nepstad, D. Cassel, and C. Reis de Carvalho. 1998. Deep soil moisture storage and transpiration in forests and pastures of seasonally-dry Amazonia. Climatic Change **39**:395-412.

Jobbagy, E. G. and R. B. Jackson. 2007. Groundwater and soil chemical changes under phreatophytic tree plantations. Journal of Geophysical Research-Biogeosciences **112**:15.

Jobbágy, E. G. and R. B. Jackson. 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. Ecological Applications **10**:423-436.

Johnson, C. M., D. J. Zarin, and A. H. Johnson. 2000. Post-disturbance aboveground biomass accumulation in global secondary forests. Ecology **81**:1395-1401.

Johnson, P., G. Sides, and D. Schmidt-Wertz. 2007. Profitability of cotton production in the Texas High Plains1996 – 2005 Beltwide Cotton Conference, New Orleans, Louisiana.

Johnston, C. D. 1987a. Distribution of environmental chloride in relation to subsurface hydrology. Journal of Hydrology **94**:67-88.

Johnston, C. D. 1987b. Preferred water flow and localised recharge in a variable regolith. Journal of Hydrology **94**:129-142.

Jolly, I., P. Cook, G. Allison, and M. Hughes. 1989. Simultaneous water and solute movement through an unsaturated soil following an increase in recharge. Journal of Hydrology(Amsterdam) **111**:391-396.

Jolly, J. 1992. The Geohydrology of the Graafwater Government Subterranean Water Control Area. Geohydrology, Cape Town.

Joshi, B. 1997. Estimation of diffuse vadose zone soil-water flux in a semi-arid region. University of Saskatchewan.

Julien, P. A., R. W. Knight, and C. L. Fischer. 1988. Water Yields from Mesquite and Grass Lysimeters on the Carrizo-Wilcox Sands Aquifer in Southwest Texas. Texas Water Development Board.

Junge, C. E. and R. Werby. 1958. Concentration of chloride, sodium, potassium, calcium, and sulfate in rain water over the United States. J. Meteorol.;(United States) **15**.

Kansas State University. K. State Research and Extension, Weather Data Library.

Keese, K. E., B. R. Scanlon, and R. C. Reedy. 2005. Assessing controls on diffuse groundwater recharge using unsaturated flow modeling. Water Resources Research **41**.

Kelly, E. F., R. G. Amundson, B. D. Marino, and M. J. DeNiro. 1991. Stable carbon isotopic composition of carbonate in Holocene grassland soils. Soil Science Society of America journal (USA).

Kendall, H. W. and D. Pimentel. 1994. Constraints on the expansion of the global food supply. Ambio **23**:198-205.

Kendy, E., P. Gérard Marchant, M. Todd Walter, Y. Zhang, C. Liu, and T. S. Steenhuis. 2003. A soil water balance approach to quantify groundwater recharge from irrigated cropland in the North China Plain. Hydrological Processes **17**:2011-2031.

Kendy, E., Y. Zhang, C. Liu, J. Wang, and T. Steenhuis. 2004. Groundwater recharge from irrigated cropland in the North China Plain: case study of Luancheng County, Hebei Province, 1949–2000. Hydrological Processes **18**:2289-2302.

Kennett-Smith, A., P. Cook, and G. Walker. 1994a. Factors affecting groundwater recharge following clearing in the south western Murray Basin. Journal of Hydrology **154**:85-105.

Kennett-smith, A., P. G. Cook, and G. R. Walker. 1994b. Factors Affecting Groundwater Recharge Following Clearing in the South Western Murray Basin. Journal of Hydrology **154**:85-105.

Kennett-Smith, A. K., G. R. Budd, P. G. Cook, and G. R. Walker. 1990. The Effect of Lucerne on the Recharge to Cleared Mallee Lands. Centre for Groundwater Studies.

Kennett-Smith, A. K., G. R. Budd, and G. R. Walker. 1992a. Groundwater Recharge Beneath Woodlands Cleared for Grazing, in South Western New South Wales., CSIRO, Division of Water Resources and Centre for Groundwater Studies.

Kennett-Smith, A. K., R. Thorne, and G. R. Walker. 1992b. Comparison of Recharge Under Native Vegetation and Dryland Agriculture, in the Big Desert Region of Victoria. Centre for Groundwater Studies.

Kennett-Smith, A. K., R. Thorne, and G. R. Walker. 1993. Comparison of Recharge Under Native Vegetation and Dryland Agriculture Near Goroke, Victoria. Centre for Groundwater Studies.

Kergoat, L. 1998. A model for hydrological equilibrium of leaf area index on a global scale. Journal of Hydrology **213**:268-286.

Keywood, M., A. Chivas, L. Fifield, R. Cresswell, and G. Ayers. 1997. The accession of chloride to the western half of the Australian continent. Australian Journal of Soil Research **35**:1177-1190.

Kienzle, S. and R. Schulze. 1992. A simulation model to assess the effect of afforestation on ground-water resources in deep sandy soils. Water S. A. **18**:265-272.

Kim, J. H. and R. B. Jackson. , In review. A global analysis of groundwater recharge for vegetation, climate, and soils.

Klein Goldewijk, C. G. M. and J. J. Battjes. 1997. A Hundred Year (1890 - 1990) Database for Integrated Environmental Assessments (HYDE, version 1.1). Bilthoven.

Klute, A. 1986. Methods of soil analysis. Part 1, Physical and mineralogical methods. Soil Science Society of America, Madison, Wisconsin.

Knapp, A. K., J. M. Briggs, S. L. Collins, S. R. Archer, M. S. Bret-Harte, B. E. Ewers, D. P. Peters, D. R. Young, G. R. Shaver, E. Pendall, and M. B. Cleary. 2008. Shrub encroachment in North American grasslands: shifts in growth form dominance rapidly alters control of ecosystem carbon inputs. Global Change Biology **14**:615-623.

Knight, R. W. and T. L. Thurow. 1991. Can brush management benefit water quantity and quality? Page 23 *in* T. G. Welch, editor. Brush Management Symposium, Giddings, Texas.

Knoche, D., A. Embacher, and J. Katzur. 2002. Water and element fluxes of red oak ecosystems during stand development on post-mining sites (Lusatian Lignite District). Water, Air, & Soil Pollution **141**:219-231.

Kohavi, R. 1995. A study of cross-validation and bootstrap for accuracy estimation and model selection. Pages 1137-1145 International Joint Conference on Artificial Intelligence. Citeseer.

Kowalski, A. S., P. Serrano-Ortiz, I. A. Janssens, S. Sanchez-Moral, S. Cuezva, F. Domingo, A. Were, and L. Alados-Arboledas. 2008. Can flux tower research neglect geochemical CO2 exchange? Agricultural and Forest Meteorology **148**:1045-1054.

Krajenbrink, G., D. Ronen, W. Van Duijvenbooden, M. Magaritz, and D. Wever. 1988. Monitoring of recharge water quality under woodland. Journal of Hydrology **98**:83-102.

Kucharik, C. J., J. A. Foley, C. Delire, V. A. Fisher, M. T. Coe, J. D. Lenters, C. Young-Molling, N. Ramankutty, J. M. Norman, and S. T. Gower. 2000. Testing the performance of a Dynamic Global Ecosystem Model: Water balance, carbon balance, and vegetation structure. Global Biogeochemical Cycles **14**:795-825.

Külls, C. 2000. Groundwater of the North-Western Kalahari, Namibia-Estimation of recharge and quantification of the flow systems. Universität Würzburg.

Kuzyakov, Y. and G. Domanski. 2000. Carbon input by plants into the soil. Review. Journal of Plant Nutrition and Soil Science **163**:421-431.

Ladekarl, U., K. R. Rasmussen, S. Christensen, K. H. Jensen, and B. Hansen. 2005. Groundwater recharge and evapotranspiration for two natural ecosystems covered with oak and heather. Journal of Hydrology **300**:76-99.

Lal, R. and J. Kimble. 2000. Pedogenic carbonates and the global carbon cycle. Global climate change and pedogenic carbonates:1–13.

Lambers, H., F. S. Chapin, and T. L. Pons. 1998. Plant physiological ecology. Springer.

Larsen, F., R. Owen, T. Dahlin, P. Mangeya, and G. Barmen. 2002. A preliminary analysis of the groundwater recharge to the Karoo formations, mid-Zambezi basin, Zimbabwe. Physics and Chemistry of the Earth, Parts A/B/C **27**:765-772.

Laurance, W. F. 2007. A new initiative to use carbon trading for tropical forest conservation. Biotropica **39**:20-24.

Laurance, W. F., P. M. Fearnside, S. G. Laurance, P. Delamonica, T. E. Lovejoy, J. M. Rankin-de Merona, J. Q. Chambers, and C. Gascon. 1999. Relationship between soils and Amazon forest biomass: a landscape-scale study. Forest Ecology and Management **118**:127-138.

Leaney, F. and G. Allison. 1986. Carbon-14 and stable isotope data for an area in the Murray Basin: its use in estimating recharge. Journal of Hydrology **88**:129-145.

Leaney, F. and A. Herczeg. 1999. The origin of fresh groundwater in the SW Murray Basin and its potential for salinisation. CSIRO Land and Water.

Leaney, F. W. and A. L. Herczeg. 1995. Regional recharge to a karst aquifer estimated from chemical and isotopic composition of diffuse and localised recharge, South Australia. Journal of Hydrology **164**:363-387.

Leduc, C., G. Favreau, and P. Schroeter. 2001. Long-term rise in a Sahelian water-table: The Continental Terminal in south-west Niger. Journal of Hydrology **243**:43-54.

LeQuéré, C., P. Ciais, T. Conway, C. Field, S. Houghton, G. Marland, and P. Canadell. 2008. Global Carbon Project: Carbon budget and trends 2007.

Li, S. G., C. T. Lai, G. Lee, S. Shimoda, T. Yokoyama, A. Higuchi, and T. Oikawa. 2005. Evapotranspiration from a wet temperate grassland and its sensitivity to microenvironmental variables. Hydrological Processes **19**:517-532.

Lin, R. and K. Wei. 2001. Environmental isotope profiles of the soil water in loess unsaturated zone in semi-arid areas of china. International Atomic Energy Agency, Vienna, Austria.

Liu, R., Y. Li, and Q. X. Wang. 2011. Variations in water and CO2 fluxes over a saline desert in western China. Hydrological Processes.

Loh, I. C. and R. A. Stokes. 1981. Predicting stream salinity changes in south-western Australia. Agricultural Water Management **4**:227-254.

Lull, H. W. and E. N. Munns. 1950. Effect of land use practices on ground water. Jour Soil and Water Conserv **5**:169-179.

Lvovitch, M. I. 1970. World water balance (general report). International Association of Hydrological Sciences.

Maréchal, J. C., B. Dewandel, S. Ahmed, L. Galeazzi, and F. Zaidi. 2006. Combined estimation of specific yield and natural recharge in a semi-arid groundwater basin with irrigated agriculture. Journal of Hydrology **329**:281-293.

Maréchal, J. C., M. R. R. Varma, J. Riotte, J. M. Vouillamoz, M. Kumar, L. Ruiz, M. Sekhar, and J. J. Braun. 2009. Indirect and direct recharges in a tropical forested watershed: Mule Hole, India. Journal of Hydrology **364**:272-284.

Marion, G. M. 2001. Carbonate mineral solubility at low temperatures in theNa-K-Mg-Ca-H-Cl-SO4-OH-HCO3-CO3-CO2-H2O system. Geochimica et Cosmochimica Acta. **65**: 1883-1896.

Marion, G. M., W. H. Schlesinger, P. J. Fonteyn. 1985. CALDEP: A regional model for soil $CaCO_3$ (caliche) deposition in southwestern deserts. Soil Science **139**: 468-481.

Matson, P., W. Parton, A. Power, and M. Swift. 1997. Agricultural intensification and ecosystem properties. Science **277**:504.

McCarl, B. A. and U. A. Schneider. 2001. Climate change - Greenhouse gas mitigation in US agriculture and forestry. Science **294**:2481-2482.

McCulley, R. L., E. G. Jobbagy, W. T. Pockman, and R. B. Jackson. 2004. Nutrient uptake as a contributing explanation for deep rooting in arid and semi-arid ecosystems. Oecologia **141**:620-628.

McDowall, M., D. Hall, D. Johnson, J. Bowyer, and P. Spicer. 2003. Kikuyu and annual pasture: a characterisation of a productive and sustainable beef production system on the South Coast of Western Australia. Australian Journal of Experimental Agriculture **43**:769-783.

McHale, M., I. Burke, M. Lefsky, P. Peper, and E. McPherson. 2009. Urban forest biomass estimates: is it important to use allometric relationships developed specifically for urban trees? Urban Ecosystems **12**:95-113.

McMahon, P., K. Dennehy, B. Bruce, J. Böhlke, R. Michel, J. Gurdak, and D. Hurlbut. 2006. Storage and transit time of chemicals in thick unsaturated zones under rangeland and irrigated cropland, High Plains, United States. Water Resour. Res **42**:W03413.

McMahon, P., K. Dennehy, R. Michel, M. Sophocleous, K. Ellett, and D. Hurlbut. 2003. Water Movement Through Thick Unsaturated Zones Overlying the Central High Plains Aquifer, Southwestern Kansas. Water-Resources Investigations Report 03–4171, Reston, VA.

MEA. 2000. Millenium Ecosystem Assessment: Ecosystems and Human Well being: A Framework for Assessment, Overall Synthesis. Island Press, Washington, DC.

Meyer, W. B. and B. L. Turner. 1994. Changes in land use and land cover: a global perspective. Cambridge University Press, Cambridge.

Mielnick, P., W. Dugas, K. Mitchell, and K. Havstad. 2005. Long-term measurements of CO2 flux and evapotranspiration in a Chihuahuan desert grassland. Journal of arid environments **60**:423-436.

Mikhailova, E. A. and J. Christopher. 2006. Effects of land use on soil inorganic carbon stocks in the Russian Chernozem. Journal of environmental quality **35**:1384.

Mileham, L., R. Taylor, J. Thompson, M. Todd, and C. Tindimugaya. 2008. Impact of rainfall distribution on the parameterisation of a soil-moisture balance model of groundwater recharge in equatorial Africa. Journal of Hydrology **359**:46-58.

Milly, P. C. D. 1994. Climate, soil water storage, and the average annual water balance. Water Resources Research **30**:2143-2156.

Milly, P. C. D. 1997. Sensitivity of greenhouse summer dryness to changes in plant rooting characteristics. Geophysical Research Letters **24**:269-271.

Milroy, S., S. Asseng, and M. Poole. 2008. Systems analysis of wheat production on low water-holding soils in a Mediterranean-type environment:: II. Drainage and nitrate leaching. Field Crops Research **107**:211-220.

Mokany, K., R. Raison, and A. S. Prokushkin. 2006. Critical analysis of root: shoot ratios in terrestrial biomes. Global Change Biology **12**:84-96.

Monger, H. C. and R. A. Gallegos. 2000. Biotic and abiotic processes and rates of pedogenic carbonate accumulation in the southwestern united states-relationship to atmospheric CO2 sequestration. Pages 273–289 *in* R. Lal, J. M. Kimble, H. Eswaran, and B. A. Stewart, editors. Global climate change and pedogenic carbonates. CRC Press, Boca Raton, USA.

Monirul Islam, M. and P. Kanungoe. 2005. Natural recharge to sustainable yield from the Barind aquifer: a tool in preparing effective management plan of groundwater resources. Water science and technology **52**:251-258.

Moody, P. and R. Aitken. 1997. Soil acidification under some tropical agricultural systems. 1. Rates of acidification and contributing factors. Australian Journal of Soil Research **35**:163-173.

Müller, J. and A. Bolte. 2009. The use of lysimeters in forest hydrology research in north-east Germany. Landbauforschung—vTI Agriculture and Forest Research **59**:1-10.

Murray, B. C., W. A. Jenkins, R. A. Kramer, and S. P. Faulkner. 2009. Valuing Ecosystem Services from Wetlands Restoration in the Mississippi Alluvial Valley. Nicholas Institute for Environmental Policy Solutions. Duke University.

NADP. National Atmospheric Deposition Program.*in* E. Cooperative State Research, and Extension Service, U.S. Department of Agriculture, editor.

NADP. 2011. NADP/NTN National Trends Network. National Atmospheric Deposition Program Office, Champaign, IL.

Navada, S. V., A. R. Nair, U. K. Sinha, U. P. Kulkarni, and T. B. Joseph. 2001. Application of isotopes and chemistry in unsaturated zone in arid areas of Rajasthan, India. International Atomic Energy Agency, Vienna, Austria.

Neilson, R. P. 1995. A Model for Predicting Continental-scale Vegetation Distribution and Water-balance. Ecological Applications **5**:362-385.

New, M., D. Lister, M. Hulme, and I. Makin. 2002. A high-resolution data set of surface climate over global land areas. Climate Research **21**:1-25.

Newman, B. D., A. R. Campbell, and B. P. Wilcox. 1997. Tracer-based studies of soil water movement in semi-arid forests of New Mexico. Journal of Hydrology **196**:251-270.

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**.

Nichols, D. S. and E. S. Verry. 2001. Stream flow and ground water recharge from small forested watersheds in north central Minnesota. Journal of Hydrology **245**:89-103.

Nieswiadomy, M. 1985. The demand for irrigation water in the High Plains of Texas, 1957-80. American Journal of Agricultural Economics:619-626.

Niu, G. Y., Z. L. Yang, R. E. Dickinson, L. E. Gulden, and H. Su. 2007. Development of a simple groundwater model for use in climate models and evaluation with Gravity Recovery and Climate Experiment data. Journal of Geophysical Research-Atmospheres **112**:14.

NOAA. National Climatic Data Center. http://www.ncdc.noaa.gov/oa/ncdc.html. National Environmental Satellite, Data and information Service.

NRC. 1997. National Research Council. Valuing ground water: economic concents and approaches. National Academy Press, Washington, DC.

NRC. 2005. National Research Council. Valuing Ecosystem Services: Toward Better Environmental Decision-Making. National Academies Press, Washington, D.C.

O'Connell, M. G., M. Incerti, and O. L. G.J. 1990. Effect of Fallowing on Groundwater Recharge. Murray-Darling Basin.*in* 1990 Workshop on Groundwater Research and Management, Mildura, Victoria.

O'Connell, M. G., G. J. O'Leary, and D. J. Connor. 2003. Drainage and change in soil water storage below the root-zone under long fallow and continuous cropping sequences in the Victorian Mallee. Australian Journal of Agricultural Research **54**:663-675.

Ojeda, C. G. 2001. Aquifer recharge estimation at the Mesilla Bolson and Guaymas aquifer systems, Mexico. International Atomic Energy Agency, Vienna, Austria.

Olenick, K. L., J. R. Conner, R. N. Wilkins, U. P. Kreuter, and W. T. Hamilton. 2004. Economic implications of brush treatments to improve water yield. Journal of Range Management **57**:337-345.

Pacala, S. and R. Socolow. 2004. Stabilization wedges: solving the climate problem for the next 50 years with current technologies. Science **305**:968.

Pakrou, N. and P. Dillon. 2000. Comparison of type and depth of lysimeter for measuring the leaching losses of nitrogen under urine patches. Soil Use and Management **16**:108-116.

Palmer, M., E. Bernhardt, E. Chornesky, S. Collins, A. Dobson, C. Duke, B. Gold, R. Jacobson, S. Kingsland, and R. Kranz. 2004. Ecology for a crowded planet. Science **304**:1251.

Papiernik, S., M. Lindstrom, T. Schumacher, J. Schumacher, D. Malo, and D. Lobb. 2007. Characterization of soil profiles in a landscape affected by long-term tillage. Soil and Tillage Research **93**:335-345.

Pastor, J. and W. Post. 1986. Influence of climate, soil moisture, and succession on forest carbon and nitrogen cycles. Biogeochemistry **2**:3-27.

Paydar, Z. and J. Gallant. 2008. A catchment framework for one dimensional models: introducing FLUSH and its application. Hydrological Processes **22**:2094-2104.

Peck, A. and D. Hurle. 1973. Chloride balance of some farmed and forested catchments in southwestern Australia. Water Resources Research **9**:648-657.

Peck, A., C. Johnston, and D. Williamson. 1981. Analyses of solute distributions in deeply weathered soils. Agricultural Water Management **4**:83-102.

Peel, M. C., T. A. McMahon, B. L. Finlayson, and F. G. R. Watson. 2001. Identification and explanation of continental differences in the variability of annual runoff. Journal of Hydrology **250**:224-240.

Petheram, C., G. Walker, R. Grayson, T. Thierfelder, and L. Zhang. 2002. Towards a framework for predicting impacts of land-use on recharge: 1. A review of recharge studies in Australia. Australian Journal of Soil Research **40**:397-417.

Phillips, F. M. 1994. Environmental Tracers for Water-movement in Desert Soils of the American Southwest. Soil Science Society of America Journal **58**:15-24.

Piñeiro, G., E. G. Jobbágy, R. B. Jackson, C. S. Santoni, S. I. Portela, and C. Di Bella. 2007. RP-RainNet: The Rio de la Plata Atmospheric Deposition Network. Set up and Preliminary Results. Meeting of the American Geophysical Union-Joint Assembly., Cancún, Mexico.

Point Carbon. European Union Emission Trading Scheme over-the-counter Price.

Postel, S. and S. Carpenter. 1997. Freshwater ecosystem services. Pages 95–214 *in* G. C. Daily, editor. Nature’s services: societal dependence on natural ecosystems. Island Press, Washington, D.C.

Potter, N. J., L. Zhang, P. C. D. Milly, T. A. McMahon, and A. J. Jakeman. 2005. Effects of rainfall seasonality and soil moisture capacity on mean annual water balance for Australian catchments. Water Resources Research **41**:W06007.

Pracilio, G., S. Asseng, S. E. Cook, G. Hodgson, M. Wong, M. Adams, and T. Hatton. 2003. Estimating spatially variable deep drainage across a central-eastern wheatbelt catchment, Western Australia. Australian Journal of Agricultural Research **54**:789-802.

Priyadarshi, N. 2008. Groundwater concept in ancient civilization. Earth Day Network.

Prych, E. A. 1998. Using chloride and chlorine-36 as soil-water tracers to estimate deep percolation at selected locations on the US Department of Energy Hanford Site, Washington. 0886-9308, US Geological Survey, Denver, CO.

Radford, B., D. Silburn, and B. Forster. 2009. Soil chloride and deep drainage responses to land clearing for cropping at seven sites in central Queensland, northern Australia. Journal of Hydrology **379**:20-29.

Ragab, R., J. Finch, and R. Harding. 1997. Estimation of groundwater recharge to chalk and sandstone aquifers using simple soil models. Journal of Hydrology **190**:19-41.

Ramankutty, N., A. T. Evan, C. Monfreda, and J. A. Foley. 2008. Farming the planet: 1. Geographic distribution of global agricultural lands in the year 2000. Global Biogeochemical Cycles **22**.

Rangarajan, R., N. Mondal, V. Singh, and S. Singh. 2009. Estimation of natural recharge and its relation with aquifer parameters in and around Tuticorin town, Tamil Nadu, India. Current Science **97**:217-226.

Rawls, W. J., D. L. Brakensiek, and K. E. Saxton. 1982. Estimation of Soil-water Properties. Transactions of the Asae **25**:1316-&.

Raymond, P. A., N. H. Oh, R. E. Turner, and W. Broussard. 2008. Anthropogenically enhanced fluxes of water and carbon from the Mississippi River. Nature **451**:449-452.

Renard, K., L. Lane, J. Simanton, W. Emmerich, J. Stone, M. Weltz, D. Goodrich, and D. Yakowitz. 1993. Agricultural impacts in an arid environment: Walnut Gulch studies. Hydrological Science and Technology **9**:145-190.

Renger, M., O. Strebel, G. Wessolek, and W. Duynisveld. 1986. Evapotranspiration and groundwater recharge–A case study for different climate, crop patterns, soil properties and groundwater depth conditions. Zeitschrift für Pflanzenernährung und Bodenkunde **149**:371-381.

Renger, M. and G. Wessolek. 1990. Auswirkungen von Grundwasserabsenkungen und Nutzungsänderungen auf die Grundwasserneubildung. Mitt. Inst. für Wasserwesen, Bundeswehr-Hochschule München **38 B**:295–305.

Richardson, S. B. and K. A. Narayan. 1995. The effectiveness of management options for dryland salinity control at Wanilla, South Australia. Agricultural Water Management **29**:63-83.

Richter, D. D. and R. Houghton. 2011. Gross CO2 fluxes from land-use change: implications for reducing global emissions and increasing sinks. Carbon Management **2**:41-47.

Ridley, A., R. White, R. Simpson, and L. Callinan. 1997. Water use and drainage under phalaris, cocksfoot, and annual ryegrass pastures. Australian Journal of Agricultural Research **48**:1011-1024.

Roberts, J. and P. Rosier. 2006. The effect of broadleaved woodland on Chalk groundwater resources. Quarterly Journal of Engineering Geology & Hydrogeology **39**:197.

Robinson, J. R. C. and D. A. McCorkle 2006. Trends and Prospects for Texas Cotton. Cotton Outlook's Special Feature. Texas; Connecting the Old West to the New East.

Rodriguez, J. P., T. D. Beard, E. M. Bennett, G. S. Cumming, S. J. Cork, J. Agard, A. P. Dobson, and G. D. Peterson. 2006. Trade-offs across space, time, and ecosystem services. Ecology and Society **11**.

Rodvang, S., D. Mikalson, and M. Ryan. 2002. Changes in ground water quality in an irrigated area of southern Alberta. Journal of Environmental quality **33**:476-487.

Rosenberger, R. S. and J. B. Loomis. 2003. Benefit transfer. Pages 445-482 *in* P. A. Champ, K. J. Boyle, and T. C. Brown, editors. A primer on nonmarket valuation. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Ruesink, L. E. 1979. Water's Worth. Texas Water Resources Institute.

Sami, K. and D. Hughes. 1996. A comparison of recharge estimates to a fractured sedimentary aquifer in South Africa from a chloride mass balance and an integrated surface-subsurface model. Journal of Hydrology **179**:111-136.

Sankaran, M., J. Ratnam, and N. Hanan. 2008. Woody cover in African savannas: the role of resources, fire and herbivory. Global Ecology and Biogeography **17**:236-245.

Santoni, C., E. Jobbágy, and S. Contreras. 2010. Vadose zone transport in dry forests of central Argentina: Role of land use. Water Resources Research **46**:W10541.

Sartori, F., R. Lal, M. H. Ebinger, and J. A. Eaton. 2007. Changes in soil carbon and nutrient pools along a chronosequence of poplar plantations in the Columbia Plateau, Oregon, USA. Agriculture, ecosystems & environment **122**:325-339.

Scanlon, B. R. 1991. Evaluation of moisture flux from chloride data in desert soils. Journal of Hydrology **128**:137-156.

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., I. Jolly, M. Sophocleous, and L. Zhang. 2007a. Global impacts of conversions from natural to agricultural ecosystems on water resources: Quantity versus quality. Water Resources Research **43**.

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.

Scanlon, B. R., R. P. Langford, and R. S. Goldsmith. 1999. Relationship between geomorphic settings and unsaturated flow in an arid setting. Water Resources Research **35**:983-999.

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 of the United States of America **102**:6033-6038.

Scanlon, B. R., R. C. Reedy, R. L. Baumhardt, and G. Strassberg. 2008. Impact of deep plowing on groundwater recharge in a semiarid region: Case study, High Plains, Texas. Water Resources Research **44**:11.

Scanlon, B. R., R. C. Reedy, D. A. Stonestrom, D. E. Prudic, and K. F. Dennehy. 2005b. Impact of land use and land cover change on groundwater recharge and quality in the southwestern US. Global Change Biology **11**:1577-1593.

Scanlon, B. R., R. C. Reedy, and J. A. Tachovsky. 2007b. Semiarid unsaturated zone chloride profiles: Archives of past land use change impacts on water resources in the southern High Plains, United States. Water Resources Research **43**:W06423.

Schimel, D. S. 1995. Terrestrial ecosystems and the carbon cycle. Global Change Biology **1**:77-91.

Schimel, D. S., B. Braswell, E. A. Holland, R. McKeown, D. Ojima, T. H. Painter, W. J. Parton, and A. R. Townsend. 1994. Climatic, edaphic, and biotic controls over storage and turnover of carbon in soils. Global Biogeochemical Cycles **8**:279-293.

Schimel, D. S., I. G. Enting, M. Heimann, T. M. L. Wigley, D. Raynaud, D. Alves, and U. Siegenthaler. 1995. CO_2 and the carbon cycle.*in* J. T. Houghton, L. G. Meira Filho, J. Bruce, H. Lee, B. A. Callander, E. Haites, N. Harris, and K. Maskell, editors. Climate Change 1994. Cambridge University Press, Cambridge.

Schlesinger, W. H. 1982. Carbon storage in the caliche of arid soils: a case study from Arizona. Soil Science **133**:247.

Schlesinger, W. H., J. F. Reynolds, G. L. Cunningham, L. F. Huenneke, W. M. Jarrell, R. A. Virginia, and W. G. Whitford. 1990. Biological feedbacks in global desertification. Science **247**:1043.

Schröter, D., W. Cramer, R. Leemans, I. C. Prentice, M. B. Araújo, N. W. Arnell, A. Bondeau, H. Bugmann, T. R. Carter, and C. A. Gracia. 2005. Ecosystem service supply and vulnerability to global change in Europe. Science **310**:1333.

Schwarz, G. 1978. Estimating the dimension of a model. The annals of statistics **6**:461-464.

Searchinger, T., R. Heimlich, R. A. Houghton, F. Dong, A. Elobeid, J. Fabiosa, S. Tokgoz, D. Hayes, and T. H. Yu. 2008. Use of US croplands for biofuels increases greenhouse gases through emissions from land-use change. Science **319**:1238.

Selaolo, E. T. 1998. Tracer studies and groundwater recharge assessment in the eastern fringe of the Botswana Kalahari: the Letlhakeng – Botlhapatlou Area. Vrije Universiteit Amsterdam.

Selaolo, E. T., H. E. Beekman, A. S. M. Gieske, and J. J. de Vries. 2003. Multiple tracer profiling in Botswana–GRES findings. Pages 33–50 *in* Y. Xu and H. E. Beekman, editors. Groundwater Recharge Estimation in Southern Africa. UNESCO International Hydrological Programme, Cape Town.

Serrano-Ortiz, P., F. Domingo, A. Cazorla, A. Were, S. Cuezva, L. Villagarcía, L. Alados-Arboledas, and A. Kowalski. 2009. Interannual CO2 exchange of a sparse Mediterranean shrubland on a carbonaceous substrate. Journal of Geophysical Research **114**:G04015.

Seyfried, M. S. and B. P. Wilcox. 2006. Soil water storage and rooting depth: key factors controlling recharge on rangelands. Hydrological Processes **20**:3261-3275.

Shah, T., D. Molden, R. Sakthivadivel, and D. Seckler. 2000. The global groundwater situation: Overview of opportunities and challenges. International Water Management Institute, Colombo, Sri Lanka.

Sharda, V., R. Kurothe, D. Sena, V. Pande, and S. Tiwari. 2006. Estimation of groundwater recharge from water storage structures in a semi-arid climate of India. Journal of Hydrology **329**:224-243.

Sharma, P. and S. Gupta. 1987. Isotopic investigation of soil water movement: a case study in the Thar desert, western Rajasthan/Recherches isotopiques sur les déplacements de l'eau dans le sol. Hydrological Sciences Journal **32**:469-483.

Shiklomanov, I. A. 1997. Comprehensive assessment of the freshwater resources of the world. World Meteorological Organization, Stockholm, Sweden.

Silburn, D., B. Cowie, and C. Thornton. 2009. The Brigalow Catchment Study revisited: Effects of land development on deep drainage determined from non-steady chloride profiles. Journal of Hydrology **373**:487-498.

Šimůnek, J., D. Jacques, M. T. Van Genuchten, and D. Mallants. 2006. Multicomponent geochemical transport modeling using HYDRUS-1D and HP1. J. Am. Water Resour. Assoc **42**:1537-1547.

Simunek, J., M. T. van Genuchten, and M. Sejna. 2008. Development and applications of the HYDRUS and STANMOD software packages and related codes. Vadose Zone Journal **7**:587.

Singh, J., W. Wapakala, and P. Chebosi. 1984. Estimating groundwater recharge based on the infiltration characteristics of layered soil. Challenges in African Hydrology and Water Resources, IAHS Publ **144**:37-45.

Skiles, J. W. and J. D. Hanson. 1994. Responses of Arid and Semiarid Watersheds to Increasing Carbon-dioxide and Climate-change as Shown by Simulation Studies. Climatic Change **26**:377-397.

Sloots, R. R. and M. M. Wijnen. 1990. Groundwater recharge to a fractured aquifer in S-E Botswana. Results of a survey of the Molepolole-East wellfield. Vrije Universiteit Amsterdam.

Smettem, K. 1998a. Deep drainage and nitrate losses under native vegetation and agricultural systems in the Mediterranean climate region of Australia. Occasional Paper RAPPS02/98, CSIRO, Land Water Resources Research Development Corporation, Canberra, Australia.

Smettem, K. R. J. 1998b. Deep Drainage and Nitrate Losses under Native Vegetation and Agricultural Systems in the Mediterranean Climate Region of Australia. CSIRO, Land and Water Resources Research and Development Corporation, Canberra ACT.

Smith, S., R. Menzel, E. Rhoades, J. Williams, and H. Eck. 1983. Nutrient and sediment discharge from Southern Plains grasslands. Journal of Range Management:435-439.

Smith, C. J., F. X. Dunin, S. J. Zegelin, and R. Poss. 1998. Nitrate leaching from a Riverine clay soil under cereal rotation. Australian Journal of Agricultural Research **49**:379-389.

Snow, V., W. Bond, B. Myers, S. Theiveyanathan, C. Smith, and R. Benyon. 1999. Modelling the water balance of effluent-irrigated trees. Agricultural Water Management **39**:47-67.

Sophocleous, M. 2005. Groundwater recharge and sustainability in the High Plains aquifer in Kansas, USA. Hydrogeology Journal **13**:351-365.

Sophocleous, M. and J. McAllister. 1987. Basinwide water balance modeling with emphasis on spatial distribution of groundwater recharge. Journal of the American Water Resources Association **23**:997-1010.

St. Arnaud, R. J. 1979. Nature and distribution of secondary soil carbonates within landscapes in relation to soluble Mg++/Ca++ ratios. Canadian Journal of Soil Science **59**:87-98.

Stone, W. A., J. M. Thorp, O. P. Gifford, and D. J. Hoitink. 1983. Climatological summary for the Hanford area. Pacific Northwest Laboratory, Richland, WA.

Stonestrom, D. A., J. Constantz, T. P. A. Ferré, and S. A. Leake. 2007. Ground-Water Recharge in the Arid and Semiarid Southwestern United States. U.S. Department of the Interior, U.S. Geological Survey.

Stonestrom, D. A., D. Prudic, R. J. Laczniak, K. C. Akstin, R. A. Boyd, and K. K. Henkelman. 2003. Estimates of deep percolation beneath native vegetation, irrigated fields, and the Amargosa-River channel, Amargosa Desert, Nye County, Nevada. United States Geological Survey-Nevada, Las Vegas, NV (US).

Sukhija, B., D. Reddy, P. Nagabhushanam, and R. Chand. 1988. Validity of the environmental chloride method for recharge evaluation of coastal aquifers, India. Journal of Hydrology **99**:349-366.

Sumioka, S. and H. H. Bauer. 2003. Estimating Ground-Water Recharge from Precipitation on Whidbey and Camano Islands, Island County, Washington, Water Years 1998 and 1999. US Dept. of the Interior, US Geological Survey, Tacoma, Washington.

Sun, H. and P. Cornish. 2005. Estimating shallow groundwater recharge in the headwaters of the Liverpool Plains using SWAT. Hydrological Processes **19**:795-807.

Talsma, T. and E. Gardner. 1986. Groundwater recharge and discharge response to rainfall on a hillslope. Australian Journal of Soil Research **24**:343-356.

Taylor, R. G. and K. W. F. Howard. 1996. Groundwater recharge in the Victoria Nile basin of east Africa: support for the soil moisture balance approach using stable isotope tracers and flow modelling. Journal of Hydrology **180**:31-53.

Teepe, R., R. Brumme, and F. Beese. 2000. Nitrous oxide emissions from frozen soils under agricultural, fallow and forest land. Soil Biology and Biochemistry **32**:1807-1810.

Thorburn, P. J., B. A. Cowie, and P. A. Lawrence. 1991. Effect of land development on groundwater recharge determined from non-steady chloride profiles. Journal of Hydrology **124**:43-58.

Thornthwaite, C. W., J. R. Mather, and D. B. Carter. 1957. Instructions and tables for computing potential evapotranspiration and the water balance. Drexel Institute of Technology, Laboratory of Climatology, Centerton, N.J.

Thorpe, P. M. 1987. Tritium as an indicator of groundwater recharge to the Gnangara groundwater mound on the Swan Coastal Plain, Perth, Western Australia. Pages 41–55 *in* M. L. Sharma, editor. Symposium on Groundwater Recharge. A.A. Balkema, Rotterdam, Mandurah, Western Australia.

Timmerman, L. R. A. 1985. Possibilities for the development of groundwater from the Cenozoic sediments in the lower Berg River Region. Geohydrology, Cape Town.

Timmerman, L. R. A. 1986. Sandveld Region: Possibilities for the Development of a Groundwater Supply Scheme from a Primary Aquifer Northwest of Graafwater. Geohydrology, Cape Town.

Tomasella, J., M. G. Hodnett, L. A. Cuartas, A. D. Nobre, M. J. Waterloo, and S. M. Oliveira. 2007. The water balance of an Amazonian micro-catchment: the effect of interannual variability of rainfall on hydrological behaviour. Hydrological Processes **22**:2133-2147.

Torell, L. A., J. D. Libbin, and M. D. Miller. 1990. The market value of water in the Ogallala aquifer. Land Economics **66**:163-175.

Turpie, J., C. Marais, and J. N. Blignaut. 2008. The working for water programme: Evolution of a payments for ecosystem services mechanism that addresses both poverty and ecosystem service delivery in South Africa. Ecological Economics **65**:788-798.

TWDB. Texas Water Development Board, Water Information, Integration & Dissemination.

U.S. Census Bureau. Statistical Abstracts of United States; The National Data Book.

UNEP. 2007. United Nations Environment Programme. Global Environment Outlook 4: Environment for Development. Progress Press Ltd, Malta.

Unkovich, M., K. Blott, A. Knight, I. Mock, A. Rab, and M. Portelli. 2003. Water use, competition, and crop production in low rainfall, alley farming systems of south-eastern Australia. Australian Journal of Agricultural Research **54**:751-762.

US EPA. 2011. Clean Air Status and Trends Network (CASTNET). National Dry Deposition Network (NDDN), Washington, DC.

USEPA, U. S. E. P. A. 2008. EPA Analysis of the Lieberman– Warner Climate Security Act of 2008.*in* S. i. t. t. Congress, editor.

USGS. National Water Information System. USGS Water-Quality Data for the Nation.

Van Auken, O. W. 2000. Shrub invasions of North American semiarid grasslands. Annual Review of Ecology and Systematics **31**:197-215.

Van der Kamp, G., W. Stolte, and R. Clark. 1999. Drying out of small prairie wetlands after conversion of their catchments from cultivation to permanent brome grass. Hydrological Sciences Journal **44**:387-397.

van Lanen, H. A. J. and R. Dijksma. 1999. Water flow and nitrate transport to a groundwater-fed stream in the Belgian-Dutch chalk region. Hydrological Processes **13**:295-307.

Vandoolaeghe, M. A. C. and E. Bertram. 1982. Atlantis Grondwatersisteem: Herevaluasie van Versekerde Lewering.*in* Geohydrology, editor., Cape Town.

Vegter, J. R. 1995. An Explanation of a Set of National Groundwater Maps. Water Research Commission.

Verhagen, B. T. 1994. Semiarid zone groundwater mineralization processes as revealed by environmental isotope studies. Pages 245-266 Application of Tracers in Arid Zone Hydrology. International Association of Hydrological Sciences, Vienna, Austria.

Vitousek, P. M., H. A. Mooney, J. Lubchenco, and J. M. Melillo. 1997. Human domination of Earth's ecosystems. Science **277**:494-499.

Vorosmarty, C., D. Lettenmaier, C. Leveque, and e. al. 2004. Humans transforming the global water system. Eos **85**:509–520.

Vorosmarty, C. J., P. Green, J. Salisbury, and R. B. Lammers. 2000. Global water resources: Vulnerability from climate change acid population growth. Science **289**:284-288.

Walker, G., M. Gilfedder, and J. Williams. 1999. Effectiveness of current farming systems in the control of dryland salinity.

Walker, G. R., R. M. Blom, and A. K. Kennett-Smith. 1992a. Preliminary Results of Recharge Investigations in the Upper South-East Region of South Australia. Centre for Groundwater Studies.

Walker, G. R., G. R. Budd, P. Pavelic, A. K. Kennett-Smith, and P. G. Cook. 1990a. Groundwater Recharge Beneath Open Woodlands in South Western New South Wales. Centre for Groundwater Studies.

Walker, G. R., P. J. Dillon, P. Pavelic, and A. K. Kennett-Smith. 1992b. Preliminary Results of Recharge and Discharge Investigations at Cooke Plains, South Australia. Centre for Groundwater Studies.

Walker, G. R., I. D. Jolly, and P. G. Cook. 1991. A New Chloride Leaching Approach to the Estimation of Diffuse Recharge Following a Change in Land-Use. Journal of Hydrology **128**:49-67.

Walker, G. R., I. D. Jolly, M. H. Stadter, F. W. Leaney, W. J. Stone, P. G. Cook, R. F. Davie, and L. K. Fifield. 1990b. Estimation of Diffuse Recharge in the Naracoorte Ranges Region, South Australia: an Evaluation of Chlorine-36 for Recharge Studies. Dept Prim. Indust., Canberra.

Walvoord, M. A. and F. M. Phillips. 2004. Identifying areas of basin-floor recharge in the Trans-Pecos region and the link to vegetation. Journal of Hydrology **292**:59-74.

Wang, B., M. Jin, J. R. Nimmo, L. Yang, and W. Wang. 2008. Estimating groundwater recharge in Hebei Plain, China under varying land use practices using tritium and bromide tracers. Journal of Hydrology **356**:209-222.

Wang, X. P., R. Berndtsson, X. R. Li, and E. S. Kang. 2004. Water balance change for a re-vegetated xerophyte shrub area/Changement du bilan hydrique d’une zone replantée d’arbustes xérophiles. Hydrological Sciences Journal **49**:1-295.

Wanke, H., A. Duenkeloh, and P. Udluft. 2008. Groundwater recharge assessment for the Kalahari catchment of north-eastern Namibia and north-western Botswana with a regional-scale water balance model. Water Resources Management **22**:1143-1158.

Ward, P. R., F. X. Dunin, and S. F. Micin. 2002. Water use and root growth by annual and perennial pastures and subsequent crops in a phase rotation. Agricultural Water Management **53**:83-97.

Watson, A. J., T. J. A. Davie, W. Breck'Bowden, and J. J. Payne. 2004. Drainage to groundwater under a closed-canopy radiata pine plantation on the Canterbury Plains, South Island, New Zealand. journal of Hydrology (NZ) **43**:111-123.

Weaver, T. B., N. R. Hulugalle, and H. Ghadiri. 2005. Comparing deep drainage estimated with transient and steady state assumptions in irrigated vertisols. Irrigation Science **23**:183-191.

Webb, R. M. T., M. E. Wieczorek, B. T. Nolan, T. C. Hancock, M. W. Sandstrom, J. E. Barbash, E. R. Bayless, R. W. Healy, and J. Linard. 2008. Variations in pesticide leaching related to land use, pesticide properties, and unsaturated zone thickness. Journal of Environmental quality **37**:1145-1157.

Wechsung, F., V. Krysanova, M. Flechsig, and S. Schaphoff. 2000. May land use change reduce the water deficiency problem caused by reduced brown coal mining in the state of Brandenburg? Landscape and Urban Planning **51**:177-189.

Wegehenkel, M., Y. Zhang, T. Zenker, and H. Diestel. 2008. The use of lysimeter data for the test of two soil–water balance models: A case study. Journal of Plant Nutrition and Soil Science **171**:762-776.

Weltz, M. A. and W. H. Blackburn. 1995. Water budget for south Texas rangelands. Journal of Range Management **48**:45-52.

White, R. E. 1997. Soil water and nitrogen dynamics under perennial and annual pastures. Temperate Pastures Sustainability Key Program–Southern Site.

White, R. E., B. P. Christy, A. M. Ridley, A. E. Okom, S. R. Murphy, W. H. Johnston, D. L. Michalk, P. Sanford, M. R. McCaskill, and I. R. Johnson. 2003. SGS Water Theme: influence of soil, pasture type and management on water use in grazing systems across the high rainfall zone of southern Australia. Australian Journal of Experimental Agriculture **43**:907-926.

White, W. B., D. C. Culver, J. S. Herman, T. C. Kane, and J. E. Mylroie. 1995. Karst Lands. American Scientist **83**:450-459.

Wickens, G. E. 1998. Ecophysiology of economic plants in arid and semi-arid lands. Springer.

Wilcox, B. P., M. K. Owens, R. W. Knight, and R. K. Lyons. 2005. Do woody plants affect streamflow on semiarid karst rangelands? Ecological Applications **15**:127-136.Wilcox, B. P. and Y. Huang. 2010. Woody plant encroachment paradox: Rivers rebound as degraded grasslands convert to woodlands. Geophysical Research Letters **37**:L07402.

Williams, E. L., K. J. Szramek, L. Jin, T. C. W. Ku, and L. M. Walter. 2007. The carbonate system geochemistry of shallow groundwater–surface water systems in temperate glaciated watersheds (Michigan, USA): Significance of open-system dolomite weathering. Geological Society of America Bulletin **119**:515.

Williams, E. L., L. M. Walter, T. C. W. Ku, G. W. Kling, and D. R. Zak. 2003. Effects of CO2 and nutrient availability on mineral weathering in controlled tree growth experiments. Global biogeochemical cycles **17**:1041.

Williams, R. E., B. Allred, R. M. Denio, and H. A. Paulsen Jr. 1968. Conservation, development, and use of the world's rangelands. Journal of Range Management:355-360.

Williamson, T. N., B. D. Newman, R. C. Graham, and P. J. Shouse. 2004. Regolith water in zero-order chaparral and perennial grass watersheds four decades after vegetation conversion. Vadose Zone Journal **3**:1007.

Wilson, M. A. and S. R. Carpenter. 1999. Economic valuation of freshwater ecosystem services in the United States: 1971-1997. Ecological Applications **9**:772-783.

Wohlfahrt, G., L. F. Fenstermaker, and J. A. ARNONE III. 2008. Large annual net ecosystem CO2 uptake of a Mojave Desert ecosystem. Global Change Biology **14**:1475-1487.

Wright, T. A., R. W. Knight, and R. K. Heitschmidt. 1988. Water Yield of North Texas Native Grasslands. Texas Water Development Board.

Zeppel, M. J. B., I. A. M. Yunusa, and D. Eamus. 2006. Daily, seasonal and annual patterns of transpiration from a stand of remnant vegetation dominated by a coniferous Callitris species and a broad leaved Eucalyptus species. Physiologia Plantarum **127**:413-422.

Zhang, L., W. R. Dawes, T. J. Hatton, I. H. Hume, M. G. O'Connell, D. C. Mitchell, P. L. Milthorp, and M. Yee. 1999. Estimating episodic recharge under different crop/pasture rotations in the Mallee region. Part 2. Recharge control by agronomic practices. Agricultural Water Management **42**:237-249.

Zhu, C. 2000. Estimate of recharge from radiocarbon dating of groundwater and numerical flow and transport modeling. Water Resources Research **36**:2607-2620.

Zouari, K., M. A. Maliki, L. Moumni, and J. F. Aranyossy. 2001. Chemical (Cl) and isotopic (18O, 2H, 3H) study of the unsaturated zone in the arid region of Nefta (South Tunisia). International Atomic Energy Agency, Vienna, Austria.

Biography

John H. Kim was born in Seattle, WA, USA as the first child of Yoonsoo Choi and Hyung Kwon Kim. After a year in USA, John and his family moved to South Korea where John grew up until age of 14. John spent most of his summers in Korea exploring the eastern coastal mountains for interesting flora and fauna, which has engraved deep fascination and appreciation for biology. He received Bachelor of Arts from Department of Ecology and Evolutionary Biology at Princeton University. At Princeton, he worked with Claire Kremen and others in her lab, examining native bee nesting densities in and around Sunflower farms in Central Valley of California. After his Bachelor's, he spent a year researching predator-prey communities in typhoon-disturbed forest landscape in Japan before starting his doctorate with Rob Jackson at Duke University.

AWARDS AND FELLOWSHIPS

NSF/FLAD Award for Collaboration	2010
NSF Doctoral Dissertation Improvement Grant	2008
NSF Graduate Research Fellowship Program	2008-2011
EPA Science to Achieve Results (STAR) Fellowship	2006-2008
Billings Fellowship, Duke	2005-2006
Fulbright Fellowship to Japan	2004-2005
Departmental Highest Honors and Senior Book Prize, Princeton	2004

PUBLICATIONS

Kim, J.H. and R.B. Jackson. *Submitted*. A global analysis of groundwater recharge and the importance of climate, soils and vegetation.

Armas, C.*, J.H. Kim*, T. Bleby, and R.B. Jackson. *In Press*. *Oecologia*. Hydraulic lift enhances nutrient acquisition of a grass species. *These authors contributed equally to the paper

Jayawickreme, D., C.S. Santoni, J.H. Kim, E.G. Jobbágy, and R.B. Jackson. *In press*. *Ecological Applications*. Changes in hydrology and salinity accompanying a century of agricultural conversion in Argentina.

Kim, J.H., N. Williams, and C. Kremen. 2006. *Journal of Kansas Entomological Society*. Effects of cultivation and proximity to natural habitat on native ground-nesting bees in California sunflower fields. **79**: 309-320.