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Climate moderates potential shifts in streamflow from changes in pinyon-juniper woodland cover across the western U.S.

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Abstract

Pinyon-juniper (PJ) cover has increased up to 10-fold in many parts of the western U.S. in the last 140+ years. The impacts of these changes on streamflows are unclear and may vary depending on the intra-annual distribution and amount of precipitation. Given the importance of streamflow in the western U.S., it is important to understand how shifts in PJ woodland cover may produce changes in streamflow across the region's diverse hydroclimates. To this end, we simulated the land surface water balance with contrasting woodland and grassland cover with the Hydrologiska Byråns Vattenbalansavdelning (HBV) model at a 4 km resolution across the distribution of PJ woodlands in the western U.S. We used shifts in evapotranspiration (ET) between woodland and grassland cover as a proxy for potential changes in streamflows. Comparison of HBV model results with paired catchment studies indicated the model reasonably simulated annual decreases in ET with changes from woodland to grassland cover. For the northern and western ecoregions of the PJ distribution in the western U.S. where precipitation predominantly occurs in the winter, HBV simulated a 25 mm (37%) annual decrease in ET with conversion to grassland from woodland. Conversely, in southern ecoregions of PJ distribution with prominent summer monsoons, annual differences in ET were only 6 mm (19%). Our results suggest that only 29% of the PJ distribution, compared to an estimated 45% based on precipitation amount alone, has the potential for meaningful increases in streamflow with land cover change from woodland to grassland.

Key words: pinyon-juniper, woody plant encroachment, evapotranspiration, streamflow, conceptual runoff model, western U.S. land cover change

1. Introduction

Woodland encroachment into shrublands or grasslands is a global phenomenon (Ghersa *et al*., 2002; Lunt *et al*., 2010, Eldridge *et al*., 2011). In much of the western U.S. (i.e. continental U.S. west of 100° longitude) specifically, pinyon-juniper (PJ) woodlands have expanded 10-fold in the last 140+ years and are currently the largest forest cover class in the region (Tausch *et al*., 1981; Miller *et al*., 2005; Romme *et al*., 2009). This expansion is primarily attributed to expansion in grazing (Oliphant, 1968; Miller *et al*., 1994) and fire suppression (Romme *et al*., 2009). PJ cover has also been reduced in some areas due to PJ die-off (Breshears *et al*., 2005) and large-scale PJ removal (e.g. Bureau of Land Management, 2015). Some research suggests a reduction in PJ cover will augment streamflow, whereas others note a lack of evidence for this conclusion (Belsky, 1996; Roundy and Vernon, 1999; Huxman *et al*., 2005; Wilcox *et al*., 2006; Ffolliott and Gottfried, 2012). The importance of streamflow in the semi-arid western U.S. necessitates a better understanding of how shifts in PJ woodland cover may result in changes in streamflow across dissimilar hydroclimates.

Assertions that loss of PJ canopy cover will produce changes in streamflow are typically based on process-based evidence at small (i.e. 1 - 10 m^2) scales. When considering this small scale process-based evidence, we would expect PJ expansion into grass or shrublands to reduce streamflow due to increases in evaporation of canopy-intercepted water and transpiration. For example, tree level PJ interception loss was found to range from 14% to 58% of incoming precipitation (Collings, 1966; Eddleman, 1986; Eddleman and Miller, 1991; Larsen, 1993; Owens *et al*., 2006; Niemeyer *et al*., 2016). Niemeyer *et al*. (2016) simulated water available for runoff using a physically-based numerical model and found that western juniper decreased annual water available for runoff by 121 mm and 155 mm when compared to mountain sagebrush and low sagebrush, respectively. Evapotranspiration is typically higher in PJ woodlands than adjacent grass-dominated areas (Dugas *et al*., 1998; Heilman *et al*., 2009; Liu *et al*., 2010; Banta and Slattery, 2011; Qiao *et al*., 2015). PJ roots have been observed to extend well below 1 m (McCole and Stern, 2007) and as deep as 20 m below the surface (McElrone *et al*., 2004). Recent geophysical surveys in a western juniper stand also showed evidence of subsurface moisture extraction in saprolite 12 m below the surface (Niemeyer *et al*., *accepted*). PJ roots also extend laterally well beyond the canopy edge, typically from approximately one to three times the height of the tree (Hall, 1952; Miller *et al*., 2005; Barrett, 2007). Conversely, although studies have noted a maximum rooting depth of 150 cm for sagebrush and grass, the majority of roots are typically concentrated in the top 30 cm with minimal lateral roots in grasses and lateral roots extending less than the height of vegetation in sagebrush canopies (< 100 cm) (Hull and Klomp, 1974; Sturges and Trlica, 1978). PJ were also observed to transpire and hence extract subsurface moisture in late winter or early spring when grasses are still dormant (Zou *et al*., 2014; Caterina *et al*., 2014). As a result, a greater amount of soil moisture depletion occurs in PJ cover compared to adjacent shrub or grassland cover (Zou *et al*., 2014; Niemeyer *et al*., *accepted*). Based on these differences between PJ, shrub, and grassland cover, we would expect increases (decreases) in PJ cover to decrease (increase) streamflow. In addition to process-based evidence, ranchers in the northern Great Basin cite anecdotal evidence of streamflows either decreasing with PJ expansion (Cockle, 2013) or likewise increasing with PJ removal (Kuhn *et al*., 2007; Merriman, 2008).

Despite both process-based and anecdotal evidence, results from paired catchment studies have not revealed consistent trends in streamflow changes with shifts in PJ cover. Seven paired-catchment studies have assessed how PJ cover changes affect streamflows (Table 1). Of three paired-catchment woody plant removal studies in Arizona, two produced marginal increases in streamflow (3.2 – 9.7 mm year^{-1}) (Robinson 1965; Myrick, 1971; Clary *et al*., 1974; Baker, 1984). The third was a mesquite removal study, a tree that is structurally similar to PJ trees (Huang *et al*., 2009), and produced decreases in streamflow due to increases in grass cover which reduced overland flow, the primary streamflow generation mechanism (Pierini *et al*., 2014). Two PJ paired-catchment studies in Texas revealed PJ removal producing marginal streamflow increases of 2.4 and 3.8 mm year^{-1} (Richardson *et al*., 1979; Wright, 1996). In a third study in Texas, changes in streamflow were inconclusive but a 25.5 mm year^{-1} decrease in ET after PJ removal was observed (Dugas *et al*., 1998). Finally, a paired-catchment PJ removal study in Oklahoma revealed a substantial increase in streamflow of 72 mm year^{-1} (Zou *et al*., 2014). By comparing all seven studies, we see four of the seven woody plant removal studies initially produced statistically significant increases in streamflow in the first two to five years, but the increases in streamflow were typically small (< 10 mm year^{-1}), except the study in Oklahoma. Furthermore, the mesquite study revealed that the presence of mesquite increased runoff ratios during large rain events, due to more bare ground with mesquite cover and greater overland flow (Pierini *et al*., 2014). This counter-intuitive drop in streamflow with reduction of woody plant cover is confirmed by a large scale analysis of streamflow after widespread PJ mortality in the southwestern U.S., which revealed decreases in streamflow after tree die-off, which was attributed to decreases in overland flow due to increases in grass cover (Guardiola-Claramonte *et al*., 2011). In sum, there appears to be no consistent change in streamflow with changes from grassland to woodland cover.

A key aspect of nearly all of these studies is that they were conducted in the Southwest, Midwest, and Texas, where a relatively large portion of precipitation can occur in the summer months (Jul-Sep), as indicated by the monsoon index, defined as the ratio of Jul-Sep precipitation to annual precipitation (Table 1). One paired-catchment study was established in Oregon where the monsoon index is low, and although the results suggested an increase in groundwater with mechanical removal of juniper, the effects on streamflow were inconclusive due to poor streamflow correlations between the treatment and control catchments (Deboodt, 2008). Based on these studies, many have concluded that reduction of PJ species does not have the potential to increase streamflow across the broad region of PJ woodland canopy cover (Hibbert, 1983; Ffolliott and Gottfried, 2012). These conclusions are in contrast to aforementioned small-scale process-based evidence and anecdotal evidence from land owners that reduction in PJ woodland cover increases streamflow.

Discrepancies between the effect of PJ cover change on streamflow between small-scale process-based investigations, anecdotal evidence, and paired-catchment studies may be the result of differences in climate. Many have cited that both soil and climate factors will have a first order control on whether vegetation change will alter deep drainage and/or streamflow (Thurow and Hester, 1997; Wilcox, 2002; Huxman *et al*., 2005; Seyfried and Wilcox, 2006; Wilcox *et al*., 2006). In regards to climate, Hibbert (1983) compared semi-arid vegetation removal studies and asserted that approximately 450 mm of annual precipitation was required to produce an increase in streamflow after vegetation removal. Two of the Arizona paired-catchment studies with marginal increases in streamflow, Beaver Creek and Cibecue Ridge, receive an annual average precipitation of 463 mm and 488 mm, respectively – both above the 450 mm year^{-1} threshold (Figure 1a). However, like much of the southwestern U.S., these areas receive much of their precipitation during the summer monsoons (Table 1, Figure 1b). The timing of precipitation matters because when precipitation occurs in the summer when soil water deficits and evaporative demands are high, more water is lost to the atmosphere regardless of vegetation cover than in the winter when evaporative demand is low (Snyder *et al*., 2004; Seyfried *et al*., 2005). Conversely, when precipitation falls in the winter when evaporative demand is low and the soil moisture deficit may be low as well, more water may replenish the soil profile and percolate beyond the root zone to recharge groundwater and potentially generate streamflow (Comstock and Ehleringer, 1992). Most paired-catchment studies were limited to locations with a larger portion of monsoonal precipitation (Figure 1b) whereas PJ cover spans areas with a large range in a) total precipitation, b) precipitation seasonality (i.e. summer vs. winter dominated) (Figure 1). The monsoon index is much lower (i.e. precipitation is more winter-dominated) in the northern range of PJ woodland cover where there has been only one paired-catchment study and this study did not have a well-established pre-treatment period (Deboodt, 2008). Therefore, there is a potential for PJ cover change in these areas to have a different hydrological effect from what has been observed in areas with higher monsoon index values and to produce a potentially meaningful change in streamflow.

Given the combined importance of precipitation amount and seasonality, coupled with large variations across PJ woodland cover in the western U.S., there is a need to assess how these differences in precipitation climatology may produce changes in streamflow with shifts in PJ cover. The general objective of this study is to understand how spatiotemporal differences in climate control how PJ cover change will potentially alter streamflow across the entire extent of PJ woodlands in the U.S. Our specific objectives are to assess: 1) how differences in the timing and amount of precipitation moderates the potential for PJ cover change to alter water availability via changes in evapotranspiration (ET), 2) how sensitive ET is to changes in plant available water, 3) how climate factors (temperature and precipitation) moderate inter-annual shifts in ET with PJ woodland cover change. The outcome of this work will help to inform land managers where and when PJ cover change has the greatest potential to alter streamflow and prompt further research into how vegetation, climate, and geology interact to control streamflow dynamics in complex terrain.

2. Methods

Overview

To estimate how differences in climate affect how PJ cover change may potentially alter streamflows, we simulated changes in ET with vegetation cover shifts across PJ woodland cover in the western U.S. Streamflow is a function of precipitation minus evapotranspiration (P-ET). Here we use changes in ET as a proxy for potential streamflow changes based on shifts between grassland and woodland cover in the same area, thereby controlling for differences in geology and catchment physiography.

Modelling Approach

The Hydrologiska Byråns Vattenbalansavdelning (HBV) Light model (Seibert, 2005; Seibert and McDonnell, 2010) which is closely based on the original HBV model (Bergström, 1995; Lindström *et al*., 1997), was selected because it is a relatively simple, daily, one-dimensional conceptual hydrological model with explicit snow and soil storage routines (Figure 2). This approach was used to simulate fundamental hydrological fluxes over a large area with a reasonable degree of process accuracy to explore the sensitivity of land cover changes and hydrometeorological variations on changes to ET. The HBV model has been successfully used to test the impacts of changes in land cover on streamflow. For example Seibert and McDonnell (2010) accurately simulated the differences in timing and amount of streamflow between conifer forests and harvested catchments in the Pacific Northwest USA across both small (< 1 km^2) and large (62 km^2) drainages. Likewise Brandt *et al*. (1988) similarly captured changes in streamflow with HBV between forested and clear-cut catchments in central Sweden. And Seibert *et al*. (2010) demonstrated that HBV could capture changes in timing and amount of streamflow in conifer catchments in Eastern Washington USA before and after wildfire. Specific model parameters are listed in Table 2 and based on these previous studies.

The portion of the HBV model structure we used is given in Figure 2. The snow routine develops a snowpack based on precipitation falling below a threshold temperature. Snow melt is computed by a degree-day method. Surface water input is comprised of rainfall and snowmelt that either replenish the root-zone soil storage or bypass the soil storage. When any surface water input exceeds the threshold that is based on the degree of soil storage saturation (Figure 3), it bypasses the soil storage. Note that all surface water input bypasses the soil water storage when the soil is saturated (Figure 3). The total depth of simulated soil water storage is merely the root zone soil water storage or the field capacity multiplied by the assumed depth of roots, referred to here as the soil storage capacity (SC). The model therefore simulates both water that is either mobile and ultimately becomes streamflow or is tightly bound and transpired by plants (e.g. Brooks *et al*., 2010). Since we are interested in water that will potentially generate streamflow, we did not use either of the groundwater flow subroutines and assumed any water that bypasses the soil column, leaves the system (i.e. there is no groundwater-soil water feedback). Furthermore, although subsurface storage is larger than what can be accessed by the roots, we assume any water that bypasses the root zone is available for streamflow and therefore not available for ET.

Actual ET is estimated based on the potential ET and amount of water in soil storage. Canopy interception is not explicitly simulated here but is implicitly included in the soil subroutine via ET estimation (Seibert and McDonnell, 2010). Daily estimates of potential ET in HBV model were estimated using the Hamon equation based on temperature and day length (Hamon, 1961). Daily ET was assumed to equal the potential ET when the ratio of soil water content to SC is greater than 0.5 (Table 2). When the ratio of soil water content to SC is below 0.5, simulated actual ET decreases linearly.

Differences in ET between vegetation play out based on differences in SC between woodland and grassland cover, with a larger SC for woodland and therefore a greater amount of SC storage available for ET. We chose to only simulate the soil storage available to plants because we can assume that the water storage change from year to year is negligible and therefore any water that percolates past the rooting depth would be available for streamflow (Seyfried and Wilcox, 2006). Differences in rooting depth exist between shallow rooted grasses and forbs that use moisture closer to the surface compared to PJ species that use both shallow and deep moisture pools (Walker and Noy-Meir, 1982; Pelaez *et al*., 1994; Jackson, 1996; Ryel *et al*., 2008; Breshears *et al*., 2009; Flerchinger and Seyfried, 2014; Niemeyer et al., *accepted*). For example, Seyfried and Wilcox (2006) found that the rooting depth of grass and forb post-fire vegetation was 140 cm, 60 cm less than pre-fire vegetation of dense shrubs, which resulted in a shift in SC from 175 mm to 250 mm. Similarly, Williamson *et al*. (2004) found that water at a tension available to roots persisted from 100 to 260 cm in grassland vegetation, but no water available to roots persisted through the summer as deep as 150 cm in chaparral. To simulate change from woodland to grassland vegetation for this relatively simple sensitivity assessment, we simulated SC of 200 mm in woodland and 100 mm in grassland which represents reasonable general differences between the two cover types. Although PJ cover often replaces sagebrush or other shrubs, not grasslands, we feel a shift in SC of 100 mm is a good general approximation for shifts in rooting depth between PJ cover and shrub or grasslands for the purposes of this investigation. In reality, SC varies across both vegetation types and subsurface characteristics, therefore we also assessed the sensitivity of ET across incremental differences in SC.

HBV does not explicitly model overland flow generated by either infiltration excess or saturation excess overland flow, but can still approximate shifts in ET from changes in land cover. This model limitation may seem like an impediment to reasonable estimation of potential changes in ET since overland flow may be the primary streamflow generation process in PJ dominated areas in the southwestern U.S. (Guardiola-Claramonte *et al*., 2011; Pierini *et al*., 2014). However, this does not impede the reasonable estimation of shifts in ET across PJ cover for several reasons. First, in areas with summer monsoons HBV typically does underestimate runoff, however this underestimation is usually small and HBV still simulates the timing and magnitude of runoff from summer events relatively well (Konz *et al*., 2007; Norman *et al*., 2010; Jia and Sun, 2012; Hong *et al*., 2014; Hilgert *et al*., 2015). Second, we are principally concerned with water that bypasses the soil water storage via any process, whether it is overland flow, shallow subsurface flow, or groundwater recharge. In other words, we are concerned with water that is not available for plant water uptake. In HBV, the portion of water that bypasses soil water storage increases with the degree of soil saturation (Figure 3). Likewise an increased degree of soil saturation increases streamflows generated from precipitation events via overland flow (Bonell and Gilmour, 1978; Pierini *et al*., 2014), shallow subsurface flow (Freer *et al*., 2002; McNamara *et al*., 2005, Tromp-van Meerveld and McDonnell, 2006), or groundwater (McGlynn *et al*., 2004; Gabrielli *et al*., 2012). The increase in overland flow with degree of soil saturation even can occur in systems in the southwest U.S. dominated by infiltration excess overland flow (Pierini *et al*., 2014). We can therefore reasonably estimate shifts in ET in systems with diverse streamflow generation mechanisms that are affected by the degree of soil storage saturation. Third, since some water bypasses the soil storage even when the soil saturation is low (Figure 3), potential runoff generation during summer monsoons when soils are dry can still be reasonably approximated. For example, in HBV with a summer monsoon situation with the soil at 0.3 soil saturation, the runoff ratio would be 0.027 (Figure 3). In the southwestern U.S. with predominant summer monsoons, annual runoff ratios are typically very low, ranging from 0.002 to 0.15 (Gallo *et al*., 2013; Chang *et al*., 2014) and summer ratios can be lower than in the winter (Clary *et al*., 1974). Due to this, although with a large portion of precipitation occurring in the summer sites in the southwestern U.S., streamflow can principally occur in the winter (Clary *et al*., 1974). Taken together, the model effectively simulates the removal of excess water that is not stored in the soil profile and hence can be used to assess annual changes in ET resulting from land cover shifts.

Model Implementation

We simulated daily snow and soil water dynamics at a 4 km resolution with the HBV model across the full range of PJ cover in the western U.S. (Figure 1). PJ cover in the western U.S. was based on the 1 km resolution national land cover map from USDA Forest Service (USFS) and US Geological Survey (USGS) (2002). Daily 4 km PRISM mean temperature and precipitation from 1981 to 2010 (Daly *et al*., 1994) were used to drive the model. PJ land cover overlapped with 33956 of the 4 km climate resolution grid cells. We simulated the daily water balance for both woodland and grassland vegetation with the HBV model in each of these cells. To estimate potential change in streamflow we calculated woodland and grassland cover P-ET, and then subtracted woodland P-ET from grassland P-ET. In this calculation, the precipitation terms fall out, resulting in woodland ET minus grassland ET ($\Delta ET_{tree\text{-}grass}$). This term intuitively makes sense since shifts in streamflow from changes in vegetation cover is principally driven by ET (Macdonald and Stednick, 2003; Huxman *et al*., 2005; Hubbart *et al*., 2007). For the SC sensitivity analysis, we simulated daily ET for the 30 years across a range of SC from 50 to 250 mm (in 10 mm intervals) that corresponds to the range of grass and tree cover previously described. Since for this analysis each cell required 21 different simulations, compared to two different simulations for the grassland and woodland analysis, we used systematic sampling to reduce the number of grid cells for this analysis to 2000.

To evaluate model performance, we compared measured streamflow or groundwater recharge from previous studies with the average simulated $\Delta ET_{tree\text{-}grass}$ across all 30 years. There exists a combined five paired catchment and plot studies that assessed changes between grassland and woodland cover that overlap with our simulated PJ cover which allowed for model comparison: Beaver Creek in Arizona, Reynolds Creek Experimental Watershed plot study in Idaho, Blackland Prairie in Texas, and two studies in Seco Creek in Texas (Table 1). These sites represent a range of conditions since the increase in annual streamflow or deep drainage for these studies ranged from 2.4 to 60 mm and the monsoon index ranged from 0.08 to 0.32. Although some of these studies were not in land cover shifts from PJ woodland to grassland, the land cover shifts in all five of these studies are from deep-rooted woody plant cover to herbaceous plant cover. We did not compare our simulations to the other studies in Table 1 because they either A) did not overlap the PJ cover extent from the USGS and USDA (2002) data or B) in the case of Cibecue Ridge the two manuscripts on the study did not give values or gave conflicting values for changes in streamflow.

To evaluate if ET differed between woodland and grassland cover, we conducted a two-tailed t-test between the 30 years of annual ET for the woodland and grassland synthetic datasets. For each PJ cover cell, we determined statistical difference in ET between woodland and grassland cover based on a p-value of 0.1. We also compared the percent change in ET between woodland and grassland cover with the following equation:

$$\%\ change\ in\ ET = \left[ET_{tree} - ET_{grass}\right]/ET_{grass}$$

We also used linear regression to evaluate the correlation between the average annual precipitation and monsoon index on the mean $\Delta ET_{tree\text{-}grass}$. This analysis was done across the systematically sampled number of cells (2000 total). Furthermore, we evaluated the relationship between annual precipitation and temperature on $\Delta ET_{tree\text{-}grass}$. For the 30 years of data, we regressed $\Delta ET_{tree\text{-}grass}$ against annual precipitation and temperature. To quantify the sensitivity of ET to SC, for each year we calculated the slope of the linear regression of SC (50 to 250 mm) and ET. We then averaged each slope across all 30 years.

Finally, we calculated how regional differences in climate at the EPA Level III ecoregion classifications (Bailey, 1983) produce differences in $\Delta ET_{tree\text{-}grass}$. To explore how within-Level III differences due to complex terrain impact $\Delta ET_{tree\text{-}grass}$, we also compared $\Delta ET_{tree\text{-}grass}$ across Level IV ecoregion classifications in southern Idaho. For both ecoregion analyses, to determine statistical differences in $\Delta ET_{tree\text{-}grass}$ across regions, we compared annual $\Delta ET_{tree\text{-}grass}$ across 30 years with a pair-wise t-test. Since we were testing multiple hypotheses, to reduce the chance of obtaining false positive results, we adjusted p-values with the Bonferroni correction method. We assumed statistical difference for the t-tests at p-value of less than 0.0001. We used a p-value of less than 0.0001 because of the higher rate of statistical similarity between ecoregions with a p-value of 0.1.

3. Results

Model Evaluation

Our simulations reasonably represented the observations from the five studies used in the model performance evaluation despite the relative simplicity and relatively coarse spatial resolution of the modeling approach. In the Beaver Creek study the observed increase in annual streamflow (9 year average) after juniper removal was 9.7 mm year^{-1} (Table 1). In comparison, the simulated $\Delta ET_{tree\text{-}grass}$ in the grid cell of their study was 7.6 mm year^{-1} for the 30-year average climate data used, assuming static land cover conditions after conversion to grassland. In Texas, Richardson *et al*. (1979) observed a 2.4 mm year^{-1} increase in streamflow after PJ removal and the average simulated $\Delta ET_{tree\text{-}grass}$ was 6.9 mm year^{-1}. At Seco Creek in Texas, Wright (1996) observed a 3.8 mm year^{-1} average increase in streamflow, and Dugas *et al*. (1998) observed a $\Delta ET_{tree\text{-}grass}$ of 25.5 mm year^{-1}, compared to simulated $\Delta ET_{tree\text{-}grass}$ of 3.1 mm year^{-1}. Finally, in the study with the lowest monsoon index, Seyfried and Wilcox (2006) observed, in an area after a fire burned dense shrubs, a gain of 60 mm year^{-1} compared to a simulated $\Delta ET_{tree\text{-}grass}$ of 64.6 mm year^{-1} in this same area.

Differences in Woodland and Grassland ET

Mapping distributed $\Delta ET_{tree\text{-}grass}$ shows clear spatial trends (Figure 4). First, we see in the northern and western areas of the PJ range - California, Oregon, Idaho, Utah, Nevada, and Colorado – are the only states with large areas of $\Delta ET_{tree\text{-}grass}$ ranging from 30 mm year^{-1} to upwards of 60 mm year^{-1} (Figure 4). The areas with statistically different ET between grassland and woodland cover are likewise predominantly in these states (Figure 4 inset). Second, we see that areas in the southern parts of PJ distribution – Arizona, New Mexico, and Texas – are dominated by differences in ET of less than 20 mm year^{-1} (Figure 4). Similarly, almost no cells in these areas have a statistically significant change in ET (Figure 4 inset). The map of percent change in ET shows many sites of greater than 100% increase in ET with shifts from grassland to woodland cover (Figure 5). Similar to mapped $\Delta ET_{tree\text{-}grass}$, these large increases in percent change in ET are predominantly in the northern range of PJ cover. Conversely the southern distribution of PJ cover is dominated by percent change in ET of less than 50% (Figure 5).

Comparing $\Delta ET_{tree\text{-}grass}$ across Level III ecoregion classifications reveals some clear trends (Figure 6). First, both the coastal ecoregions (Marine and Mediterranean) had the first and second highest $\Delta ET_{tree\text{-}grass}$ and were both statistically different than ecoregions with lower $\Delta ET_{tree\text{-}grass}$ (Figure 6, Table 3). Second, the average $\Delta ET_{tree\text{-}grass}$ for northern temperate ecoregions (Temperate Steppe Mountains, Temperate Desert, and Temperate Desert Mountains) ranged from 28.9 to 29.9 mm year^{-1} and based on two-tailed t-tests were statistically similar to each other but greater than other ecoregions (except the two coastal ecoregions) (Table 3). Average $\Delta ET_{tree\text{-}grass}$ in Prairie, Subtropical Steppe, Subtropical Desert, and Temperate Steppe were all statistically similar populations and ranged from 3.1 to 9.3 mm year^{-1}. Finally, $\Delta ET_{tree\text{-}grass}$ differences in Subtropical Steppe Mountains and Temperate Steppes was statistically similar at 9.1 and 9.3 mm year^{-1} respectively.

Although the average $\Delta ET_{tree\text{-}grass}$ are very similar for the Level III ecoregion classifications that exist for PJ cover in Idaho - Temperate Desert and Temperate Steppe Mountains (Figure 6, Table 3), when further parsed into Level IV ecoregion classifications, there are clear spatial differences related to complex terrain in the region (Figure 6). Specifically, higher elevation ecoregions – High Forests and Shrublands, Semiarid Uplands, Semiarid Foothills, Rockies Cold Valleys, Partly Forested Mountains, Wasatch Montane Zone, and

Wasatch Semiarid Foothills – were all statistically similar. These statistically similar Level IV ecoregions also had the highest median $\Delta ET_{tree\text{-}grass}$ ranging from 55.4 to 64.9 mm year^{-1}. Differences in $\Delta ET_{tree\text{-}grass}$ for other ecoregions – High Lava Plateau, Semiarid Hills and Low Mountains, Owyhee Uplands and Canyons, and Wet Valleys- were statistically less than the highest group but still had median $\Delta ET_{tree\text{-}grass}$ exceeding 40 mm year^{-1}. Conversely the lower elevation ecoregions - Upper Snake River Plain, Mountain Home Uplands, and Eastern Snake River Plains - had the lowest median differences in $\Delta ET_{tree\text{-}grass}$ ranging from 15.9 to 23.5 mm year^{-1}.

$\Delta ET_{tree\text{-}grass}$ and Timing and Amount of Precipitation

We compared the simulated differences in grassland and woodland ET to Hibbert's (1983) threshold of 450 mm of annual gross precipitation necessary to produce in increase of streamflow due to conversion from deep to shallow rooted vegetation (Table 4). Based on this gross precipitation threshold, conversion from deep to shallow rooted vegetation would not have the potential to increase in streamflow in over 54% of PJ cover area. Conversely, 46% of PJ cover area could produce an increase in streamflow. In contrast, based on significance of t-tests between woodland and grassland ET differences ($\Delta ET_{tree\text{-}grass}$) at the p-value < 0.1 level, 70.6% of PJ cover area would likely not show an increase in streamflow and 29.4% may see increases streamflow. Many grid cells with precipitation greater than 450 mm year^{-1} and hence having the potential to increase streamflows do not meet the p-value less than 0.1 criteria, and *vice versa* (Table 4). In total, only 13.9% of the grid cells met both the precipitation and statistical significance criteria with a $\Delta ET_{tree\text{-}grass}$ of 52.6 mm year^{-1}, whereas 32.2% of the grid cells had precipitation exceeding 450 mm year^{-1} but with a p-value exceeding 0.1.

By plotting $\Delta ET_{tree\text{-}grass}$ vs. monsoon index, we see a clear trend. As monsoon index increases, $\Delta ET_{tree\text{-}grass}$ decreases (Figure 7). The slope is significant ($p < 0.0001$) for all (Figure 7a - black line), low (Figure 7a - red line), and high precipitation grid cells (Figure 7a –blue line). When $\Delta ET_{tree\text{-}grass}$ is normalized by annual precipitation at each grid cell, the differences between wetter and drier grid cells all but disappear (Figure 7b).

Maps of trends between $\Delta ET_{tree\text{-}grass}$ and both precipitation and temperature also reveal some clear spatial patterns. The majority of the grid cells (73%) exhibited a positive relationship between precipitation and $\Delta ET_{tree\text{-}grass}$ (Figure 8a), with the regression suggesting an increase in $\Delta ET_{tree\text{-}grass}$ of 0.088 units for each unit of precipitation increase. This means at locations in these grid cells, wetter years produced a greater $\Delta ET_{tree\text{-}grass}$. Higher values (> 0.2 mm mm^{-1}) occur at grid cells in the middle Rockies (Utah and Colorado), Sierras (California and Western Nevada), central Arizona, and parts of Oregon and Idaho (Figure 8a). Many of these grid cells had regression slopes exceeding 0.4 mm mm^{-1} – meaning for a 1 mm increase in precipitation on average resulted in a 0.4 mm year^{-1} increase of $\Delta ET_{tree\text{-}grass}$. The only areas with negative trends less than -0.2 mm mm^{-1} were in parts of Nevada, Arizona, New Mexico, and Texas (Figure 8a).

Patterns in $\Delta ET_{tree\text{-}grass}$ vs. temperature were less consistent across the PJ cover range (Figure 8b). The median trend was 1.4 mm $°\text{C}^{-1}$, meaning the majority of the grid cells (56%) increased $\Delta ET_{tree\text{-}grass}$ with increases in temperature. Groups of cells in California, eastern Arizona, New Mexico, Colorado, and Texas showed larger positive trends (> 20 mm $°\text{C}^{-1}$). This means for a 1 °C increase in temperature, $\Delta ET_{tree\text{-}grass}$ was on average 20 mm year^{-1} greater. Conversely, some grid cells in western Arizona, in the western and northern Great Basin (California, Oregon, Idaho), and parts of Colorado and Texas showed a negative relationship between temperature and (Figure 8b), indicating decreasing $\Delta ET_{tree\text{-}grass}$ values with increases in temperature.

SC Sensitivity

Comparing the sensitivity of ET to changes in SC across the range of monsoon index values and precipitation revealed some clear trends. First, we see that both the amount and seasonality of precipitation (monsoon index) control how SC affects sensitivity of ET to SC (Figure 9a). The grid cells where ET is most sensitive to changes in SC are both wetter and characterized by more winter-dominated precipitation. Conversely, grid cells where ET is less sensitive to changes in SC are both drier and more monsoonally-influenced regions with a larger proportion of summer precipitation. By mapping the ET sensitivity to SC across the PJ cover, we see that grid cells in the northern and western areas of PJ cover are most sensitive to SC, whereas grid cells in the southern regions are less sensitive (Figure 9c). The notable exceptions are PJ grid cells in Texas which were relatively sensitive to SC (Figure 9c).

4. Discussion

Based on our simulations, shifts from PJ woodland to grassland cover will decrease ET by 38 mm year^{-1} at 29.4% of grid cells that showed statistically significant changes in ET between the two cover types. The implication is changes from woodland to grassland cover in these areas will potentially increase in streamflow. These increases mainly occur on the northern and western portions of the PJ range, primarily in California, Oregon, Idaho, Utah, Nevada, and Colorado (Figure 4). These are areas where precipitation primarily occurs in the winter when evaporative demand is low (Figure 1). Our method of determining if streamflow will potentially change with PJ cover shifts is more conservative than the simple precipitation-based threshold approach detailed by Hibbert (1983). Based on the annual precipitation cutoff of 450 mm, 46% of the grid cells would have a significant change in streamflow as a result of shifts between PJ woodlands and grassland. This is 156% greater than our estimate of the number of grid cells that would have a significant change in streamflow based on statistical difference between woodland and grassland cover ET (Table 4). Furthermore, our modeling approach was reasonably accurate, covers a wide range of PJ cover, and represents the predominant processes affecting the water balance in these systems.

Clearly both the amount and timing of precipitation are important in determining if PJ cover change will have a meaningful impact on streamflow. Hibbert (1983) was correct in that precipitation amount is important as was confirmed in this study (Figure 9a). This study however indicates that the seasonality of precipitation is just as important (Figure 7, 9a). This conclusion is supported by the empirical investigation of Clary *et al*. (1974) in Arizona, which indicated that although a large portion of the precipitation occurred in the summer, of the 21 years that produced streamflow in a watershed with PJ cover, streamflow for 15 of those years only occurred in the winter. Likewise, a plot-scale semi-arid water balance study using lysimeters showed that increased groundwater recharge primarily occurred when a large amount of snowmelt was available for infiltration (Gee *et al*., 1994). A long-term water balance study of a small semi-arid catchment, also suggested that when a large portion of the precipitation occurs when the evaporative demand is low (i.e. in the winter), there is a greater chance that infiltration will produce deep drainage below the rooting zone and generate streamflow (Chauvin *et al*., 2011). In summary, the results of this study further support the finding that the timing of precipitation is as important as the amount of precipitation when predicting the sensitivity of streamflow to vegetation cover changes.

Wilcox *et al*. (2006) and Huxman *et al*. (2005) both identify climate and soil depth as important factors in determining if vegetation cover change will alter streamflow. Similarly, our research demonstrated that although climate was clearly important, depth of soil and how much subsurface moisture is available to plants is likely also important. Shifts in SC drove differences in ET across the two land cover types (Figure 9a). However, the impact of shifts in SC was modulated by differences in the amount and timing of precipitation (Figure 9a). Specifically, in sites with more winter-dominated precipitation, ET was more sensitive to shifts in SC. Therefore, despite the importance of soil factors in controlling $\Delta ET_{tree\text{-}grass}$, climate is likely even more important than subsurface characteristics. Confirming the greater importance of climate compared to soil characteristics, Wine *et al*. (2015) demonstrated that climate was more important than soil available water and rooting depth in determining deep drainage.

Despite the reasonable comparison of observed changes in streamflow and groundwater recharge and simulated changes in ET from woodland to grassland conversion, this study highlights the need for process-based studies to understand the ecohydrological process that drive shifts in streamflow in semi-arid systems. HBV model is a highly simplified representation of the hydrological system and does not fully represent hydrological processes that determine how streamflow will change with changes in woodland cover. For example, in our study while $\Delta ET_{tree\text{-}grass}$ was always positive (i.e. there was always a greater ET in woodland cover than grassland cover), Guardiola-Claramonte *et al*. (2011) in the southwestern U.S. observed 30% - 80% decreases in streamflow for watersheds with 11% - 21% PJ die-off. One hypothesis they posited for this counter-intuitive response was that PJ die-off increases herbaceous cover which reduces overland flow – a primary runoff generation mechanism in much of the southwestern U.S. A similar result was found by Pierini *et al*. (2014) when comparing two small catchments

(1.1 ha) in Arizona where runoff was greater in areas without woody vegetation for storms smaller than 5 mm (< 0.1 mm difference), similar for storms between 5 and 30 mm, and greater in woody vegetation-dominated catchments compared to grass-dominated catchments by on average 1.4 mm event^{-1} for events exceeding 30 mm in precipitation. They likewise posited that with more woody vegetation, there is less herbaceous cover thereby increasing overland flow during larger events. A plot scale study in Idaho revealed that indeed areas with western juniper, compared to areas where juniper were removed, generated more overland flow due to decreases in herbaceous cover (Pierson *et al*., 2007). It should be noted that although our model always simulated a decrease in ET and thereby potential increase in streamflow with shifts from woodland to grassland cover, changes in streamflow in Guardiola-Claramonte *et al*. and Pierini *et al*. are marginal and were observed in the southwestern U.S. where similarly marginal changes in streamflow between woodland and grassland ET were simulated. HBV model does not explicitly incorporate these near-surface infiltration-runoff processes such as attenuation of overland flow with increased grass cover. Further observational and/or modeling studies could further elucidate how runoff mechanisms (surface vs. groundwater) control the impact of PJ cover change on streamflow.

Another key knowledge gap is the actual root zone depth in PJ woodland cover. Field-based work reveals PJ trees can access moisture deep in the subsurface. Breshears *et al*. (2009) showed piñon pine (*Pinus edulis*), and one-seed juniper (*Juniperus monosperma*) accessing soil moisture up to 3 m depth. A study in western juniper revealed those trees accessing moisture in saprolite and weathered bedrock at depths of up to 12 m (Niemeyer *et al*., *accepted*). Further studies could quantify the timing and depth of soil moisture extraction in both PJ woodland and grassland cover through water isotope studies or geophysical methods. In addition, questions remain as to the extent of subsurface storage, especially across complex subsurface layers such as saprolite and weathered bedrock (Befus *et al*., 2011; Holbrook *et al*., 2013). Likewise, geophysical and other process-based studies could elucidate the extent of subsurface moisture storage across diverse PJ cover terrain. Finally, SC in HBV model implicitly includes canopy storage. Tree level interception loss is often greater than 35% in PJ woodlands (Collings, 1966; Eddleman, 1986; Eddleman and Miller, 1991; Larsen, 1993; Owens *et al*., 2006; Niemeyer *et al*., 2016), therefore a model that explicitly incorporates canopy storage could further elucidate the ecohydrologic processes that drive changes in streamflow with changes in PJ cover.

Most assertions that PJ cover change would not alter streamflow are based on paired-catchment studies. Paired-catchment studies for PJ cover, however, have been predominantly conducted in areas where a large portion, if not the majority, of the precipitation occurring in the summer (Figure 1b). These studies are therefore not representative of PJ cover in the entire western U.S. The one paired-plot study that overlapped our PJ cover range and was in a low monsoon-index area in southwestern Idaho, revealed an 60 mm average annual increase in groundwater recharge after grass and forb regeneration following the burning of dense shrubs (Seyfried and Wilcox, 2006). Also, there was one paired-catchment study in Oregon, but the pre-treatment correlation of streamflow between the control and treated catchments was not statistically significant, preventing a robust post-treatment analysis of streamflow (Deboodt, 2008). However, this study did observe an increase in groundwater persistence and days of streamflow in the dry season (June – November) in the catchment where PJ trees were felled. These two studies corroborate the analysis of our ecohydrological simulations that catchments in low-monsoon index locations are more likely show changes in streamflow with shifts between woodland and grassland cover. To further test this hypothesis, two-paired catchment studies are underway in the northern PJ cover/low-monsoon index range: the South Mountain paired-catchment study in southwestern Idaho by the USDA-ARS in Boise, Idaho and the Porter Canyon Experimental Forest administered by the USDA-ARS in Reno, Nevada (Figure 1). Serendipitously, both studies began felling PJ trees in 2015. Based on our simulated $\Delta ET_{tree\text{-}grass}$ at each site, the catchment where PJ were felled at South Mountain is expected to produce an increase in streamflow of approximately 69 mm year^{-1}, whereas Porter Canyon should experience an increase of roughly 16 mm year^{-1}. Considering how small the latter is, gains in streamflow at the Porter Canyon site may be marginal or only increase in wetter years. After five to ten years of post-treatment data have been collected, researchers will be able to further verify the results of this study or identify shortcomings in this simple simulation-based approach that will further advance our knowledge of semi-arid hydrology. Furthermore, it is critical that these studies continue to collect data over the long-term. Roundy *et al*. (2014) observed after western juniper removal, although initial increases in soil moisture occurred at 30 cm, the soil moisture regime returned to pre-removal status within 3 years due to vegetation recovery. Long-term studies like the two USDA-ARS studies can elucidate if and how long-term changes in streamflow persist in winter-precipitation dominated environments and provide critical information to land managers seeking to optimize ecosystem services in semi-arid systems.

5. Conclusions

Although there are disagreements as to whether PJ cover change will alter streamflow, this is likely due to the majority of paired-catchment PJ removal studies in Arizona and Texas where precipitation is more synchronous with periods of high evaporative demand. PJ woodlands in the southwestern U.S. receive a large portion of their precipitation in the summer, when evaporative demand is high and tree or herbaceous cover will produce large evaporative and transpiration losses. Our simulations revealed that indeed differences between woodland and grassland ET are minimal in the southwestern U.S. and other areas with higher portions of precipitation occurring in the summer. Conversely, in areas in the northern and western PJ range where precipitation falls when the evaporative demand is low, ET often decreased substantially with shifts from woodland to grassland cover. The amount of precipitation is also clearly important. Locations with low precipitation, even if the precipitation predominantly occurs in the winter, yielded small changes in ET between the two cover types.

Our study reveals important information for managers. First, we found that by using the previous criteria of Hibbert (1983) of 450 mm annual precipitation to determine of PJ cover change will alter streamflow is likely not accurate over the full range of PJ cover. If managers are concerned with changes in streamflow after shifts between woodland and grassland cover, they should evaluate whether there is both substantial amount of precipitation and whether the precipitation occurs when evaporative demand is low (i.e. in the winter). If this is true, then there is an increased chance that PJ cover change will alter streamflow. Second, many of the grid cells with a significant difference in ET between grassland and woodland cover occur in the northern and western range of PJ cover, where warmer years produced decreased differences in ET between woodland and grassland cover (Figure 8b). This means that with warmer temperatures under future climate change, potential changes in streamflow with shifts from woodland to grassland cover may be diminished. Although these shifts in future streamflow will also depend on concomitant changes in the precipitation regime. Therefore in the future, any potential increases in streamflow with PJ cover change may be diminished or lost altogether.

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References

Bailey RG. 1983. Delineation of ecosystem regions. *Environmental Management* **7**: 365–373. DOI: 10.1007/BF01866919.

Baker Jr MB. 1984. Changes in streamflow in an herbicide-treated pinyon-juniper watershed in Arizona. *Water Resources Research* **20**: 1639–1642. DOI: 10.1029/WR020i011p01639.

Banta JR, and Slattery RN. 2011. Effects of brush management on the hydrologic budget and water quality in and adjacent to Honey Creek State Natural Area, Comal County, Texas, 2001-10. US Department of the Interior, US Geological Survey.

Barrett H. 2007. Western juniper management: a field guide. Oregon Watershed Enhancement Board.

Belsky AJ. 1996. Viewpoint: Western Juniper Expansion: Is It a Threat to Arid Northwestern Ecosystems? *Journal of Range Management* **49**: 53. DOI: 10.2307/4002725.

Befus KM, Sheehan AF, Leopold M, Anderson SP, Anderson RS. 2011. Seismic constraints on critical zone architecture, Boulder Creek watershed, Front Range, Colorado. *Vadose Zone Journal* **10**: 915–927. DOI: 10.2136/vzj2010.0108.

Bergström S, Singh VP. 1995. The HBV model. *In* Computer models of watershed hydrology. V.P. Singh, editor. 443–476.

Bonell M, Gilmour DA. 1978. The development of overland flow in a tropical rainforest catchment. *Journal of Hydrology* **39**: 365–382. DOI: 10.1016/0022-1694(78)90012-4.

Brandt M, Bergström S, Gardelin M. 1988. Modeling the effects of clearcutting on runoff – examples from Central Sweden. *Ambio* **17**: 307–313.

Breshears DD, Cobb NS, Rich PM, Price KP, Allen CD, Balice RG, Romme WH, Kastens JH, Floyd ML, Belnap J, Anderson JJ, Myers OB, Meyer CW. 2005. Regional vegetation die-off in response to global-change-type drought. *Proceedings of the National Academy of Sciences* **102**: 15144–15148. DOI: 10.1073/pnas.0505734102.

Breshears DD, Myers OB, Barnes FJ. 2009. Horizontal heterogeneity in the frequency of plant-available water with woodland intercanopy–canopy vegetation patch type rivals that occuring vertically by soil depth. *Ecohydrology* **2**: 503–519. DOI: 10.1002/eco.75.

Brooks JR, Barnard HR, Coulombe R, McDonnell JJ. 2010. Ecohydrologic separation of water between trees and streams in a Mediterranean climate. *Nat*ure *Geosciences* **3**: 100–104. DOI: 10.1038/ngeo722.

Bureau of Land Management. 2015. Notice of Intent To Prepare an Environmental Impact Statement for the Proposed Bruneau-Owyhee SageGrouse Habitat Project, Owyhee County, Idaho.

Caterina GL, Will RE, Turton DJ, Wilson DS, Zou CB. 2014. Water use of Juniperus virginiana trees encroached into mesic prairies in Oklahoma, USA. *Ecohydrology* **7**: 1124–1134. DOI: 10.1002/eco.1444.

Chang H, Johnson G, Hinkley T, Jung IW. 2014. Spatial analysis of annual runoff ratios and their variability across the contiguous US. *Journal of Hydrology* **511**: 387–402. DOI: 10.1016/j.jhydrol.2014.01.066.

Chauvin G, Flerchinger GN, Link T, Marks D, Winstral A, Seyfried MS. 2011. Long-term water balance and conceptual model of a semi-arid mountainous catchment. *Journal of Hydrology* **400**: 133-143. DOI: 10.1016/j.jhydrol.2011.01.031.

Clary WP, Baker MB, O'Conell PF, Johnsen TN, Cambell RE. 1974. Effects of pinyon-juniper removal on natural resource products and uses in Arizona. *USDA Forest Service Research Paper, Rocky Mountain Forest and Range Experiment Station* **RM-128**. Fort Collins, CO, USA.

Cockle R. 2013. Juniper invasion marring Oregon's high desert habitat. *The Oregonian*.

Collings MR. 1966. Throughfall for summer thunderstorms in a juniper and pinyon woodland, Cibecue Ridge, Arizona. U.S. Geological Survey.

Comstock JP, Ehleringer JR. 1992. Plant adaptation in the Great Basin and Colorado Plateau. *Western North American Naturalist* **52**: 195–215.

Daly C, Neilson RP, Phillips DL. 1994. A statistical-topographic model for mapping climatological precipitation over mountainous terrain. *Journal of Applied Meteorology* **33**: 140–158. DOI: 10.1175/1520-0450(1994)033<0140:ASTMFM>2.0.CO;2.

Deboodt TL. 2008. Watershed response to western juniper control. Oregon State University. 143 pp.

Dugas WA, Hicks RA, Wright P. 1998. Effect of removal of *Juniperus ashei* on evapotranspiration and runoff in the Seco Creek watershed. *Water Resources Research* **34**: 1499–1506. DOI: 10.1029/98WR00556.

Eddleman LE. 1986. Canopy interception of precipitation. Water Resources Research Institute, Oregon State University.

Eddleman LE, Miller PM. 1991. Potential impacts of western juniper on the hydrologic cycle. *In* Proceedings, symposium in ecology and management of riparian shrub communities. 29–31.

Eldridge DJ, Bowker MA, Maestre FT, Roger E, Reynolds JF, Whitford WG. 2011. Impacts of shrub encroachment on ecosystem structure and functioning: towards a global synthesis. *Ecology Letters* **14**: 709–722. DOI: 10.1111/j.1461-0248.2011.01630.x.

Ffolliott PF, Gottfried GJ. 2012. Hydrologic processes in the pinyon-juniper woodlands: A literature review. *USDA, Forest Service, Rocky Mountain Research Station General Technical Report* **RMRS-GTR-271**. Fort Collins, CO, USA.

Flerchinger GN, Seyfried MS. 2014. Comparison of methods for estimating evapotranspiration in a small rangeland catchment. *Vadose Zone Journal* **13**: 1-11. DOI: 10.2136/vzj2013.08.0152.

Freer J, McDonnell JJ, Beven KJ, Peters NE, Burns DA, Hooper RP, Aulenbach B, Kendall C. 2002. The role of bedrock topography on subsurface storm flow. *Water Resources Research* **38**: 5-1-5-16. DOI: 10.1029/2001WR000872

Gabrielli CP, McDonnell JJ, Jarvis WT. 2012. The role of bedrock groundwater in rainfall–runoff response at hillslope and catchment scales. *Journal of Hydrology* **450–451**: 117–133. DOI: 10.1016/j.jhydrol.2012.05.023.

Gallo EL, Brooks PD, Lohse KA, McLain JE. 2013. Temporal patterns and controls on runoff magnitude and solution chemistry of urban catchments in the semiarid southwestern United States. *Hydrological Process* **27**: 995–1010. DOI: 10.1002/hyp.9199.

Gee GW, Wierenga PJ, Andraski BJ, Young MH, Fayer BJ, Rockhold ML. 1994. Variations in water balance and recharge potential at three western desert sites. *Soil Science Society of America Journal* **58**: 63–72. DOI: 10.2136/sssaj1994.03615995005800010009x.

Ghersa CM, de la Fuente E, Suarez S, Leon RJ. 2002. Woody species invasion in the Rolling Pampa grasslands, Argentina. *Agriculture, Ecosystems, & Environment* **88**: 271–278. DOI: 4101/10.1016/S0167-8809(01)00209-2.

Guardiola-Claramonte M, Troch PA, Breshears DD, Huxman TE, Switanek MB, Durcik M, Cobb NS. 2011. Decreased streamflow in semi-arid basins following drought-induced tree die-off: A counter-intuitive and indirect climate impact on hydrology. *Journal of Hydrology* **406**: 225–233. DOI: 10.1016/j.jhydrol.2011.06.017.

Hall MT. 1952. Variation and hybridization in Juniperus. *Annals of the Missouri Botanical Garden* **39**: 1–64.

Hamon WR. 1961. Estimating potential evapotranspiration. *Journal of the Hydraulics Division* **87**: 107–120.

Heilman JL, McInnes KJ, Kjelgaard JF, Owens MK, Schwinning S. 2009. Energy balance and water use in a subtropical karst woodland on the Edwards Plateau, Texas. *Journal of Hydrology* **373**: 426–435. DOI: 10.1016/j.jhydrol.2009.05.007.

Hibbert AR. 1983. Water yield improvement potential by vegetation management on western rangelands. *Water Resources Bulletin* **19**: 375–382.

Hilgert S, Wagner A, Fuchs S. 2015. Future changes in flash flood frequency and intensity of the Tha Di River (Thailand) based on rainfall–runoff modeling and advanced delta change scaling. *Hydrology and Earth System Sciences* **12**: 7327–7352. DOI: 10.5194/hessd-12-7327-2015

Holbrook WS, Riebe CS, Elwaseif M, Hayes JL, Basler-Reeder K, Harry DL, Malazian A, Dosseto A, Hartsough PC, Hopmans JW. 2013. Geophysical constraints on deep weathering and water storage potential in the Southern Sierra Critical Zone Observatory. *Earth Surface Process and Landforms* **39**: 366–380. DOI: 10.1002/esp.3502.

Huang CY, Asner GP, Martin RE, Barger NN, Neff JC. 2009. Multiscale analysis of tree cover and aboveground carbon stocks in pinyon–juniper woodlands. *Ecological Applications* **19**: 668-681. DOI: 10.1890/07-2103.1.

Hubbart JA, Link TE, Gravelle JA, Elliot WJ. 2007. Timber harvest impacts on water yield in the continental/maritime hydroclimatic region of the United States. *Forest Sciences* **53**: 169–180.

Hull AC, Klomp GJ. 1974. Yield of crested wheatgrass under four densities of big sagebrush in southern Idaho. U.S. Dept. of Agriculture. 44 pp.

Huxman TE, Wilcox BP, Breshears DD, Scott RL, Snyder KA, Small EE, Hultine K, Pockman WT, Jackson RB. 2005. Ecohydrological implications of woody plant encroachment. *Ecology* **86**: 308–319. DOI: 10.1890/03-0583.

Jackson RB, Canadell J, Ehleringer JR, Mooney HA, Sala OE, Schulze ED. 1996. A global analysis of root distributions for terrestrial biomes. *Oecologia* **108**: 389–411. DOI: 10.1007/BF00333714.

Jia QY, Sun FH. 2012. Modeling and forecasting process using the HBV model in Liao river delta. *Procedia of Environmental Science* **13**: 122–128. DOI: 10.1016/j.proenv.2012.01.012.

Konz M, Uhlenbrook S, Braun L, Shrestha A, Demuth S. 2007. Implementation of a process-based catchment model in a poorly gauged, highly glacierized Himalayan headwater. *Hydrology and Earth System Sciences Discussions* **11**: 1323–1339.

Kuhn T, Cao D, George M. 2007. Juniper removal may not increase overall Klamath River Basin water yields. *California Agriculture* **61**: 166–171.

Larsen RE. 1993. Interception and water holding capacity of western juniper. Dissertation. Oregon State University, Corvallis, Oregon, USA.

Lindström G, Johansson B, Persson M, Gardelin M, Bergström, S. 1997. Development and test of the distributed HBV-96 hydrological model. *Journal of Hydrology* **201**: 272-288.

Li H, Beldring S, Xu C-Y, Jain SK. 2014. Modelling runoff and its components in Himalayan basins. *Proceedings 7th Global FRIEND-Water* pp.7-10.

Liu W, Hong Y, Khan SI, Huang M, Vieux B, Caliskan S, Grout T. 2010. Actual evapotranspiration estimation for different land use and land cover in urban regions using Landsat 5 data. *Journal of Applied Remote Sensing* **4**: 41873–41873. DOI: 10.1117/1.3525566.

Lunt ID, Winsemius LM, McDonald SP, Morgan JW, Dehaan RL. 2010. How widespread is woody plant encroachment in temperate Australia? Changes in woody vegetation cover in lowland woodland and coastal ecosystems in Victoria from 1989 to 2005. *Journal of Biogeography* **37**: 722–732. DOI: 10.1111/j.1365-2699.2009.02255.x.

MacDonald LH, Stednick JD. 2003. Forests and water: A state-of-the-art review for Colorado. Colorado Water Resources Research Institute Completion Report **196**.

McCole AA, Stern LA. 2007. Seasonal water use patterns of *Juniperus ashei* on the Edwards Plateau, Texas, based on stable isotopes in water. *Journal of Hydrology* **342**: 238–248. DOI: 10.1016/j.jhydrol.2007.05.024.

McElrone AJ, Pockman WT, Martínez-Vilalta J, Jackson RB. 2004. Variation in xylem structure and function in stems and roots of trees to 20 m depth. *New Phytologist* **163**: 507–517. DOI: 10.1111/j.1469-8137.2004.01127.x.

McGlynn BL, McDonnell JJ, Seibert J, Kendall C. 2004. Scale effects on headwater catchment runoff timing, flow sources, and groundwater-streamflow relations. *Water Recourses Research* **40**. DOI: 10.1029/2003WR002494.

McNamara JP, Chandler D, Seyfried MS, Achet S. 2005. Soil moisture states, lateral flow, and streamflow generation in a semi-arid, snowmelt-driven catchment. *Hydrological Processes* **19**: 4023–4038. DOI: 10.1002/hyp.5869.

Merriman E. 2008. Ranchers continue their campaign against juniper. *Baker City Herald.*

Miller RF, Svejcar TJ, West NE. 1994. Implications of livestock grazing in the Intermountain sagebrush region: plant composition. (eds) M. Vavra, W.A. Laycock, and R.D. Pieper *In* Ecological implications of herbivory in the west. pp. 101–146.

Miller RF, Bates JD, Svejcar TJ, Pierson FB, Eddleman LE. 2005. Biology, ecology, and management of western juniper (*Juniperus occidentalis*). *Corvallis USA Oregon State University Agricultural Experimental Station Technical Bulletin.*

Myrick RM. 1971. Cibecue ridge juniper project.

Niemeyer RJ, Link TE, Seyfried MS, Flerchinger GN. 2016. Surface water input from snowmelt and rain throughfall in western juniper: potential impacts of climate change and shifts in semi-arid vegetation. *Hydrological Process* **30**: 3046–3060. DOI: 10.1002/hyp.10845.

Niemeyer RJ, Heinse R, Link TE, Seyfried MS, Klos PZ, Williams CJ, Nielson T. Spatiotemporal soil and saprolite moisture dynamics across a semi-arid woody plant gradient. *accepted* in *Journal of Hydrology*. DOI: 10.1016/j.jhydrol.2016.11.005

Normand S, Konz M, Merz J. 2010. An application of the HBV model to the Tamor basin in Eastern Nepal. *Journal of Hydrology and Meteorology* **7**: 49–58. DOI: 10.3126/jhm.v7i1.5616.

Oliphant JO. 1968. On the cattle ranges of the Oregon country. University of Washington Press Seattle, WA, USA.

Owens MK, Lyons RK, Alejandro CL. 2006. Rainfall partitioning within semiarid juniper communities: effects of event size and canopy cover. *Hydrological Process* **20**: 3179–3189. DOI:10.1002/hyp.6326.

Pelaez DV, Distel RA, Boo RM, Elia OR, Mayor MD. 1994. Water relations between shrubs and grasses in semi-arid Argentina. *Journal of Arid Environment* **27**: 71–78. DOI: 10.1006/jare.1994.1046.

Pierini NA, Vivoni ER, Robles-Morua A, Scott RL, Nearing MA. 2014. Using observations and a distributed hydrologic model to explore runoff thresholds linked with mesquite encroachment in the Sonoran Desert. *Water Resources Research* **50**. DOI: 10.1002/2014WR015781.

Pierson FB, Bates JD, Svejcar TJ, Hardegree SP. 2007. Runoff and Erosion After Cutting Western Juniper. *Rangeland Ecology and Management* **60**: 285–292. DOI: 10.2111/1551-5028(2007)60[285:RAEACW]2.0.CO;2.

Qiao L, Zou CB, Will RE, Stebler E. 2015. Calibration of SWAT model for woody plant encroachment using paired experimental watershed data. *Journal of Hydrology* **523**: 231–239. DOI: 10.1016/j.jhydrol.2015.01.056.

Richardson CW, Burnett E, Bovey RW. 1979. Hydrologic effects of brush control on Texas rangelands. *Transactions of ASAE*. **22**: 315–319. DOI: 10.13031/2013.35012.

Robinson RW. 1965. The Cibecue Project—a review. *Arizona Watershed Symposium Proceedings* **9**: 24-25.

Romme WH, Allen CD, Bailey JD, Baker WL, Bestelmeyer BT, Brown PM, Eisenhart KS, Floyd ML, Huffman DW, Jacobs BF, Miller RF, Muldavin EH, Swetnam TW, Tausch RJ, Weisberg PJ. 2009. Historical and modern disturbance regimes, stand structures, and landscape dynamics in pinon-juniper vegetation of the western United States. *Rangeland Ecology and Management* **62**: 203–222. DOI: 10.2111/08-188R1.1.

Roundy BA, Vernon JL. 1999. Watershed values and conditions associated with pinyon-juniper communities. *USDA Forest Service Rocky Mountain Research Station, Ogden Utah, USA*. **9**: 172–187.

Roundy BA, Young K, Cline N, Hulet A, Miller RF, Tausch RJ, Chambers JC, Rau B. 2014. Piñon-juniper reduction increases soil water availability of the resource growth pool. *Rangeland Ecology and Management* **67**: 495–505. DOI: 10.2111/REM-D-13-00022.1.

Ryel RJ, Ivans CY, Peek MS, Leffler AJ. 2008. Functional Differences in Soil Water Pools: a New Perspective on Plant Water Use in Water-Limited Ecosystems. *In* Progress in Botany. U. Lüttge, W. Beyschlag, and J. Murata, editors. Springer Berlin Heidelberg, Berlin, Heidelberg. 397–422.

Seibert J. 2005. HBV light version 2, User's manual. Department of Physical Geography and Quaternary Geology, Stockholm University

Seibert J, McDonnell JJ. 2010. Land-cover impacts on streamflow: a change-detection modelling approach that incorporates parameter uncertainty. *Hydrological Sciences Journal* **55**: 316–332. DOI:10.1080/02626661003683264.

Seibert J, McDonnell JJ, Woodsmith RD. 2010. Effects of wildfire on catchment runoff response: a modelling approach to detect changes in snow-dominated forested catchments. *Hydrology Research* **41**: 378-390. DOI: 10.2166/nh.2010.036.

Seyfried MS, Schwinning S, Walvoord MA, Pockman WT, Newman BD, Jackson RB, Phillips FM. 2005. Ecohydrological control of deep drainage in arid and semiarid regions. *Ecology* **86**: 277–287. DOI: 10.1890/03-0568.

Seyfried MS, Wilcox BP. 2006. Soil water storage and rooting depth: key factors controlling recharge on rangelands. *Hydrological Processes* **20**: 3261–3275. DOI: 10.1002/hyp.6331.

Snyder KA, Donovan LA, James JJ, Tiller RL, Richards JH. 2004. Extensive summer water pulses do not necessarily lead to canopy growth of Great Basin and northern Mojave Desert shrubs. *Oecologia* **141**: 325–334. DOI: 10.1007/s00442-003-1403-4.

Sturges DL, Trlica MJ. 1978. Root Weights and Carbohydrate Reserves of Big Sagebrush. *Ecology* **59**: 1282–1285. DOI: 10.2307/1938244.

Tausch RJ, West NE, Nabi AA. 1981. Tree age and dominance patterns in Great Basin pinyon-juniper woodlands. *Journal of Range Management* **34**: 259–264. DOI: 10.2307/3897846.

Thurow TL, Hester JW. 1997. How an increase or reduction in juniper cover alters rangeland hydrology. *In* Juniper Symposium Proceedings. Texas A&M University, San Angelo, Texas, USA. 9–22.

Tromp-van Meerveld HJ, McDonnell JJ. 2006. Threshold relations in subsurface stormflow: 2. The fill and spill hypothesis. *Water Resources Research* **42**. DOI: 10.1029/2004WR003800.

USDA Forest Service (USFS), and US Geological Survey (USGS). 2002. Forest Cover Types. US Geological Survey, Reston, VA.

Walker BH, Noy-Meir I. 1982. Aspects of the stability and resilience of savanna ecosystems. *In* Ecology of tropical savannas. Springer. 556–590.

Wilcox BP. 2002. Shrub control and streamflow on rangelands: A process based viewpoint. *Journal of Range Management* **55**: 318–326. DOI: 10.2307/4003467.

Wilcox BP, Owens MK, Dugas WA, Ueckert DN, Hart CR. 2006. Shrubs, streamflow, and the paradox of scale. *Hydrological Process* **20**: 3245–3259. DOI: 10.1002/hyp.6330.

Williamson TN, Newman BD, Graham RC, Shouse PJ. 2004. Regolith water in zero-order chaparral and perennial grass watersheds four decades after vegetation conversion. *Vadose Zone Journal* **3**: 1007–1016. DOI: 10.2136/vzj2004.1007.

Wine ML, Hendrickx JMH, Cadol D, Zou CB, Ochsner TE. 2015. Deep drainage sensitivity to climate, edaphic factors, and woody encroachment, Oklahoma, USA. *Hydrological Process* **29**: 3779-3789. DOI: 10.1002/hyp.10470.

Wright PN. 1996. Spring enhancement in the Seco Creek water quality demonstration project. Annual Project Report, Seco Creek Water Quality Demonstration Project.

Zou CB, Turton DJ, Will RE, Engle DM, Fuhlendorf SD. 2014. Alteration of hydrological processes and streamflow with juniper (*Juniperus virginiana*) encroachment in a mesic grassland catchment. *Hydrological Processes* **28**: 6173–6182. DOI: 10.1002/hyp.10102.

Table Captions

Table 1: Paired-catchment and one paired-plot studies of pinyon-juniper (PJ) or other woodland and shrub vegetation cover in the western U.S. Annual precipitation based on either annual precipitation given in each study or the 1981 - 2010 PRISM average annual precipitation. Monsoon index based on 1981 - 2010 PRISM derived monthly precipitation.

Table 2: Model parameters used in the HBV model simulations.

Table 3: Table of predicated increase and percent increase in $\Delta ET_{tree\text{-}grass}$ between Level III ecoregions with standard deviations.

Table 4: Table of predicted increase in streamflow with shifts from woodland to grassland cover based on the Hibbert (1983) 450 mm cutoff and prediction of significant gain in evapotranspiration (ET) based on two-tail t-test between woodland to grassland ET across 30 years. In table cells are the percent of total cells for the given condition and 30-year average simulated difference in ET between woodland to grassland cover.

Tables

Table 1

location	citation	woodland vegetation	non-woodland/grassland vegetation	annual precipitation (mm)	monsoon index (mm/mm)	change in streamflow relative to control (mm)
Cibecue Ridge, AZ	Robinson (1965), Myrick (1971)	Pinyon-juniper woodland	herbaceous plants	488	0.37	increase
Beaver Creek, AZ	Clary *et al.* (1974), Baker (1984)	Pinyon pine and Utah Juniper	grasses	463	0.32	9.7
Oklahoma State University Range Research Station, OK	Zou *et al*. (2014)	75% encroached with (*J. virginiana* or eastern red cedar)	grassland under 3-yr burn regime	900	0.27	72
Santa Rita Experimental Range, AZ	Pierini *et al*. (2014)	Mesquite (*Prosopis velutina*) (32%), with some grasses (44%) and bare soil (24%)	grasses (62%), bare soil (19%) and mesquite (19%)	458	0.49	conflicting*
Seco Creek, TX	Wright (1996)	*Juniperus ashei.*	grasses(?)	723	0.28	3.8
Seco Creek, TX	Dugas *et al*. (1998)	*Juniperus ashei.*	bunch grasses	723	0.28	conflicting**
Blackland Prairie, TX	Richardson *et al*. (1979)	honey mesquite (*Prosopis juliflora*)	common broomweed (*Xanthocephalum dracunculoides*) and needlegrass {*Stipa* spp.)	550	0.19	2.4
Camp Creek, OR	Deboodt (2008)	Western Juniper (*Juniperus occidentalis*)	mountain big sagebrush (*Artemisia tridentata*), green and gray rabbitbrush (*Chrysothamnus viscidiflorus* and *C. nauseosus*), and bitterbrush	350	0.13	streamflow amount inconclusive, late season flow increase by 225%, 41 more days of recorded groundwater

			(*Purshia tridentata*), Idaho fescue (*Festuca idahoensis*), bluebunch wheatgrass (*Agropyron spicatum*), Sandberg bluegrass (*Poa secunda*), prairie junegrass (*Koelaria cristata*) and Indian ricegrass (*Oryzopsis hymenoides*).			
Reynolds Creek Experimental Watershed, ID	Seyfried and Wilcox (2006)	mountain big sagebrush (*Artemesia tridentata*) bitterbrush (*Purshia tridentata*), mountain snowberry (*Symphoricarpos oreophilus*) and western juniper (*Juniperus occidentalis*)	bluebunch wheatgrass (*Agropyron spicatum*) and Sandberg bluegrass (*Poa secunda*)	550	0.08	60 mm (deep drainage)
South Mountain, ID	USDA-ARS	Western Juniper (*Juniperus occidentalis*)	low sagebrush (*Artimesia arbuscula Nutt.*) and mountain big sagebrush (*Artimesia* tridentata *Nutt.*)	768	0.06	TBD***
Porter Canyon Experimental Watershed, NV	USDA-ARS	Piñon (*Pinus* spp.) and juniper (*Juniperus* spp.)	sagebrush	351	0.15	TBD***

*’conflicting’ because the 4 years immediately after the mesquite removal, streamflow was larger in control watershed, but in two subsequent 10 year periods (1980-1989, 2000-2012), streamflow was greater in the treatment watershed

**‘conflicting’ because only large events produce runoff, and the two primary large events produced an increase and decrease

*** ‘TBD’ because the treatment to the experimental catchments have only recently been conducted, and analysis of the change in streamflow has not yet occurred

Table 2

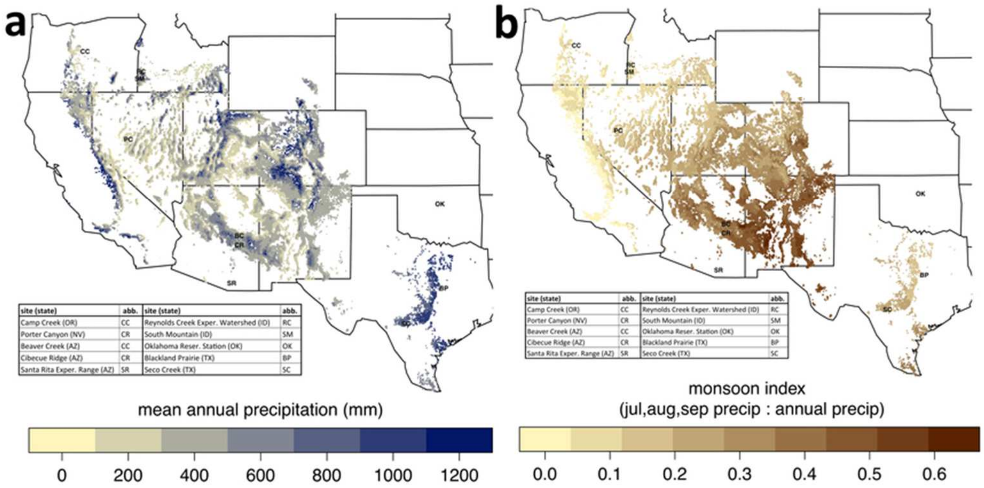
parameters	**unit**	**values used**
snow routine		
degree-day factor	mm °C^{-1}	2
snow threshold temperature	°C	0
snow water holding capacity	-	0.05
refreezing coefficient	-	0.05
soil routine		
SC	mm	50 – 250, 100 (grassland), 200 (woodland)
threshold of reduction of evaporation	-	0.5
shape coefficient		3

Table 3

	% area	average $\Delta ET_{tree\text{-}grass}$ and standard deviation (mm year^{-1})	average percent increase in ET between woodland and grassland cover and standard deviation
Marine – Mixed Forest	1.0%	49.8 +/-20.2	50.4% +/-23.0%
Prairie	1.6%	5.6 +/-1.3	6.8% +/- 3.0%
Mediterranean	0.01%	10.7 +/-0.2	3.6% +/-2.6%
Mediterranean – Mountain	7.0%	34.2 +/-13.8	33.2% +/-21.9%
Subtropical Steppe	17.9%	6.2 +/-7.9	20.2% +/-18.8%
Subtropical Steppe Mountains	14.7%	9.1 +/-10.3	18.6% +/-16.3%
Subtropical Desert	4.7%	3.1 +/-4.8	18.6% +/-19.9%
Temperate Steppe	2.0%	9.3 +/-12.1	27.2% +/28.2%
Temperate Steppe Mountains	19.3%	29.9 +/-18.7	34.9% +/-30.7%
Temperate Dessert	20.0%	29.5 +/-20.5	70.8% +/-36.6%
Temperate Dessert Mountains	11.6%	28.9 +/-17.8	69.4% +/-31.3%

Table 4

	< 450 mm	**> 450 mm**	**sum ($\Delta ET_{tree\text{-}grass}$ p-value)**
$\Delta ET_{tree\text{-}grass}$ p-value > 0.1	38.9% (9.4 mm)	32.2% (18.9 mm)	70.6% (13.7 mm)
$\Delta ET_{tree\text{-}grass}$ p-value < 0.1	15.6% (24.6 mm)	13.9% (52.6 mm)	29.4% (37.8 mm)
sum (450 mm)	54.0% (13.6 mm)	46.0% (29.1 mm)	



site (state)	abb.	site (state)	abb.
Camp Creek (OR)	CC	Reynolds Creek Exper. Watershed (ID)	RC
Porter Canyon (NV)	CR	South Mountain (ID)	SM
Beaver Creek (AZ)	CC	Oklahoma Reser. Station (OK)	OK
Cibecue Ridge (AZ)	CR	Blackland Prairie (TX)	BP
Santa Rita Exper. Range (AZ)	SR	Seco Creek (TX)	SC

Figure 1. Maps of a) average mean annual precipitation and b) monsoon index, across pinyon-juniper cover in the western U.S. based on USGS classification (USGS, 2002). Annual precipitation based on 30 year PRISM average. Monsoon index is the fraction of the annual precipitation that occurs in July, August, and September and is based on monthly PRISM precipitation over 30 years.

36x18mm (600 x 600 DPI)

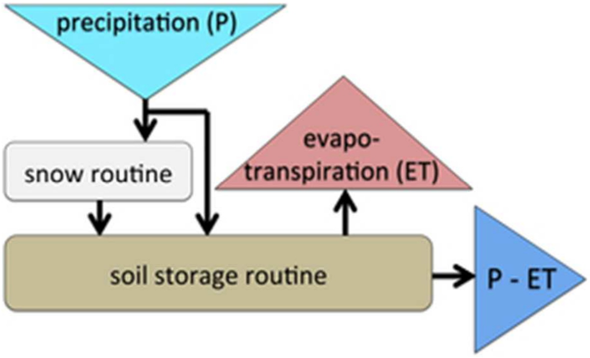


Figure 2. General HBV model structure (modified from Seibert and McDonnell, 2010). Forcing data includes daily precipitation and temperature. The maximum soil storage capacity (SC) of the soil subroutine increases or decreases with rooting depth.

12x7mm (600 x 600 DPI)

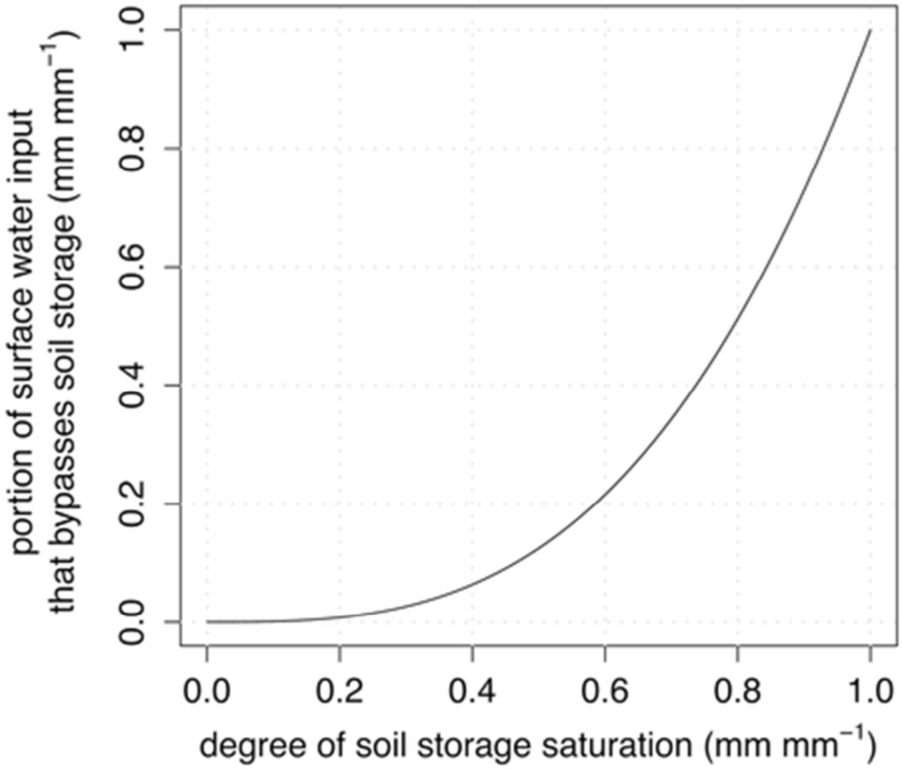


Figure 3. The portion of the surface water input (rain and snowmelt) that bypasses the soil storage based on the degree of soil storage saturation. Relationship based on Seibert (2005).

19x18mm (600 x 600 DPI)

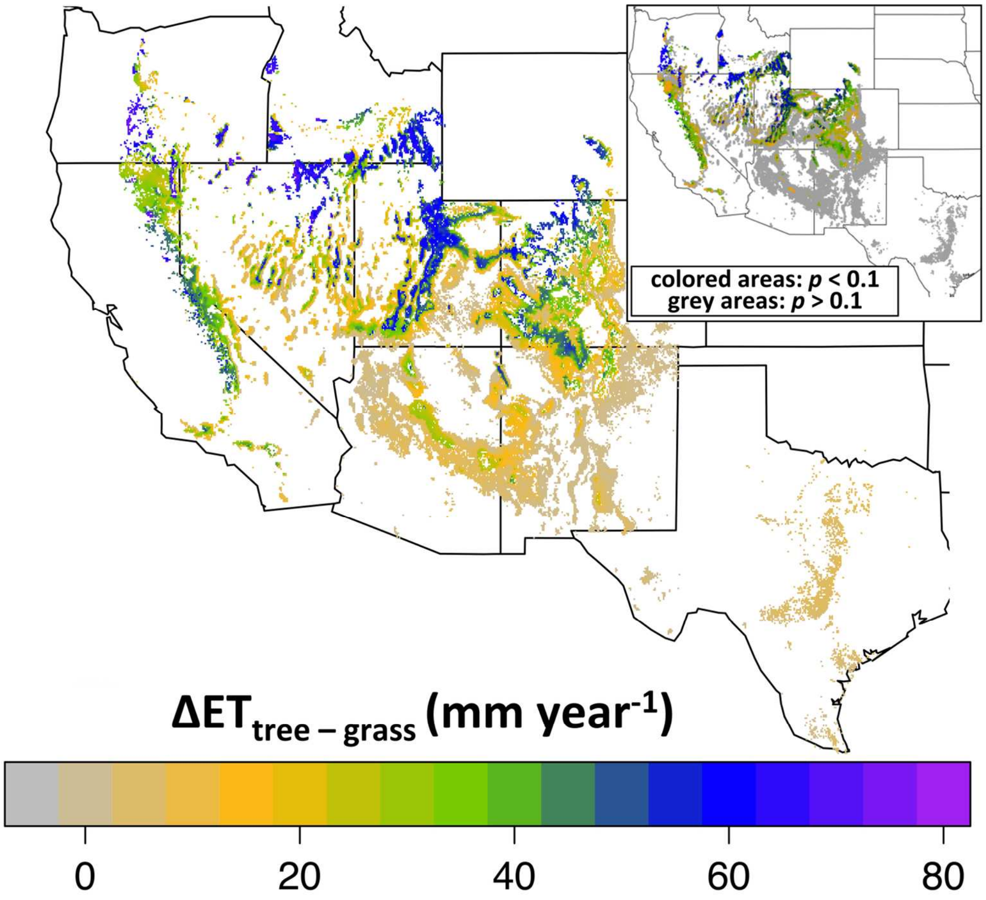


Figure 4. Maps of difference in average annual evapotranspiration (ET) between woodland and grassland ($\Delta ET_{tree-grass}$). Inset figure is the same map except with grid cells with p-values > 0.1 for t-test between 30 years of ET between woodland and grassland cover plotted in grey, and grid cells with p-values < 0.1 plotted in the corresponding $\Delta ET_{tree-grass}$ color.

67x62mm (600 x 600 DPI)

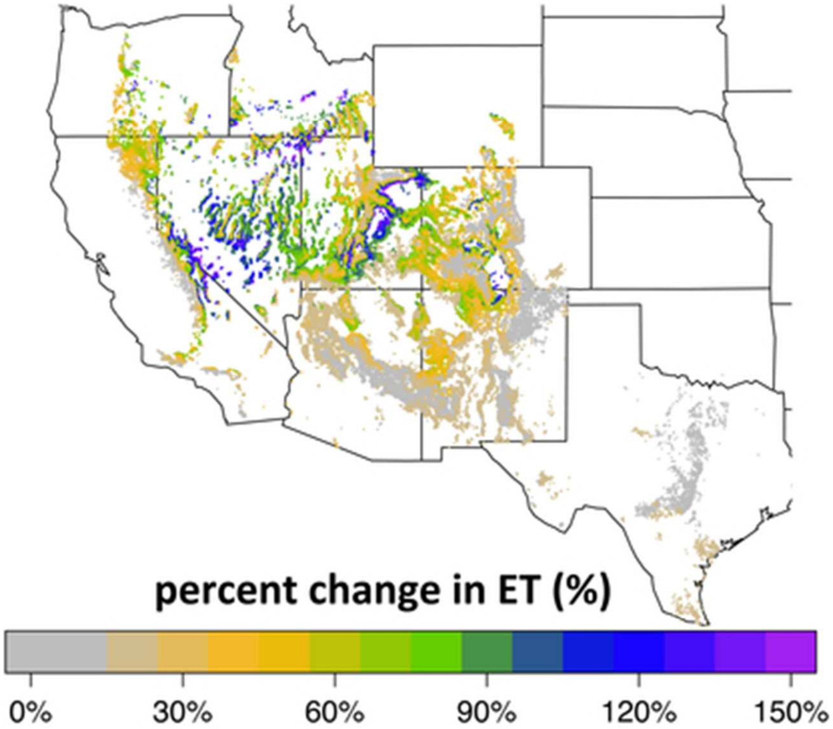


Figure 5. Map of percent difference in ET between woodland and grassland cover.

18x15mm (600 x 600 DPI)

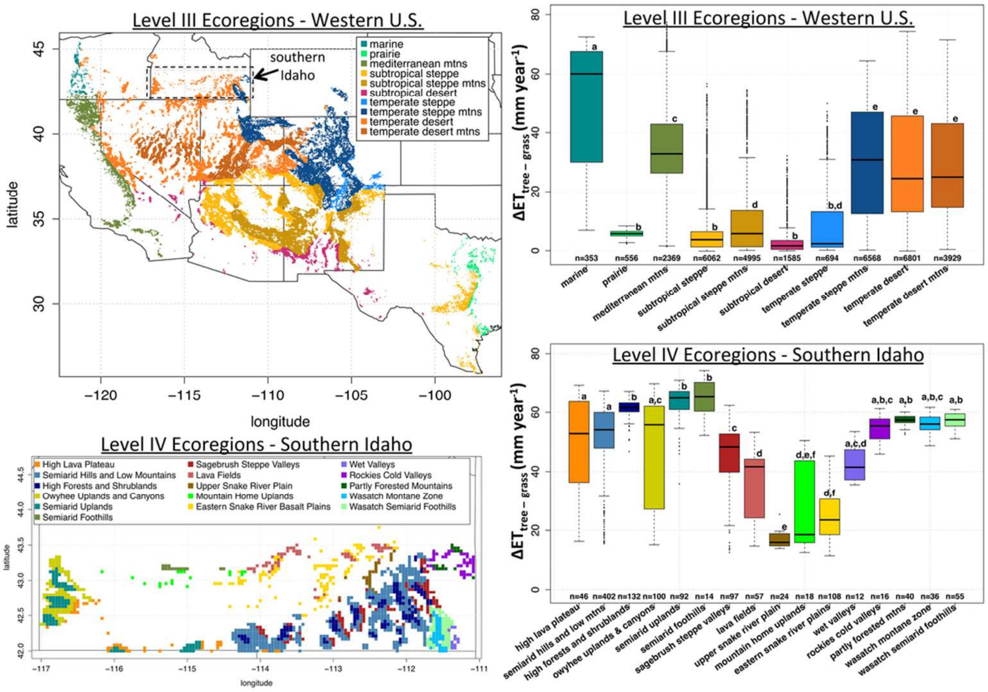


Figure 6. PJ cover within level III ecoregions across the western U.S. and within level IV ecoregions in southern Idaho. Ecoregions divisions based on Bailey (1983). Corresponding boxplots are change in 30-year average ET between woodland and grassland ($\Delta ET_{tree-grass}$) across the locations within each ecoregion. Statistical difference (similarity) in $\Delta ET_{tree-grass}$ between ecoregions indicted by different (the same) letter and determined by t-tests between ecoregions, based on p-value < 0.0001 with the Bonferonni correction method.

50x35mm (600 x 600 DPI)

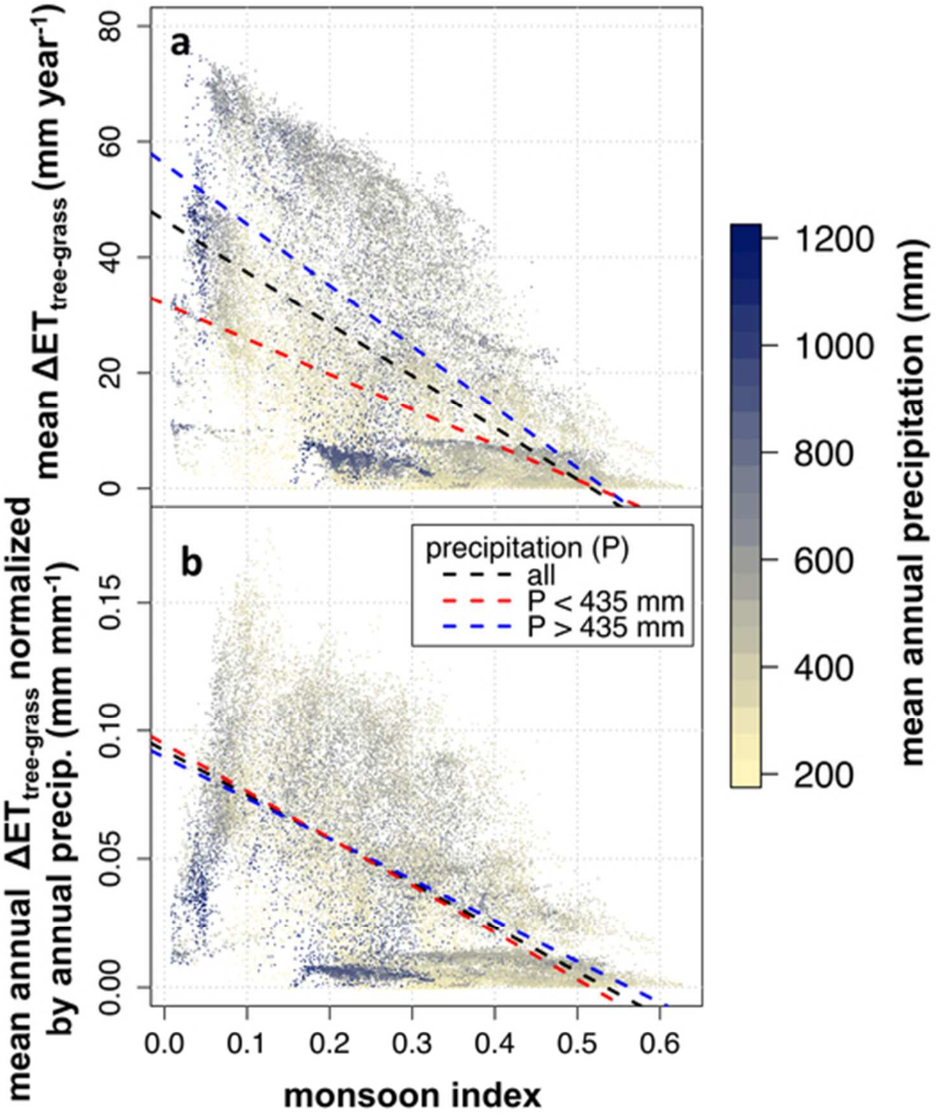


Figure 7. a) Mean annual difference in evapotranspiration (ET) between woodland and grassland cover (ΔET$_{tree-grass}$) and b) mean annual ΔET$_{tree-grass}$ normalized by average annual precipitation vs. monsoon index. Color denotes mean annual precipitation. Regression lines are cells (black line) and separated out by mean annual precipitation greater (blue line) or less than (red line) median precipitation of 435 mm. All regression lines are significant ($p < 0.0001$). For the top plot (a), R^2 values were 0.38, 0.41, 0.37 for all cells, precipitation > 435 mm, and precipitation < 435 mm respectively. For the top plot (b), R^2 values were 0.37, 0.30, 0.44 for all cells, precipitation > 435 mm, and precipitation < 435 mm respectively.

23x27mm (600 x 600 DPI)

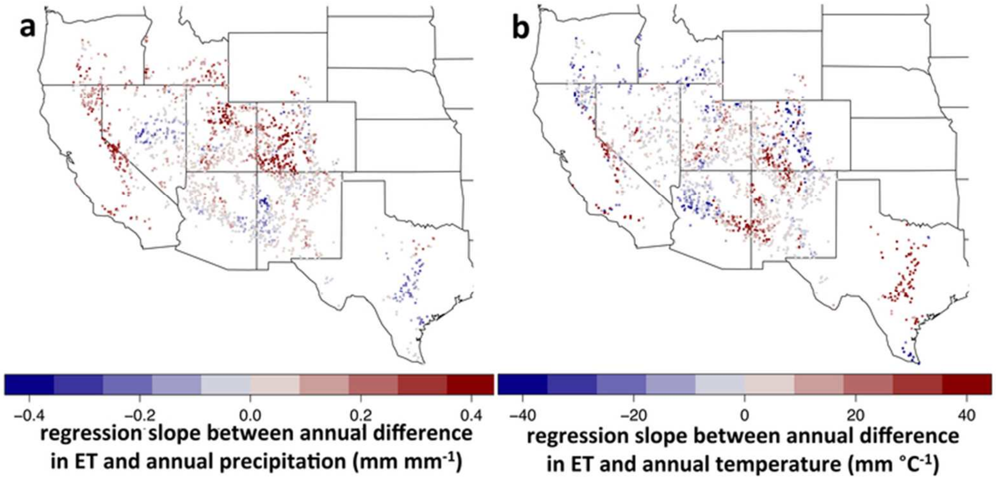


Figure 8. Maps of a) regression slope between annual $\Delta ET_{tree\text{-}grass}$ and annual precipitation within each site and b) regression slope between annual $\Delta ET_{tree\text{-}grass}$ and annual temperature within each site.

35x17mm (600 x 600 DPI)

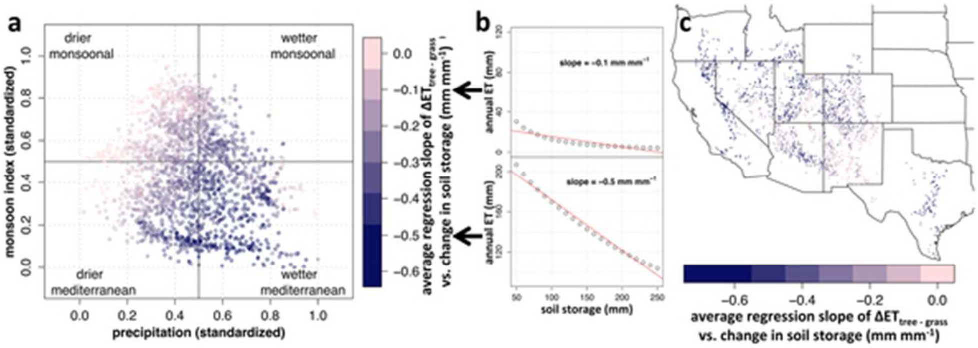


Figure 9. Maps of a) average mean annual precipitation and b) monsoon index, across pinyon-juniper cover in the western U.S. based on USGS classification (USGS, 2002). Annual precipitation based on 30 year PRISM average. Monsoon index is the fraction of the annual precipitation that occurs in July, August, and September and is based on monthly PRISM precipitation over 30 years.

25x9mm (600 x 600 DPI)