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Key Points:
- Streamflow is altered by increasing incidence of extreme precipitation events and forest land management
- We quantify changes in streamflow due to the interaction between forest management and precipitation
- Interaction between forest management and precipitation suggests alteration of streamflow at low and high flows

Supporting Information:
- Supporting Information S1

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Streamflow response to increasing precipitation extremes altered by forest management

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Abstract Increases in extreme precipitation events of floods and droughts are expected to occur worldwide. The increase in extreme events will result in changes in streamflow that are expected to affect water availability for human consumption and aquatic ecosystem function. We present an analysis that may greatly improve current streamflow models by quantifying the impact of the interaction between forest management and precipitation. We use daily long-term data from paired watersheds that have undergone forest harvest or species conversion. We find that interactive effects of climate change, represented by changes in observed precipitation trends, and forest management regime, significantly alter expected streamflow most often during extreme events, ranging from a decrease of 59% to an increase of 40% in streamflow, depending upon management. Our results suggest that vegetation might be managed to compensate for hydrologic responses due to climate change to help mitigate effects of extreme changes in precipitation.

1. Introduction

Forests dominate the headwater landscape and maintain streamflow regimes from headwater watersheds, which is vital to the ecological integrity of aquatic ecosystems, flood control efforts, drought mitigation, and for provisions of drinking water supplies [*Dudley and Stolton*, 2003; *Lowe and Likens*, 2005; *Nadeau and Rains*, 2007; *Caldwell et al*., 2014]. In the U.S., approximately 80% of freshwater supplies originate on forested lands, both public and private [*Sedell et al*., 2000]. Globally, about one third of the world's largest cities obtain a significant proportion of their drinking water from forest areas [*Dudley and Stolton*, 2003]. Changes in water availability and flood and drought severity are expected across the globe, and average annual runoff is currently projected to change anywhere from −30 to +40%, depending on the region [*Kundzewicz et al*., 2009; *Milly et al*., 2005]. Annual runoff is expected to change across 83% of global land area [*Arnell and Gosling*, 2013]. Peak flows are projected to increase and drought flows decrease across more than 50% and 44% of land area, respectively [*Arnell and Gosling*, 2013].

Forest and land management may alter freshwater availability and streamflow through changes in vegetation type, stand age, and associated evapotranspiration (ET) rates [*National Research Council*, 2008; *McGuire and Likens*, 2011; *Jones et al*., 2012]. Furthermore, changes in climate and atmospheric carbon dioxide may affect forest growth and ET, which would also lead to streamflow changes [*Leuzinger and Körner*, 2010; *Ollinger et al*., 2008; *Warren et al*., 2011]. Therefore, an understanding of how forest management or disturbance counteract or exacerbate the climate effects on streamflow (i.e., *interactive effects of management and climate*) is important given the societal dependence of this ecosystem service [*Jones*, 2011]. Understanding and quantifying this interaction effect could contribute to decreasing the level of uncertainty currently present in hydrologic predictive models [*Kundzewicz and Stakhiv*, 2010] and elucidate how management practices can "mimic, exacerbate, counteract, or mask the effects of climate change" on streamflow from headwater watersheds [*Jones et al*., 2012].

An increase in extreme precipitation has been documented at Coweeta Hydrologic Laboratory (North Carolina, USA) with wetter wet years and drier dry years [*Laseter et al*., 2012], and mean annual temperature has been increasing at a rate of 0.5°C per decade since the late 1970s. Most atmospheric-ocean general circulation models (AOGCMs) predict that as climate change continues, the frequency of extreme precipitation events will increase over the next several decades [*O'Gorman and Schneider*, 2009]. This will result in intra-annual changes in ecosystem water budgets, realized as increases in summer drought and winter flood events [*Easterling et al*., 2000; *Knapp et al*., 2008]. Most paired watershed studies have focused on annual streamflow, with less attention on seasonal and climatic variations or climatic extremes [*Brown et al*., 2005; *Burt et al*., 2015].

Forest management activities alter streamflow; forest harvest can provide short-term increases in streamflow, while species conversion can either increase or decrease streamflow for long periods, depending on the structure and water use demands of the new forest [*Bosch and Hewlett*, 1982; *Buttle*, 2011]. However, only a few studies have investigated potential interactions between climate and forest management to alter streamflow [*Ford et al*., 2011b; *Burt et al*., 2015], especially across the entire flow regime (i.e., from extreme high to extreme low flows) and across seasons. In fact, this is one of the first studies to address hydrologic responses across the entire flow regime for multiple paired watershed experiments. Here we analyze long-term data from four paired watersheds at Coweeta in southeastern U.S. under different but common forest management regimes to investigate the extent that forest management and climate change interact to alter streamflow.

2. Methods

We used *daily* streamflow and precipitation data from four paired watershed experiments established at Coweeta, dating back to the 1940s, to identify streamflow responses at daily, seasonal, and annual timescales as functions of climate and forest management. We used a nonlinear time series regression model, which included terms to quantify individual effects of precipitation and vegetation dynamics, as well as the interactive effect of precipitation and vegetation, both prior to forest management and following management. Significance of the interaction term indicates that the vegetation in the managed watershed has responded differently to changing precipitation than the vegetation in the reference watershed to affect streamflow. We used a 14 day weighted sum of precipitation to quantify wetness of climatic conditions. We analyzed flows grouped into classes based on low, medium, and high flow percentiles from two mixed hardwood watersheds (WS7 and WS37), which were clear-cut harvested in 1977 and 1963, respectively, along with two watersheds that were converted from mixed hardwoods to monocultures of eastern white pine (*Pinus strobus*) (WS1 and WS17), initially harvested in 1942 and 1941, respectively. Pines were planted in 1957 and 1956, respectively, as part of a water yield experiment [*Swank and Douglass*, 1974]. Watersheds within vegetation type (hardwood and conifer) differ by elevation (WS7 is low elevation at 722 m at the weir and WS37 is high elevation at 1033 m at the weir) and aspect (WS1 is south facing and WS17 is north facing, and both are approximately the same elevation). These management treatments of clear-cut hardwood harvest followed by regrowth and conifer conversion from hardwoods are representative of approximately 86 million hectares of forested land in the southeastern U.S. or 40% of land cover [*Alig and Butler*, 2004].

Long-term daily stream discharge and precipitation (mm d^{-1}) were collected by the U.S. Department of Agriculture Forest Service at Coweeta (Otto, NC, USA) from four treated watersheds (managed forest) and their associated reference watershed (unmanaged forest). Detailed description of watersheds and management histories and streamflow measurements and rating equations have been described elsewhere [*Swank and Crossley*, 1988; *Ford et al*., 2011b].

We fit the following nonlinear time series regression equation, which includes terms to account for precipitation, vegetation dynamics, and the interaction, using streamflow and precipitation data, both prior to treatment (set $M = 0$) and following treatment (set $M = 1$). The entire time series was used to develop the model of predicted flows in the treated watershed relative to the reference watershed. Flow percentiles were selected from the reference watershed and pooled into low (1, 5, and 10%), medium (40, 50, and 60%), and high (90, 95, and 99%) flows. Data sets used for the regression analyses thus consisted of three flow values from each watershed per year per flow class. From the date of a selected flow from the reference watershed, a precipitation index (P) was calculated as a weighted sum of precipitation from the preceding 14 days. The corresponding flow on the same day from the treated watershed was then selected. We used a 14 day weighted sum as a relative precipitation index to quantify wet and dry climatic conditions, and this was a better predictor of observed streamflow than daily precipitation alone. The predicted treated watershed flow, $\widehat{Q}_{T\%}$, that corresponded to a flow percentile selected from

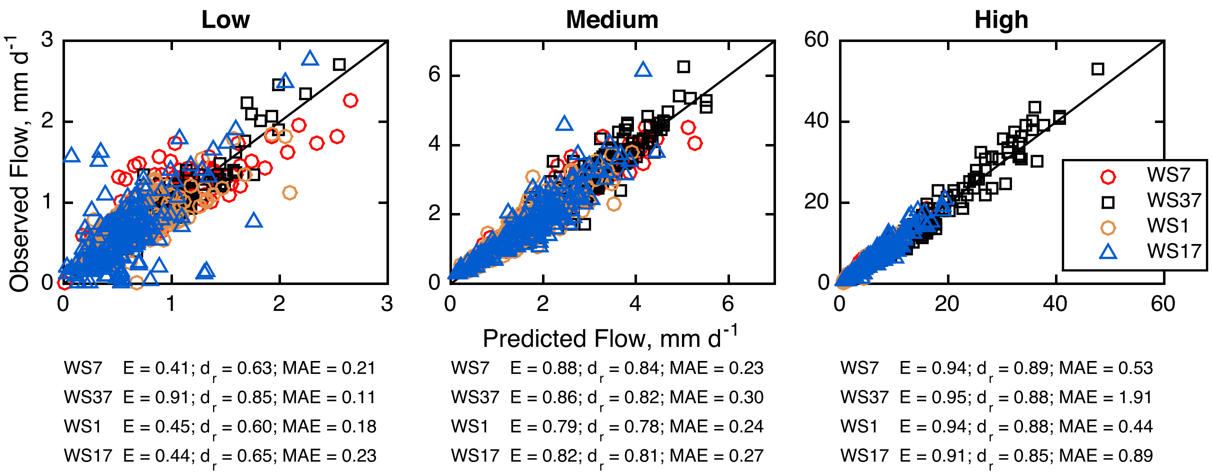

Figure 1. Model efficiency and agreement using observed streamflow from four management treatments at low, medium, and high flows versus the predicted flows from the regression model. E = Nash-Sutcliffe efficiency (values closest to one indicate better predictions), d_r = refined index of agreement (values closest to 1 indicate better predictions), and MAE = mean absolute error (higher values indicate greater model error in mm d^{-1}). Solid line indicates 1:1 line.

the reference watershed on the same day, $Q_{R\%}$, was estimated using a hyperbolic function of vegetation recovery as

$$\widehat{Q}_{T\%} = aQ_{R\%} + bP + \left(M\frac{Y}{1+\tanh(\tau - T)}\right) + c\left(PM\frac{Y}{1+\tanh(\tau - T)}\right), \tag{1}$$

where

$$P = \sum_{i=1}^{14} \frac{P_i}{i}, \tag{2}$$

and where $Q_{R\%}$ is the measured flow from the reference watershed for each percentile. P is the precipitation index, M is the management indicator variable ($M = 0$ for each year prior to treatment, $M = 1$ beginning at treatment and for each year after), Y is the magnitude of change in flow due to management, T is the number of years since management occurred, τ is the time (years) to vegetation recovery, a and b are regression coefficients, and c is the regression coefficient for the interaction.

The model was run by selecting flow percentiles for each year (water year 1 May to 30 April) from the period of record. Water years beginning on 1 May or 1 June have been commonly used for paired watershed studies in eastern U.S. forests [*Patric and Reinhart*, 1971; *Lynch and Corbett*, 1990; *Likens*, 2013]. Seasonal differences were also analyzed with this model by selecting flow percentiles during growing or dormant seasons, depending on tree phenology in each watershed. Regression analyses were performed using nonlinear least squares (lsqnonlin function) and the trust-region-reflective algorithm in MATLAB [*Mathworks*, 2010]. Model parameters were considered significant if 95% confidence intervals of parameter estimates did not include zero. Model efficiency (E), the refined index of agreement (d_r), and mean absolute error (MAE) were calculated as measures of model performance [*Willmott et al.*, 2012, 2015].

We quantified the difference in flow percentiles that could be attributed to precipitation, vegetation, and the interaction using the deviation $D = observed - predicted$, where predicted values were partial models and the terms of interest were set equal to zero. Specifically, we defined the difference in streamflow from the reference watershed that was due to only management (i.e., forest harvest or conversion) as

$$D_0 = Q_{T\%} - \left(\widehat{Q}_{T\%}, M = 0, c = 0\right) \tag{3}$$

and the difference in streamflow from the reference watershed attributed to the interaction of management and precipitation as

$$D_X = Q_{T\%} - \left(\widehat{Q}_{T\%}, c = 0\right) \tag{4}$$

where $Q_{T\%}$ is the observed flow for the treated watershed corresponding to the flow percentile selected from the reference watershed.

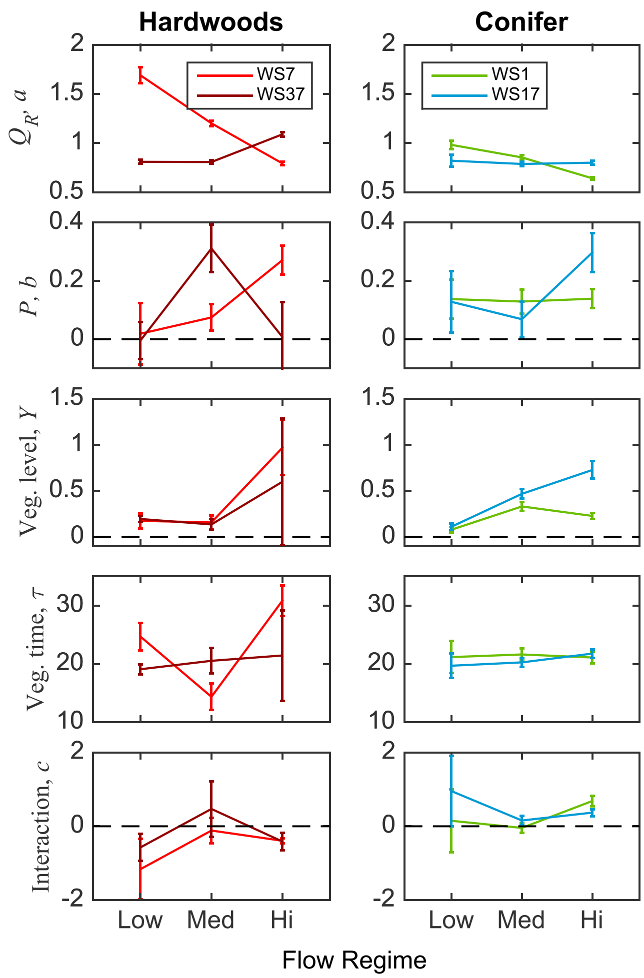

Figure 2. Values of regression model parameters to predict streamflow in equation (1) for four management treatments at annual low, medium, and high flows. The parameter *a* is the regression coefficient for Q_R, which is measured flow from reference watershed; *b* is the regression coefficient for *P*, which is the precipitation index; *c* is the regression coefficient for the interaction term; *Y* is the magnitude of change in flow due to management; and τ is number of years to vegetation recovery. Maroon and red lines represent hardwood clear-cut watersheds, and blue and green lines represent conifer-converted watersheds. Parameters are considered significant contributors to the model if 95% confidence error bars do not cross the zero line (dashed).

3. Results

The regression model predictions compared to the observed streamflow data were generally good, with high flows being predicted at greater model efficiency (*E*) than low flows (Figure 1 and Figure S1 in the supporting information). Across all flows, the majority of models (75%) yielded *E* greater than 0.75 and all models with *E* less than 0.75 were identified in the low flow analyses. Refined index of agreement (d_r) values also were highest within the high flow analyses (from 0.85 to 0.89) and lowest in the low flows (from 0.60 to 0.85), indicating greater model agreement in the high flow analyses. Mean absolute error (MAE) values ranged from 0.11 to 0.23 (mm d^{-1}) in the low flow analyses and from 0.44 to 1.91 (mm d^{-1}) in the high flow analyses, reflecting a greater magnitude of error for high flow models.

3.1. Management and Precipitation Interactions

As expected, streamflow from managed watersheds responded differently to precipitation inputs than if the watershed were unmanaged, resulting in a significant interaction term, *c*. This interaction affected the predicted streamflow from all four watersheds analyzed. However, the influence of this effect varied depending on the watershed and flow percentile class (Figure 2). When examined on a year-to-year basis, the interaction between management and precipitation is more often significant at extreme flows than medium flows (Figure 3), which are more typically examined in streamflow studies [*Huntington*, 2006]. This is especially pronounced at high flows. All four watersheds resulted in significant interaction at high flows, indicating that management could alter the magnitude of high flow, as much as −15%. This was the case for all watersheds after year 2000 except the high-elevation WS37, which was +5% (supporting information Figure S1). The interaction term was also significant at low flows in both hardwood watersheds, but not in either conifer watershed. In contrast, medium flows only had a significant interaction effect in one conifer watershed (north facing WS17). Possible hydrologic consequences of significant interaction terms are discussed below.

We documented changes in precipitation patterns within these watersheds at Coweeta since 1985, indicating that dry periods have become drier. Long-term precipitation alone had no significant effect on most of the streamflows, though the precipitation index associated with low flows has been decreasing in the low-elevation hardwood (WS7, $p = 0.008$) and south facing conifer (WS1, $p = 0.091$) watersheds (i.e., dry periods becoming drier) since 1985. Low flows in the corresponding high elevation and north facing watersheds were

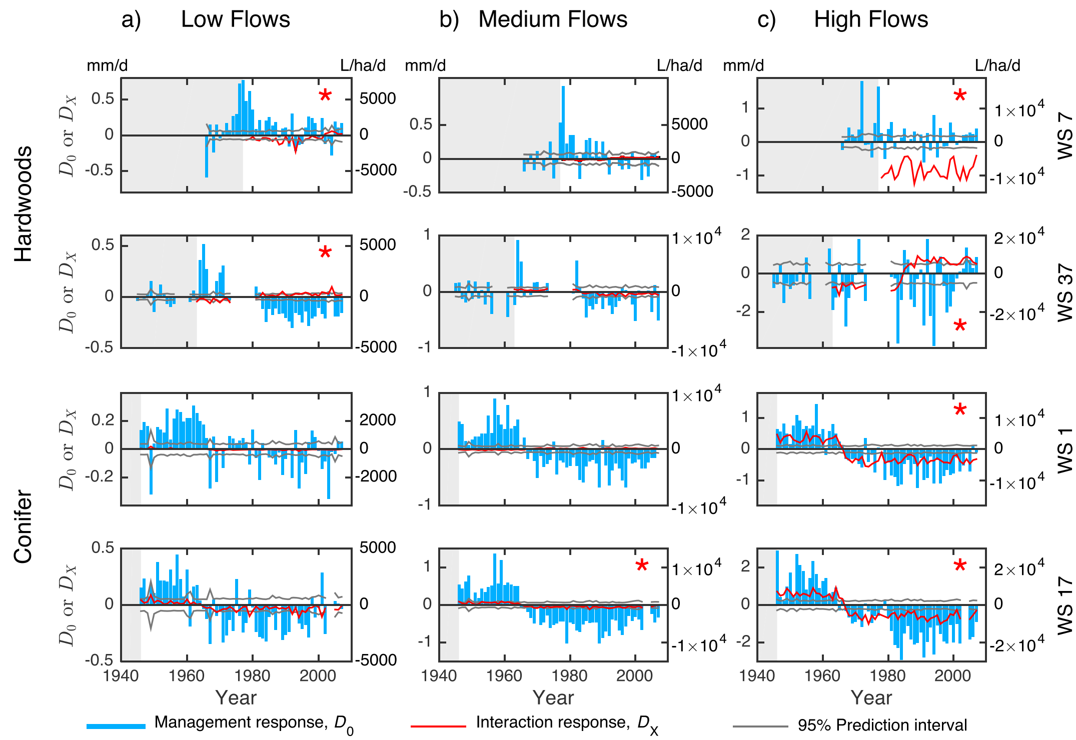

Figure 3. Streamflow deviation response ($\mathrm{mm\,d^{-1}}$ and $\mathrm{L\,ha^{-1}\,d^{-1}}$) to four management treatments at annual low, medium, and high flows. Blue bars indicate response that is attributed to both management and the management by precipitation interaction, D_0 (equation (3)). Red line indicates response that is attributed to the interaction term alone, D_X (equation (4)). Gray area represents years prior to treatment. Gray solid lines indicate the 95% standard error of the prediction. Stars indicate where interaction effect is significant at $\alpha = 0.05$.

not linked to a significant long-term trend in precipitation ($p > 0.10$), nor did we identify any significant trends in the precipitation index associated with medium or high flows.

The highest flows were most affected by the precipitation by vegetation interaction term *c* in our analysis, where the model for all four watersheds had a significant interaction at high flows. The interaction effect was positive for conifers and negative for hardwoods (Figure 2). A significant interaction is likely a function of ET, where at low precipitation, ET differences between conifer and hardwood are slight and become larger as precipitation increases and as stand age increases [*Helvey*, 1967; *Ford et al.*, 2011b]. Differences in ET under dry conditions have been attributed to differential stomatal control by different species, and these patterns have been documented in both temperate and tropical forests [*Shuttleworth et al.*, 1989]. This effect of differential ET at low flows is apparent in our analysis as a significant interaction term *c* under the lowest flow regime in both hardwood watersheds, but not in either conifer watershed (Figure 2). As precipitation increases, this differential response in ET to precipitation can again be seen by the significance of the interaction term in both conifer watersheds at the highest flows, but not the lowest flows (Figures 2 and 3). ET in the conifers under high precipitation is much greater than in the reference hardwood vegetation, mainly due to greater interception losses in the evergreen conifers. Plot level studies suggest that interception from mature white pine is as much as 100% greater than hardwood vegetation [*Helvey*, 1967].

3.2. Forest Management and Recovery

As expected, management significantly altered the streamflow from all watersheds at all flow regimes (Figure 3), where harvest activities caused relative increases in flow for a short time, and this difference decreased over time to equal or become less than pretreatment levels [e.g., *Douglass*, 1983]. The greatest change in magnitude of flow due to management, parameter *Y*, was identified in the low-elevation hardwood watershed (WS7) during high flows ($0.967 \pm 0.29\ \mathrm{mm\,d^{-1}}$) (Figure 2). This produced a difference in mean daily streamflow due to management alone (D_0) of 1.65 mm from WS7 in the first year after harvest at high flows

(Figure 3). The smallest change in daily flow due to management occurred in the south facing conifer watershed (WS1) during low flow with $Y = 0.175 \pm 0.08$ mm, which produced the smallest differences in streamflow due to management alone (D_0) compared to other watersheds at low flows. This reflects that at low precipitation, ET differences between conifer and hardwood are slight and become larger as precipitation increases and as stand age increases [*Ford et al*., 2011b; *Helvey*, 1967].

The time required for streamflow to return to expected flows (i.e., τ) differed by watershed and ranged from 14.4 to 30.9 years (Figure 2). The shortest and longest recovery times were both identified in the low-elevation hardwood clear-cut (WS7) at medium and high flows, respectively. The three other watersheds recovered to pretreatment flows in 18–22 years, regardless of flow regime. In both conifer watersheds, streamflow differences became negative because water use of the new vegetation was greater than that of the vegetation prior to treatment [*Ford et al*., 2011a]. Recovery times in both conifer watersheds were likely prolonged due to the vegetation suppression treatments that occurred until the watersheds were planted with white pine in 1957 (WS1) and 1955 (WS17).

3.3. Hydrologic Consequences of the Interaction

The amount of expected streamflow attributable to the significant interaction term (D_x) in the hardwood watersheds differed depending on elevation (Figure 3). In the low-elevation clear-cut (WS7), the vegetation used more water than expected based on the reference watershed (i.e., a negative D_x at low and high flows), while in the high-elevation clear-cut (WS37), the vegetation used less water than expected (positive D_x at low and high flows). Dominant species composition in the low-elevation clear-cut shifted from scarlet and chestnut oak prior to clear-cut to red maple and yellow poplar [*Boring et al*., 1981, 2014]. Yellow poplar has among the greatest transpiration rates across tree species in the region, such that this species differentially responds to precipitation inputs, across all precipitation regimes, than the species that previously occupied the site [*Ford et al*., 2011a; *Brantley et al*., 2013]. This increase in water use by the new vegetation interacts with precipitation inputs to result in a differential decline in streamflow. Consequences of the negative D_x would serve to decrease flows and exacerbate dry conditions and may also aid in mitigation of high flow events.

The positive D_x of the high-elevation hardwood WS37 at low and high flows suggests an opposite effect compared to low-elevation WS7 responses. This increase in streamflow indicates that the stand structure and/or species composition of the new vegetation has a lower ET than the reference watershed. Dominant species in the high-elevation clear-cut shifted from northern red and chestnut oak to sugar maple and black cherry (W. T. Swank, unpublished tree species survey, 1971). Positive D_x values at high and low flows in WS37, along with the D_x term at medium flows that does not significantly vary from zero, are somewhat contrary to the results documented by *Ford et al*. [2011b]. Their study showed that when *mean annual flows* from this same watershed were analyzed, the new vegetation resulted in higher water use, not lower. Factors leading to different results at high and low flows than mean annual flow may include differences in relative importance of canopy interception versus transpiration at different precipitation levels. It is important to note that the high-elevation WS37 produces high flows that are much higher than the other watersheds (Figure 1) [see also *Swank and Crossley*, 1988] and that during these events, the D_x values represent a much lower interaction effect (less than 5%) compared to the streamflow rates (see Figure 1 (right)).

A positive D_x would be realized as mitigation of dry conditions with water available to maintain flow, or for high flow days, increases in streamflow magnitude would be even greater. For example, under high flow conditions in WS37 (i.e., flows 10–40 mm d^{-1} or 100,000–400,000 L ha^{-1} d^{-1}), the interaction effect accounts for an increase in flow by more than 9200 L ha^{-1} d^{-1} (Figure 3) or about 5% of the total flow (supporting information Figure S1). Direction of change of the D_x in both conifer watersheds was the same; both were negative at high flows, suggesting that conifer conversion would help mitigate high flow events, attributable to higher interception rates in conifers (Figure 3).

The legacy of management activities on streamflow is persistent in time. The influence of the interaction term on the difference in streamflow (D_x) over time does not approach zero but persists throughout the analyzed time series for all watersheds (Figure 3). This suggests that after management activities alter or change the proportion of tree species composition, climate and precipitation will interact with management effects to differentially alter streamflow from managed watersheds relative to reference watershed for as long as differences in the structure and function of the new vegetation persists. For example, in the southern

Appalachians, successional changes in species composition are still occurring from logging during 1880–1920 and other disturbances such as the chestnut blight in 1920s and 1930s [*Elliott and Swank*, 2008].

3.4. Seasonal Variation

It takes longer for dormant season streamflow to recover following clear-cut harvest. On average across the three flow classes, recovery (τ) was 7.0 and 4.1 years longer (WS7 and WS37, respectively) in the dormant season relative to the growing season. The length of time to recovery in both conifer watersheds did not differ by season, as expected.

Management actions converting hardwood forests to conifer plantations can reduce growing season low flows by as much as 20%. When analyzed by season, the interaction effect becomes significant at low flows in the growing season for the north facing conifer watershed WS17, equivalent to a reduction in low flow by as much as 1,100 L ha^{-1} d^{-1} or 20.4%. However, this low flow interaction term was not significant when the entire year was considered. The outcome is a negative difference in expected streamflow due to the interaction effect (D_x), which would exacerbate any water scarcity in the growing season, but not in the dormant season in this watershed. No such seasonal difference was noted in the south facing conifer watershed WS1, likely due to a greater water stress in WS1 as it is south facing and receives slightly less rainfall than WS17. Low overall water availability in conifer-converted watersheds will result in small changes in expected streamflow relative to the previous hardwood vegetation, as the ET differences between the vegetation types at low precipitation are slight.

4. Discussion

We document an important interactive effect between forest management and changes in precipitation patterns that alter predicted streamflow from four different managed watersheds at Coweeta. We show that management activities that alter species composition and structure can have lasting, nonlinear effects on streamflow that depend on elevation and aspect, especially at the lowest and highest flows. Contrary to the paradigm that vegetation has little effect on streamflow at the highest flows, we show that vegetation may indeed influence streamflow during such events, as all four watersheds had a significant interaction at high flows. We highlight the hydrological and practical importance of investigating the entire regime of daily streamflow and at seasonal time scales. Specifically, at low flows, conversion of Appalachian hardwoods to white pine plantations has not exacerbated the effect of low precipitation on streamflow, with an important exception that occurs in the growing season in the north facing watershed.

Our results also suggest that the effect of clear-cut harvest and hardwood regeneration on low flows differs depending on the elevation, where in the high-elevation watershed, the new vegetation has lower water use (positive D_x) and may help mitigate the effects of low precipitation on streamflow (i.e., enhance flows during drought). However, at lower elevation, the new vegetation has higher water use (negative D_x) and may exacerbate drought effects by reducing flow. At high flows, the new vegetation in both conifer watersheds and low-elevation hardwood watershed may help mitigate risk of flood occurrence (negative D_x).

We suggest that there is potential to greatly strengthen paired watershed analyses with inclusion of this interaction of climate and land management. With increased predictive power, we can evaluate options for managing forests to help mitigate streamflow changes that may occur due to an increasing occurrence of extreme precipitation. In these particular watersheds at Coweeta, we show a potential for a streamflow increase of 980 L ha^{-1} d^{-1} at low flows and a decrease of 12,400 L ha^{-1} d^{-1} at high flows attributable solely to the fact that vegetation in managed watersheds responds differently to climate than vegetation in the reference watersheds. Climate and forest management interact and affect streamflow differentially across the flow regime, where increasing or decreasing forest cover, altering dominant species, or converting deciduous to conifer forests can enhance or lessen the effects of changes in precipitation patterns on low and high streamflows.

In practical terms, the state of North Carolina, USA, where these watersheds are located, had a per capita surface water consumption rate of 4621 L d^{-1} (total annual state surface water use was more than 40,136 ML and 2005 population was 8,683,000) [*Kenny et al.*, 2009]. The interaction term in our model has the potential to account for declines in available surface water volume of up to 2300 L ha^{-1} d^{-1} (e.g., in the case of low flows in WS7; equivalent to up to a 20.3% reduction in flow). Understanding the magnitude of the interaction between management and precipitation may greatly improve predictions of surface water availability pertinent to human consumption as a function of both changes in precipitation and forest management.

We contend that this analytical approach is transferrable to other forest types, but the significance and direction of the interaction would be a function of the differences in water use of the new vegetation versus the vegetation being replaced. For example, a significant interaction between forest management and precipitation would be expected where forest management alters the magnitude and sensitivity of ET to climatic variation by impacting factors such as stomatal conductance, albedo, leaf area and density, canopy resistance, and phenology. Examples of these changes include species shifts [e.g., *Nowacki and Abrams*, 2008], species conversions, and widespread losses of foundation tree species [*Ellison et al*., 2005] through infestations or outbreaks such as Emerald Ash borer in Europe and North America [*Poland and McCullough*, 2006; *Baranchikov et al*., 2008] or hemlock woolly adelgid in the eastern U.S. [e.g., *Eschtruth et al*., 2006]. In regions where species do not readily change as a result of management (e.g., Douglas fir (*Pseudotsuga menziesii*), in the Pacific northwest U.S.), a significant interaction between management and precipitation would not be expected to differentially alter streamflow when the density and water use of the stand returned to what it was prior to management, though this process may take many years [*Jassal et al*., 2009; *Best et al*., 2003].

Climate change is a gradual process with hydrologic effects that may be difficult to realize given that changes in vegetation succession or recovery from disturbance or management are convoluted with changes in climate. For example, *Keenan et al*. [2013] document widespread increased water-use efficiency (WUE) of forest vegetation with increasing CO_2, but WUE is likely to have relatively small effects on streamflow compared to large-scale alteration of dominant species composition or changes in precipitation [*Leuzinger and Körner*, 2010]. These factors must be analyzed concomitantly for improved streamflow models. Long-term experimental managed watersheds with hydroclimatic data are therefore invaluable for examining the hydrologic effects of climate change [*Jones et al*., 2012]. Forested headwater basins are an important source of water for human use and ecosystem function, and thus, maintaining streamflow from these headwaters in light of expected climate change and various land management is essential for sustainably managing water availability [*Furniss et al*., 2010; *Caldwell et al*., 2014].

5. Conclusions

We find that the interactive effects of climate change, as represented by changes in observed precipitation trends and forest management regime, alter expected streamflow ranging from a decrease of 59% to an increase of 40% in streamflow (supporting information Figure S1). This interaction effect results in a significant decrease or increase in expected streamflow. In our analysis, the effect ranged from a decrease of low flows by nearly 2300 $\mathrm{L\,ha^{-1}\,d^{-1}}$ to an increase of high flows by nearly 9300 $\mathrm{L\,ha^{-1}\,d^{-1}}$ (Figure 3), depending on vegetation type or management practice. We emphasize that representations of these interactions are missing from current streamflow predictive models used to inform climate change impact analysis. Our results suggest that forests might be managed through activities such as vegetation conversion to compensate for some of the current and anticipated hydrologic responses due to climate change and help mitigate the effects of extreme changes in precipitation.

Acknowledgments
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References

Alig, R. J., and B. J. Butler (2004), Area changes for forest cover types in the United States, 1952 to 1997, with projections to 2050, *Gen. Tech. Rep. PNW-GTR-613*, 106 pp., U.S. Dep. of Agric. For. Serv., Pac. Northwest Res. Stn., Portland, Ore.

Arnell, N. W., and S. N. Gosling (2013), The impacts of climate change on river flow regimes at the global scale, *J. Hydrol.*, *486*, 351–364.

Baranchikov, Y., E. Mozolevskaya, G. Yurchenko, and M. Kenis (2008), Occurrence of the emerald ash borer, Agrilus planipennis in Russia and its potential impact on European forestry, *EPPO Bull.*, *38*(2), 233–238.

Best, A., L. Zhang, T. McMahon, A. Western, and R. Vertessy (2003), A critical review of paired catchment studies with reference to seasonal flows and climatic variability, *CSIRO Land and Water Tech. Rep. 25/03*, Murray-Darling Basin Comm., Canberra, Australia.

Boring, L. R., C. D. Monk, and W. T. Swank (1981), Early regeneration of a clear-cut southern Appalachian forest, *Ecology*, *62*, 1244–1253.

Boring, L. R., K. J. Elliott, and W. T. Swank (2014), Successional forest dynamics: 30 years following clearcutting, in *Long-Term Response of a Forest Watershed Ecosystem: Clearcutting in the Southern Appalachians*, edited by W. T. Swank and J. R. Webster, pp. 11–35, Oxford Univ. Press, New York.

Bosch, J. M., and J. D. Hewlett (1982), A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration, *J. Hydrol.*, *55*(1/4), 3–23.

Brantley, S., C. R. Ford, and J. M. Vose (2013), Future species composition will affect forest water use after loss of eastern hemlock from southern Appalachian forests, *Ecol. Appl.*, *23*(4), 777–790.

Brown, A. E., L. Zhang, T. A. McMahon, A. W. Western, and R. A. Vertessy (2005), A review of paired catchment studies for determining changes in water yield resulting from alterations in vegetation, *J. Hydrol.*, *310*(1-4), 28.

Burt, T. P., N. J. K. Howden, J. J. McDonnell, J. A. Jones, and G. R. Hancock (2015), Seeing the climate through the trees: Observing climate and forestry impacts on streamflow using a 60-year record, *Hydrol. Processes*, *29*(3), 473–480.

Buttle, J. M. (2011), The effects of forest harvesting on forest hydrology and biogeochemistry, in *Forest Hydrology and Biogeochemistry: Synthesis of Past Research and Future Directions*, *Ecol. Stud. Ser.*, vol. 216, edited by D. F. Levia, D. E. Carlyle-Moses, and T. Tanaka, pp. 659–677, Springer, Heidelberg, Germany.

Caldwell, P., C. Muldoon, C. Ford-Miniat, E. Cohen, S. Krieger, G. Sun, S. McNulty, and P. V. Bolstad (2014), Quantifying the role of National Forest system lands in providing surface drinking water supply for the Southern United States, *Gen. Tech. Rep. SRS-197*, 135 pp., U.S. Dep. of Agric. For. Serv., Southern Res. Stn., Asheville, N. C.

Douglass, J. E. (1983), Potential for water yield augmentation from forest management in the eastern United States, *Water Resour. Bull.*, *19*(3), 351–358.

Dudley, N., and S. Stolton (Eds.) (2003), *Running Pure: The Importance of Forest Protected Areas to Drinking Water*, pp. 112, World Bank/WWF Alliance for Forest Conservation and Sustainable Use, Gland, Switzerland and Washington, D. C.

Easterling, D. R., G. A. Meehl, C. Parmesan, S. A. Changnon, T. R. Karl, and L. O. Mearns (2000), Climate extremes: Observations, modeling, and impacts, *Science*, *289*(5487), 2068–2074.

Elliott, K. J., and W. T. Swank (2008), Long-term changes in forest composition and diversity following early logging (1919–1923) and the decline of American chestnut (Castanea dentata), *Plant Ecol.*, *197*(2), 155–172.

Ellison, A. M., et al. (2005), Loss of foundation species: Consequences for the structure and dynamics of forested ecosystems, *Front. Ecol. Environ.*, *3*(9), 479–486.

Eschtruth, A. K., N. L. Cleavitt, J. J. Battles, R. A. Evans, and T. J. Fahey (2006), Vegetation dynamics in declining eastern hemlock stands: 9 years of forest response to hemlock woolly adelgid infestation, *Can. J. Forest. Res.*, *36*(6), 1435–1450.

Ford, C. R., R. M. Hubbard, and J. M. Vose (2011a), Quantifying structural and physiological controls on variation in canopy transpiration among planted pine and hardwood species in the southern Appalachians, *Ecohydrology*, *4*(2), 183–195.

Ford, C. R., S. H. Laseter, W. T. Swank, and J. M. Vose (2011b), Can forest management be used to sustain water-based ecosystem services in the face of climate change? *Ecol. Appl.*, *21*(6), 2049–2067.

Furniss, M. J., et al. (2010), Water, Climate Change, and Forests: Watershed Stewardship for a Changing Climate, *Gen. Tech. Rep. PNW-GTR-812*, 75 pp., U.S. Dep. of Agric. For. Serv., Pac. Northwest Res. Stn., Portland, Ore.

Helvey, J. D. (1967), Interception by eastern white pine, *Water Resour. Res.*, *3*(3), 723–729, doi:10.1029/WR003i003p00723.

Huntington, T. G. (2006), Evidence for intensification of the global water cycle: Review and synthesis, *J. Hydrol.*, *319*(1-4), 83–95.

Jassal, R. S., T. A. Black, D. L. Spittlehouse, C. Brümmer, and Z. Nesic (2009), Evapotranspiration and water use efficiency in different-aged Pacific Northwest Douglas-fir stands, *Agric. Forest Meteorol.*, *149*(6-7), 1168–1178.

Jones, J. A. (2011), Hydrologic responses to climate change: Considering geographic context and alternative hypotheses, *Hydrol. Processes*, *25*(12), 1996–2000.

Jones, J. A., et al. (2012), Ecosystem processes and human influences regulate streamflow response to climate change at long-term ecological research sites, *BioScience*, *62*(4), 390–404.

Keenan, T. F., D. Y. Hollinger, G. Bohrer, D. Dragoni, J. W. Munger, H. P. Schmid, and A. D. Richardson (2013), Increase in forest water-use efficiency as atmospheric carbon dioxide concentrations rise, *Nature*, *499*(7458), 324–327.

Kenny, J. F., N. L. Barber, S. S. Hutson, K. S. Linsey, J. K. Lovelace, and M. A. Maupin (2009), *Estimated Use of Water in the United States in 2005, Circular 1344*, pp. 52, U.S. Geol. Surv., Reston, Va.

Knapp, A. K., et al. (2008), Consequences of more extreme precipitation regimes for terrestrial ecosystems, *BioScience*, *58*(9), 811–821.

Kundzewicz, Z. W., and E. Z. Stakhiv (2010), Are climate models “ready for prime time” in water resources management applications, or is more research needed? *Hydrol. Sci. J.*, *55*(7), 1085–1089.

Kundzewicz, Z. W., L. J. Mata, N. W. Arnell, P. Döll, B. Jimenez, K. Miller, T. Oki, and Z. Şen (2009), Water and climate projections, *Hydrol. Sci. J.*, *54*(2), 406–415.

Laseter, S. H., C. R. Ford, J. M. Vose, and L. W. S. Jr (2012), Long-term temperature and precipitation trends at the Coweeta Hydrologic Laboratory, Otto, North Carolina, USA, *Hydrol. Res.*, doi:10.2166/nh.2012.2067.

Leuzinger, S., and C. Körner (2010), Rainfall distribution is the main driver of runoff under future CO2-concentration in a temperate deciduous forest, *Global Change Biol.*, *16*(1), 246–254.

Likens, G. E. (2013), *Biogeochemistry of a Forested Ecosystem*, 3rd ed., 208 pp., Springer, New York.

Lowe, W. H., and G. E. Likens (2005), Moving headwater streams to the head of the class, *BioScience*, *55*(3), 196–197.

Lynch, J. A., and E. S. Corbett (1990), Evaluation of best management-practices for controlling nonpoint pollution from silvicultural operations, *Water Resour. Bull.*, *26*(1), 41–52.

Mathworks (2010), *Image Processing Toolbox—User’s Guide*, Mathworks, Natick, Mass.

McGuire, K. J., and G. E. Likens (2011), Historical roots of forest hydrology and biogeochemistry, in *Forest Hydrology and Biogeochemistry: Synthesis of Past Research and Future Directions*, *Ecol. Stud. Ser.*, vol. 216, edited by D. F. Levia, D. E. Carlyle-Moses, and T. Tanaka, pp. 3–26, Springer, Heidelberg, Germany.

Milly, P. C. D., K. A. Dunne, and A. V. Vecchia (2005), Global pattern of trends in streamflow and water availability in a changing climate, *Nature*, *438*(7066), 347–350.

Nadeau, T.-L., and M. C. Rains (2007), Hydrological connectivity between headwater streams and downstream waters: How science can inform policy, *J. Am. Water Resour. Assoc.*, *43*(1), 118–133.

Nowacki, G. J., and M. D. Abrams (2008), The demise of fire and “mesophication” of forests in the eastern United States, *BioScience*, *58*(2), 123–138.

National Research Council (2008), *Hydrologic Effects of a Changing Forest Landscape*, 168 pp., Natl. Acad. Press, Washington D. C.

O’Gorman, P. A., and T. Schneider (2009), The physical basis for increases in precipitation extremes in simulations of 21st-century climate change, *Proc. Natl. Acad. Sci. U.S.A.*, *106*(35), 14,773–14,777.

Ollinger, S. V., C. L. Goodale, K. Hayhoe, and J. P. Jenkins (2008), Potential effects of climate change and rising CO2 on ecosystem processes in northeastern U.S. forests, *Mitig. Adapt. Strategies Global Change*, *13*(5-6), 467–485.

Patric, J. H., and K. G. Reinhart (1971), Hydrologic effects of deforesting 2 mountain watersheds in West Virginia, *Water Resour. Res..*, *7*(5), 1182–1188, doi:10.1029/WR007i005p01182.

Poland, T. M., and D. G. McCullough (2006), Emerald ash borer: Invasion of the urban forest and the threat to North America’s ash resource, *J. Forest*, *104*(3), 118–124.

Sedell, J., M. Sharpe, D. D. Apple, M. Copenhagen, and M. Furniss (2000), *Water and the Forest*, *Service Rep. USDA Forest Service FS-660*, 26 pp., U.S. Dep. of Agric. For. Serv., Washington, D. C.

Shuttleworth, W. J., R. Leuning, T. A. Black, J. Grace, P. G. Jarvis, J. Roberts, and H. G. Jones (1989), Micrometeorology of temperate and tropical forest [and discussion], *Philos. Trans. R. Soc. London Ser. B*, *324*(1223), 299–334.

Swank, W. T., and D. A. Crossley (1988), *Forest Hydrology and Ecology at Coweeta*, vol. 66, pp. 469, Springer, New York.

Swank, W. T., and J. E. Douglass (1974), Streamflow greatly reduced by converting deciduous hardwood stands to pine, *Science*, *185*(4154), 857–859.

Warren, J. M., E. Pötzelsberger, S. D. Wullschleger, P. E. Thornton, H. Hasenauer, and R. J. Norby (2011), Ecohydrologic impact of reduced stomatal conductance in forests exposed to elevated CO2, *Ecohydrology*, *4*(2), 196–210.

Willmott, C. J., S. M. Robeson, and K. Matsuura (2012), A refined index of model performance, *Int. J. Climatol.*, *32*(13), 2088–2094.

Willmott, C. J., S. M. Robeson, K. Matsuura, and D. L. Ficklin (2015), Assessment of three dimensionless measures of model performance, *Environ. Modell. Software*, *73*, 167–174.