

## Can forest watershed management mitigate climate change effects on water resources?

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**Abstract** Long-term hydrology and climate data from United States Forest Service Experimental Forests and Ranges (EFR) provide critical information on the interactions among climate, streamflow, and forest management practices. We examined the relationships among streamflow responses to climate variation and forest management using long-term data. Analysis of climate data from a subset of EFRs suggested an increase in air temperature over the past 20–30 years. Streamflow increased initially after cutting for all sites and cutting treatments, but the longer term responses vary considerably across sites and treatments. Streamflow response following cutting without vegetation conversion depends on variation in treatment, soils, vegetation, and climatic regimes among sites. Statistical models indicate that many of the management treatments result in forest stand structure and species composition interact with climate differently than the unmanaged reference stand, indicating that forest management may exacerbate or mitigate the effects of future climatic conditions.

**Key words** United States Forest Service paired watersheds; climate change; streamflow; forest management; interactions

### INTRODUCTION

For more than a century, research attention has been focused on resolving the relationship between forests and water resources (Andreassian, 2004). Research from paired experimental watersheds in the United States clearly demonstrates that forested watersheds are especially important (Brown *et al.*, 2005) because both managed and unmanaged forests provide the cleanest and most stable water supplies for drinking water, aquatic habitat, and groundwater recharge compared to all other land uses. Forests are unique among land uses because they are long-lived, relatively stable and respond to climate through the process of evapotranspiration; yet, their structure and function can be substantially altered by management and/or natural disturbances (Vose *et al.*, 2011). These structural and functional changes can be either positive or negative, and either transient or long-term, depending on the magnitude and intensity of the management action or disturbance. Understanding how climate change effects the supply of water from forested watersheds, and developing management strategies to mitigate or offset those impacts, is critical to maintaining water supplies for human use and aquatic species and habitats.

Climate change is likely to both directly and indirectly affect water resources in the United States (Brian *et al.*, 2004; Sun *et al.*, 2008). The direct effects on water resources will depend on

how climate change alters the amount, type (i.e. snow vs rain), and timing of precipitation and how this influences baseflow, stormflow, groundwater recharge, and flooding. Long-term streamflow data suggest that average annual streamflow has increased and this increase has been linked to greater precipitation in the eastern continental US over the past 100 years (Karl & Knight, 1998; Lins & Slack, 1999; IPCC, 2007) and to climate warming at global scales (Labat *et al.*, 2004). In addition, many regions of the US have experienced an increased frequency of precipitation extremes over the last 50 years (Easterling *et al.*, 2000a; Huntington, 2006; IPCC, 2007). Most Global Change Models (GCMs) predict that as climate warms, the frequency of extreme precipitation events increases across the globe (O’Gorman & Schneider, 2009); however, the timing and spatial distribution of extreme events are among the most uncertain aspects of future climate scenarios (Karl *et al.*, 1995; Allen & Ingram, 2002). Despite this uncertainty, recent experience with droughts and low flows in many areas of the US indicate that even small changes in drought severity and frequency will have a major impact on society, including drinking water supplies (Easterling *et al.*, 2000b; Luce & Holden, 2009).

Indirect impacts are related to changes in temperature and increasing atmospheric CO<sub>2</sub> concentration. In the short term, increased temperature has the potential to increase plant water use via transpiration and evaporation (as temperatures increase, the energy available for evapotranspiration increases), and hence decrease residual precipitation available for streamflow or groundwater recharge. Increasing CO<sub>2</sub> concentration typically decreases stomatal aperture in plants (Ehleringer & Cerling, 1995; Beerling, 1996; Prentice & Harrison, 2009). As leaf stomatal conductance declines, transpiration can decline as well; but this is often dependent on several factors, including stand age and structure (Leuzinger & Korner, 2007; Warren *et al.*, 2011). Warmer temperatures will also influence the duration and timing of snowmelt, a critical factor in ecosystems where snowmelt influences hydrologic processes. The impacts of temperature may be offset (or exacerbated) by changes in other factors that influence evapotranspiration such as vapour pressure (warm air holds more water), wind patterns (which impact boundary layer resistance), increases in CO<sub>2</sub> (which decrease stomatal conductance), and changes in net radiation (influenced by changes in cloud cover and aerosols). In the longer term, climate warming (in combination with changes in precipitation) will likely produce shifts in tree species distributions (Iverson *et al.*, 2008). Many tree species differ considerably in the amount annual and seasonal water use via transpiration and interception loss (Ford *et al.*, 2011a). For example, in some geographic regions, a shift from deciduous hardwood to evergreen conifer forests may result in greater water use by evergreen conifer forests due to year-round transpiration and interception loss (Stoy *et al.*, 2006; Ford *et al.*, 2011a). Controlled studies have demonstrated that increased atmospheric CO<sub>2</sub> reduces transpiration in many tree species which may translate into increased streamflow (Ainsworth & Rogers, 2007; Betts *et al.*, 2007; Warren *et al.*, 2011); however, it is less well known if these patterns will persist over the long-term. Taken together, the impacts of climate change on water resources are complex and variable over space and time.

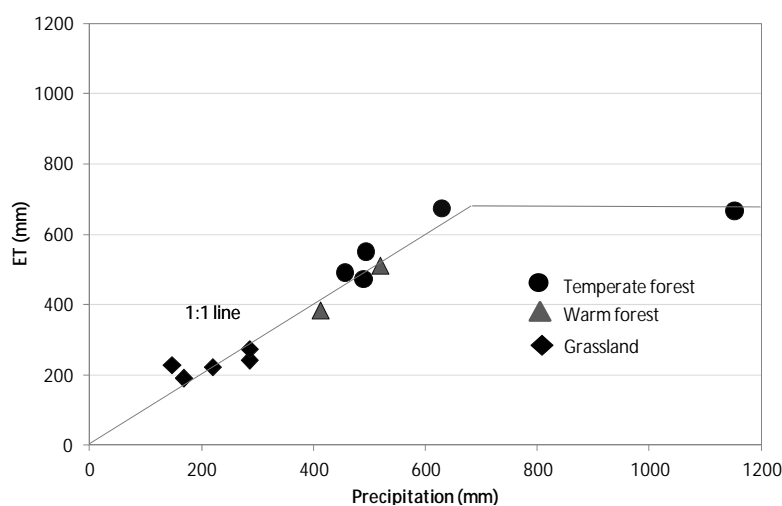
Despite this complexity, our current understanding of the relationships among forests, climate and climate variability, and streamflow provides significant guidance on the potential responses of forest ecosystems to climate change. Much of this understanding comes from long-term forest watershed experiments at USDA Forest Service Experimental Forests and Ranges (EFRs) (Table 1) (Moran *et al.*, 2008). Some watershed EFRs were established as early as the 1930s and most continue with the primary purpose of understanding relationships among forests, forest management and other disturbances, and water resources. Although not originally established to examine the impacts of climate change, the long-term hydrology and climate data at many of the EFRs can be used to examine how forested watersheds have responded to climate variability in the past. Because the retrospective approach is built on decades of observations, it is a powerful approach for projecting future responses to climate change (Lindenmayer *et al.*, 2010). In addition, many of these EFRs have management treatments that include forest harvesting, species conversions, land-use change, and natural disturbances such as insect outbreaks and hurricanes. These managed and unmanaged disturbances provide an excellent opportunity to examine the interaction between climate change and variability and managed and unmanaged disturbances.

Understanding these interactions will provide guidance to land managers as they consider management practices that can be used to mitigate or offset the impacts of future climate change and variability on watershed resources.

**Table 1** Location of watersheds in USDA Forest Service Experimental Forests and Ranges (EFRs) used for analyses<sup>1</sup>.

| Experimental forest | Location      | Major vegetation                                | Streamflow record | Annual precipitation (mm) |
|---------------------|---------------|---|-------------------|---------------------------|
| Caspar Creek        | Northern CA   | Coastal redwood, Douglas fir, W. hemlock        | 1962              | 1200                      |
| Coweeta             | Western NC    | Oak-hickory, cove, northern hardwoods           | 1934              | 1800–2300                 |
| Fernow              | West Virginia | Mixed mesophytic hardwood                       | 1951              | 1473                      |
| Fraser              | Colorado      | Subalpine forest, alpine tundra                 | 1941              | 432–711                   |
| H. J. Andrews       | Western OR    | Douglas fir, western hemlock                    | 1949              | 2400                      |
| Hubbard Brook       | New Hampshire | Northern hardwoods, spruce-fir                  | 1945 (USGS)       | 1400                      |
| Luquillo            | Puerto Rico   | Evergreen broadleaf tropical forest             | 1988 (USFS)       | 2000–5000                 |
| Marcell             | Minnesota     | Forested peatlands, upland hardwoods            | 1961              | 780                       |
| San Dimas           | Southern CA   | Mixed chaparral                                 | 1939              | 486                       |
| Santee              | Coastal SC    | Loblolly and longleaf pine, mixed pine-hardwood | 1967              | 1350                      |

<sup>1</sup>Information developed from a combination of EFR websites and Adams *et al.* (2008).



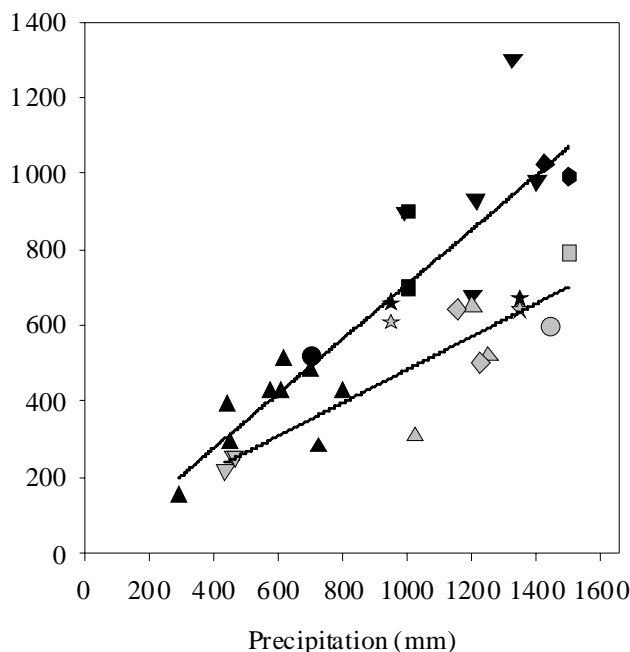
**Fig. 1** Growing season precipitation vs *ET* for a wide range of ecosystems (Sun *et al.*, 2011).

### Climate and streamflow in unmanaged forest watersheds

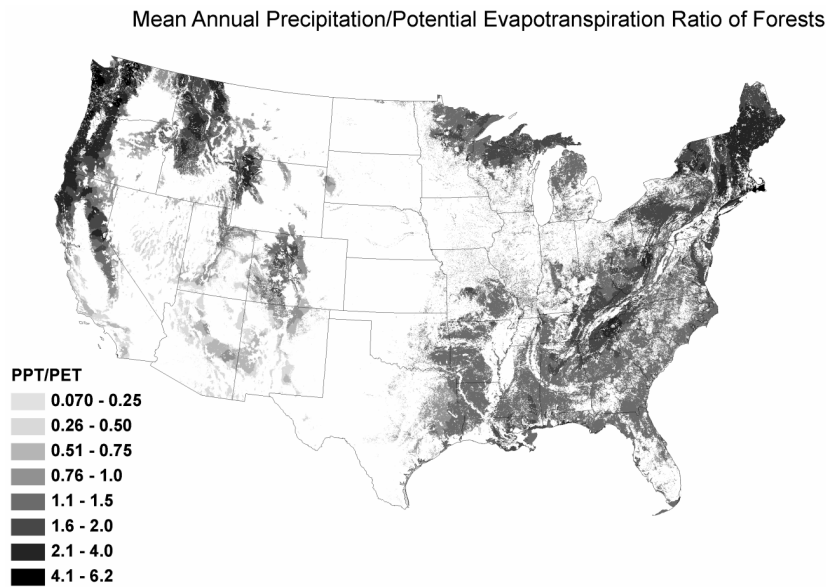
Across forest biomes, growing season evapotranspiration is highly correlated with annual precipitation (Fig. 1); and for a given amount of precipitation, coniferous forests generally have greater annual evapotranspiration than deciduous forests (Fig. 2). When streamflow is primarily

driven by the difference between precipitation and evapotranspiration over annual periods ( $PPT - ET = \text{streamflow}$ ), a strong correlation also exists between annual precipitation and annual streamflow (Jones & Post, 2004). Interpretation of coarse scale syntheses shown in Figs 1 and 2 suggest that: (1) streamflow will decrease with drought, and increase with excess precipitation, and (2) the impacts of drought on streamflow may be more severe in coniferous forest watersheds, whereas the impacts of excess precipitation on streamflow may be more severe in deciduous forest watersheds because of differences in soil water storage between forest types. Our analyses of long-term data from watershed-based EFRs in the US provide an opportunity to more rigorously examine these interpretations.

The ecological and socio-economic significance of streamflow responses to altered climatic regimes will depend on the balance between precipitation and potential evapotranspiration (Fig. 3). For example, areas where precipitation ( $PPT$ ) far exceeds potential evapotranspiration ( $PET$ ) will be less impacted by low precipitation events, but more impacted by high precipitation events that may cause flooding, landslides and increased erosion. By contrast, areas where  $PET$  far exceeds  $PPT$  already have substantial ecosystem and societal adaptations to low precipitation. In these areas, extreme droughts could completely eliminate low flows (Luce & Holden, 2009) and increase susceptibility to secondary disturbances such as insect outbreaks and fire (Dale *et al.*, 2001). From a societal perspective, many areas meet water demands for human consumption by trans-basin water transfers. The most uncertain and perhaps vulnerable areas with respect to streamflow responses may be those where  $PPT$  and  $PET$  are in near balance (Jackson *et al.*, 2009). This is because neither the forested watershed ecosystems nor the socio-economic structures and policies of human communities that depend upon these watersheds have faced pressure in the past to adapt to frequent extreme events (e.g. social and policy decisions such as water conservation policies, protected watersheds, reservoirs, inter-basin transfers, strict zoning). If this contention is true, then among the most vulnerable areas in the continental US are in the southeast and intermountain west.



**Fig. 2** Relationship between annual evapotranspiration (ET) and precipitation ( $PPT$ ) for different forest biomes dominated by either conifers (black symbols) or hardwood (grey symbols) species. Data are from Knight *et al.* (1985), Vose & Swank (1992), Ewel & Gholz (1991), Waring & Schlesinger (1985), Grip *et al.* (1979), Whitehead & Kelliher (1991), Stoy *et al.* (2006). Vose & Swank (1992) IFS sites NC, CAN, WA; Granier *et al.* (2000); Wilson *et al.* (2001). Regression lines and fits for conifer and hardwood data sets are  $ET = 0.72 * PPT - 10.48$ ,  $R^2_{adj} = 0.75$ ,  $P < 0.001$ , respectively; and  $ET = 0.42 * PPT + 46.35$ ,  $R^2_{adj} = 0.74$ ,  $P = 0.002$ . All values are for annual ET except (from Ford *et al.*, 2011a).



**Fig. 3** The ratio of precipitation (*PPT*) to potential evapotranspiration (*PET*) across the continental US.

### Forest management and streamflow response

Considerable information is available on the effects of forest management on streamflow throughout the US (Jones & Post, 2004; Brown *et al.*, 2005). For example, removal of the forest canopy generally increases streamflow for the first few years, but the magnitude, timing, and duration of the responses vary considerably among ecosystems. In some cases, streamflow returns to pre-harvest levels within 10–20 years; whereas in other cases, streamflow remains higher, or can even be lower than pre-harvest flow, for several decades after cutting (Ford *et al.*, 2011b). This wide variation in response patterns is attributable to the complex interactions between climate and vegetation; the former can vary considerably from dry to wet to snow-dominated climatic regimes, and the latter can vary in structure and phenology (coniferous *vs* deciduous forest). It is generally accepted that forest cutting in the US is not a viable tool to provide the extra water required to meet the growing demands for human consumption; however, forest management may be a useful tool for helping both aquatic ecosystems and humans adapt to climatic change (National Research Council of the National Academies, 2008).

### Climate and streamflow in managed forest watersheds

Forest management has the potential to alter the hydrological responses to climate change by influencing biological factors that determine evapotranspiration (*ET*). For example, management activities that favour or replace one species (or several species) over another can alter *ET* through changes in transpiration ( $E_t$ ) or interception ( $E_i$ ). Species vary considerably in transpiration per unit leaf area, and overall whole-tree water use due to differences in rooting depth, tree height, leaf boundary layer resistance, leaf chemistry, and stomatal sensitivity to vapour pressure deficit (Stoy *et al.*, 2006; Ford *et al.*, 2011a). In addition, stand density influences the amount of water evaporated from surfaces (interception) through changes in live and dead leaf, branch, and stem area in a forest. In general, stands with higher stocking levels (i.e. greater leaf and branch area) will intercept a greater percentage of precipitation than those with lower stocking levels. Another major factor is leaf area duration, with the greatest contrast between evergreen and deciduous trees species. Interception is generally higher in conifers than hardwoods (Fig. 2); and the difference is greatest at higher precipitation levels. Higher annual *ET* in conifers has been attributed to a combination of greater transpiration and interception due to year-round leaf area, and autumn, winter, and spring-time transpiration in temperate climates. This suggests that the difference in *ET*

between conifers and deciduous hardwoods might be greatest in the more southern latitudes of the US, where precipitation inputs are evenly distributed across all seasons and winters are relatively mild; indeed, watershed studies confirm this (Brown *et al.*, 2005). For example, in the southern Appalachians, conversion of a deciduous hardwood to a white pine plantation reduced annual streamflow by 20% (Swank & Douglass, 1974).

Forest management can also alter the physical characteristics of the watershed, thereby impacting streamflow. The most direct impacts are those related to activities that create soil disturbances or alter hydrologic flow paths. Generally, if Best Management Practices (BMPs) are implemented, most of the physical impacts related to soil disturbances (e.g. skid trails, landing decks, etc.) are short-lived and have little impact on streamflow over the long-term. In contrast, the construction of roads and associated engineering related to road surfacing, drainage, culvert design and location are much longer-lasting; and, depending on the design and surface area impacted, these can permanently alter hydrologic flow paths.

### Quantifying management, climate, and streamflow interactions

To examine the interactions among forest management, climate change, and streamflow, we examined historical climate and streamflow records from several EFRs located throughout the continental US (Table 1). Our objectives were: (1) to use a simple empirical model to quantify the sensitivity of *ET* and streamflow to forest cutting, warmer temperatures, and greater precipitation variability on all sites; (2) to use gauged watershed data on a subset of EFRs sites to quantify the impacts of forest management treatments on water yield; and (3) to assess the interactions between observed treatment response and climate at one of the EFR sites (Coweeta Hydrologic Laboratory) with a wide range of forest management treatments. This paper summarizes results of our static modelling on all experimental forests, with more intensive analyses on the effects of cutting alone, and cutting followed by species conversion, from a subset of four EFRs (Coweeta, Fernow, Hubbard Brook, and Marcell). Further details of the watersheds and experimental treatments are provided in Adams *et al.* (2008).

## METHODS

### Modelling sensitivity to management and climate interactions among all sites

We used the simple model described by Zhang *et al.* (2001) to examine the potential sensitivity of evapotranspiration (*ET*) to forest cutting and climate change among all sites:

$$ET = \left( \frac{1 + w \frac{PET}{PPT}}{1 + w \frac{PET}{PPT} + \frac{PPT}{PET}} \right) \times PPT$$

Mean *ET* for each experimental site under forested and cut conditions was estimated as a function of potential *ET* (*PET*), precipitation (*PPT*), and an empirical parameter ( $w = 0.5$  or  $2.0$  for cut or forested catchments, respectively). This model was chosen due to the small number of parameters and the ease of data availability for all of the sites noted in Table 1. Potential *ET* for each site is estimated as a function of air temperature using Hamon's method (Sun *et al.*, 2005) and *PET* was increased by 10% to simulate the effects of climate change. For perspective, a 10% increase in *PET* represents an increase of annual air temperature of about 1.5°C at Coweeta. Change of annual precipitation by 20% would be considered an extreme in increase in climate variability. Streamflow was estimated by  $PPT - ET$ , and streamflow response was estimated by subtracting cut forest streamflow from uncut forest streamflow. Percent response was calculated as the absolute response divided by the predicted streamflow for uncut forest using site specific average climatic conditions.

## Quantifying the effects of management on streamflow response using gauged watershed data

The effect of treatment on annual water yield was determined using the paired-watershed approach on a subset of watersheds (Coweeta, Fernow, Hubbard Brook, Marcell) that met the data conditions (e.g. data availability, sufficient record length, corresponding climatic data, etc.) required for more complex statistical analyses described below. This method uses the relationship between gauged streamflow from watersheds similar in size, aspect, elevation, pretreatment cover conditions, and climate. In the subsequent treatment period, one watershed serves as a control and remains undisturbed while a treatment is applied to the other watershed. One of the advantages of using the paired-watershed approach is that year-to-year variation in climate is “adjusted” for and the treatment effect is isolated. If this adjustment does not account for all the post-treatment variation in streamflow, then the inference is that the managed watershed responds differently to climate than what would have been expected had the treatment not occurred.

We applied statistical models that could identify whether the forest management treatments resulted in vegetation that responded significantly differently to climate than what would have been expected if the catchment had not been treated (i.e. we tested for climate by management treatment interactions). Our analysis focuses primarily on streamflow response ( $D$ , cm year<sup>-1</sup>), which is the difference in streamflow of the treated watershed ( $Q_t$ ) from the streamflow that would be expected if the watershed were untreated ( $\hat{Q}_t$  when  $M = 0$ );  $D$  values = 0 indicate no response,  $D$  values < 0 indicate decreased flow, and  $D$  values > 0 indicate increased flow.

Post-treatment streamflow was explained by the following model:

$$\hat{Q}_T = bQ_C + \left[ M \cdot c \cdot \left( h - \frac{1}{1 + \exp^{k_1 - k_2 t}} \right) \right] + \left[ M \cdot d \cdot PPT \cdot \left( h - \frac{1}{1 + \exp^{k_1 - k_2 t}} \right) \right]$$

where  $Q_C$  = streamflow from the untreated catchment;  $M = 0$  prior to treatment and 1 after treatment;  $PPT$  = precipitation;  $b$ ,  $c$ , and  $d$  are coefficients for terms;  $k_1$  and  $k_2$  describe the slope and intercept of the decline in streamflow with vegetation recovery through time; and  $h$  sets the level of the logistic function.

Further details of the statistical analyses are provided in Ford *et al.* (2011b); however, the most important metric for determining if management activities alter the resulting vegetation in the treatment watershed such that it responds differently to climate than the unmanaged watershed with respect to streamflow responses is one of the parameters ( $d$ ) in the model that fits temporal variation in streamflow response. As shown in the model above, when  $d$  is significantly different than zero, post-treatment streamflow is further modified by precipitation ( $PPT$ ).

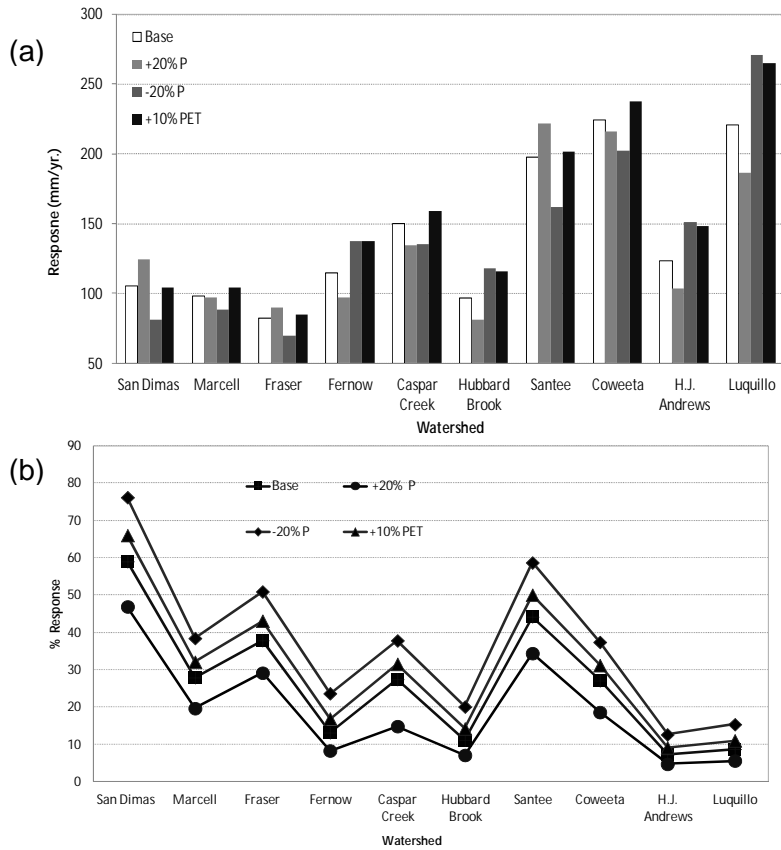
## RESULTS

### Sensitivity to climate change and management interactions among sites

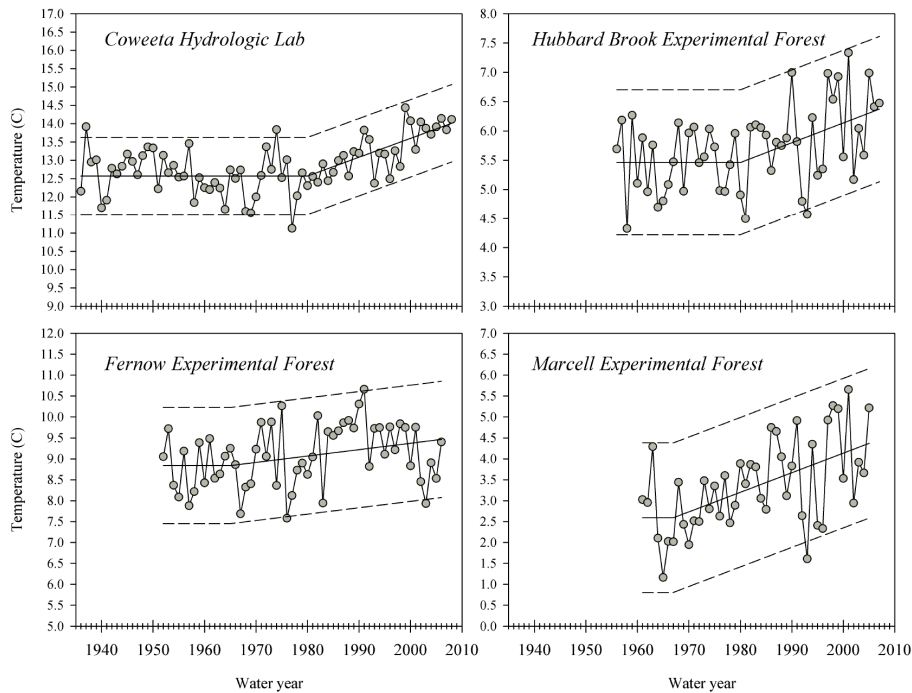
From the Zhang *et al.* (2001) model, we conclude that the Santee, San Dimas, and Marcell sites exhibited the greatest sensitivity to forest cutting, based on percent response at the base level; the Coweeta, H. J. Andrews, Luquillo, and Hubbard Brook sites exhibited the least sensitivity; and the remaining sites (Fernow, Fraser, and Caspar Creek) were intermediate in simulated changes in streamflow response from cutting (Fig. 4(a),(b)). For all three of the most sensitive sites, the streamflow response to cutting also differed considerably among all climate scenarios (base climate and three hypothetical climates). In contrast, the four sites that are least responsive to cutting under current condition also had the least response to climatic changes. Because the same  $w$  was applied to all sites in the simulation, the results of this analysis demonstrate how site-specific climatic conditions can influence response patterns.

### Observed climate variation

Although all four of the intensively analysed EFR sites experienced an increase in temperature over the past 20–30 years (Fig. 5), temperature was not a significant parameter in statistical

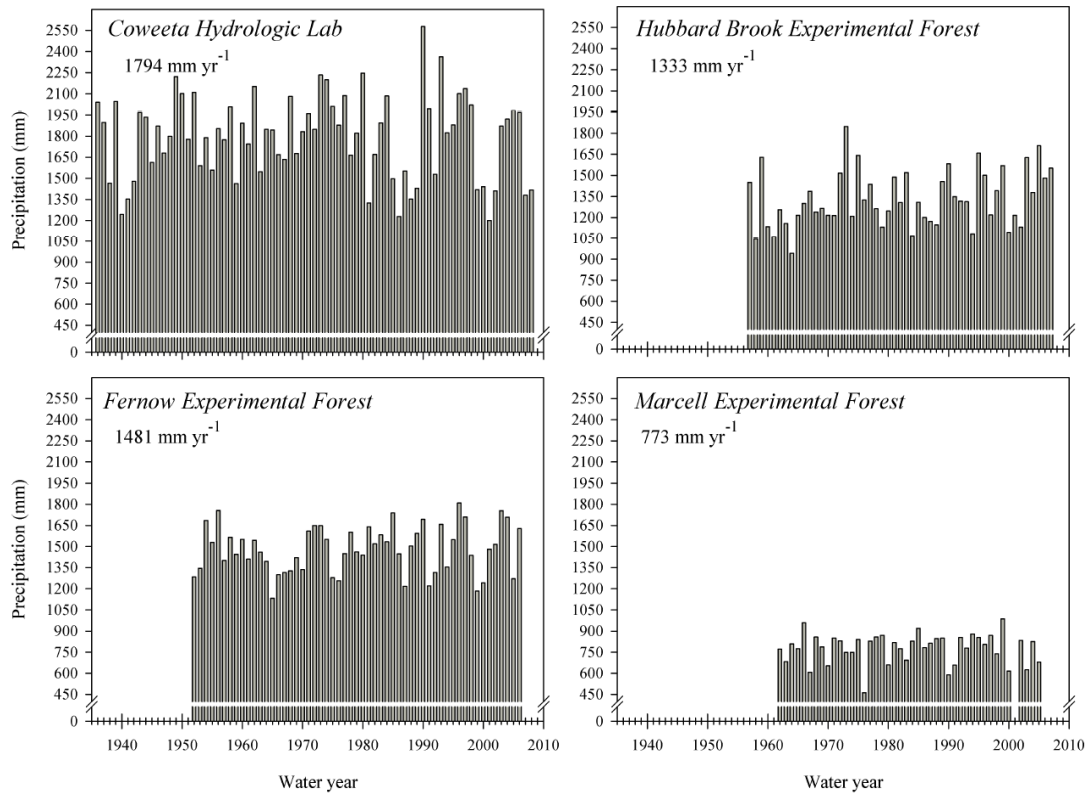


**Fig. 4** Simulated absolute ( $\text{mm year}^{-1}$ ) (a) and relative (%) (b) first year streamflow response to cutting using long-term mean measured climate (base), and simulated climate regimes that increase precipitation (+20% *PPT*), decrease precipitation (-20% *PPT*), and increase *PET* (+10% *PET*).



**Fig. 5** Long-term average annual (water year) air temperature. All sites exhibited a significant increase during the period of record noted by the break point in the solid line. Breakpoints were determined using a time series intervention model (detailed methods described in Ford *et al.*, 2011b). Dashed lines are 95% confidence intervals about the modelled mean.





**Fig. 6** Long-term annual (water year) precipitation inputs. Long-term annual precipitation is also given for reference in each panel.

**Table 2** Parameters for model explaining post-treatment streamflow<sup>1</sup>. Model parameters represent a combination of leaf area index recovery and interactions with climate. The  $d$  parameter explains variation related to post-treatment streamflow interactions with precipitation. Temperature was never significant in the models.

| Experimental forest | Management                        | Parameters |           |         |        |        |        |
|---------------------|-----------------------------------|------------|-----------|---------|--------|--------|--------|
|                     |                                   | $b$        | $c$       | $d$     | $k1$   | $k2$   | $H$    |
| Coweeta             | Sp. conversion (S)                | 0.94**     | -29.43**  | 0.31**  | 5.16** | 0.46** | 0.29** |
|                     | Sp. conversion (N)                | 0.84**     | 13.13     | 0.18**  | 3.47** | 0.34** | 0.56** |
|                     | Clear-cut high elev.              | 0.94**     | 1.59      | 0.18**  | .      | 0.14** | 0.83** |
|                     | Clear-cut low-elev                | 1.15**     | 76.08**   | -0.29** | 3.23   | 1.21   | 1.21** |
|                     | Coppice <i>first cut</i>          | 0.92**     | 62.35**   | -0.05   | .      | 0.22** | 1.14** |
|                     | <i>second cut</i>                 |            | 62.88*    | -0.03   | .      | 1.23** | 1.14** |
|                     | Old-field succession              | 0.85**     | -141.10** | 0.86**  | .      | 0.37** | 0.88** |
| Fernow              | Low-elev clearcut (no BMP)        | 0.94**     | -29.43**  | 0.31**  | 5.16** | 0.46** | 0.29** |
|                     | Clearcut, fert 3x/yr since 1989   | 0.99**     | -7.93     | 0.10*   | 2.62   | 0.31   | 1.13** |
|                     | Sp. conversion <i>Picea abies</i> | 0.82**     | 32.70**   | .       | 5.31** | 0.31** | 0.44** |
|                     | Low-elev clearcut + herbicide     | 1.22**     | 36.76**   | .       | .      | 0.15** | 1.03** |
| Hubbard Brook       | Low-elev clearcut (WS2)           | 1.11**     | 39.89**   | .       | .      | 0.36** | 0.90** |
|                     | Low-elev clearcut (WS4)           | 1.07**     | 23.02**   | .       | .      | 0.48** | 0.85** |
|                     | Low-elev clearcut (WS5)           | 0.97**     | -71.33*   | 0.59*   | .      | 0.11   | 0.86** |
| Marcell             | Clearcut                          | 3.21**     | -85.34**  | 1.50**  | .      | 0.26** | 1.00** |

Notes: All models were significant at  $P < 0.001$ . Model  $R^2 > 0.99$  for all sites and treatments. Superscripts denote parameter estimates significant at  $P > 0.10$  (\*), and  $P > 0.05$  (\*\*).

<sup>1</sup>Post-treatment streamflow ( $Q_T$ ) is explained by the following model:

$$\hat{Q}_T = bQ_C + \left[ M \cdot c \cdot \left( h - \frac{1}{1 + \exp^{k1 - k2t}} \right) \right] + \left[ M \cdot d \cdot PPT \cdot \left( h - \frac{1}{1 + \exp^{k1 - k2t}} \right) \right]$$

models explaining streamflow response to management treatments. This suggests that observed increases affected vegetation in the reference and treated watershed in a similar way, not that temperature was not an important factor affecting *ET* and streamflow. Most sites also exhibited a large amount of variation in annual precipitation (Fig. 6) and precipitation was a significant variable in the post-treatment streamflow model; in other words, the vegetation on the managed watershed responded differently to annual precipitation than the vegetation on the reference watershed.

### Conversion of hardwood forests to coniferous forests

In both the southern (Coweeta) and central (Fernow) Appalachians, conversion of deciduous hardwood to coniferous forests reduced streamflow considerably from what would have been expected (Table 3). After an initial increase in streamflow ranging from 11% (Coweeta S-facing) to 31% (Coweeta N-facing) in the first 2 years after treatment, average streamflow ranged from -21% (Coweeta S-Facing) to -34% (Fernow) over the most recent five years, and from -3% (Fernow) to -17% (Coweeta S-facing) for the entire post-treatment period. Hence, conversion of deciduous forests to coniferous forests resulted in lower streamflow from what was expected at both Coweeta and Fernow.

**Table 3** Post-treatment streamflow response expressed as a percentage increase (+) or decrease (-) relative to expected streamflow.

| Experimental forest | Management                           | Initial response (% , first two years) | Average response (%) | Current response (% , most recent five years) |
|---------------------|--------------------------------------|--|----------------------|---|
| Coweeta             | Sp. conversion (S) to pine           | 10.8                                   | -17.4                | -20.6   |
|                     | Sp. conversion (N) to pine           | 30.5                                   | -14.9                | -27.0   |
|                     | Clear-cut high elev                  | 10.3                                   | -1.4                 | -2.6  |
|                     | Clear-cut low-elev                   | 29.2                                   | 10.9                 | 7.2   |
|                     | Coppice*                             | 44.1                                   | 14.4                 | 10.2  |
|                     | Old-field succession                 | 14.3                                   | -0.5                 | -6.5  |
| Fernow              | Low-elev clearcut (no BMP)           | 14.0                                   | 1.7                  | 0.1   |
|                     | Clearcut, fert 3x/yr since 1989      | 10.8                                   | 3.7                  | 5.1   |
|                     | Sp. conversion to <i>Picea abies</i> | 18.2                                   | -2.7                 | -33.9   |
|                     | Low-elev clearcut + herbicide        | 26.5                                   | 7.6                  | 4.1   |
| Hubbard Brook       | Low-elev clearcut (WS2)              | 18.9                                   | -1.6                 | -0.1  |
|                     | Low-elev clearcut (WS4)              | 6.9                                    | -2.4                 | -2.6  |
|                     | Low-elev clearcut (WS5)              | 1.7                                    | 1.3                  | -1.5  |
| Marcell             | Clearcut                             | 51.7                                   | 8.8                  | -8.6  |

Notes: for CWT coppice management, initial response was the mean of the two years for each cut ( $n = 4$ ).

In the central (Fernow) Appalachians, there was no interaction between post-treatment vegetation and climate on streamflow response for the deciduous to conifer conversion (non-significant  $d$  parameter, Table 2); the coniferous forest responded to climate in a similar way to what would have been expected if it had remained a hardwood forest. However, in the southern Appalachians (Coweeta), both of the coniferous conversion watersheds exhibited a significant interaction between post-treatment vegetation and climate on streamflow response (Table 2). The implications for these interactions are discussed below.

### Forest cutting

Initially, cutting the forest vegetation resulted in 2% (Hubbard Brook WS 5) to 52% (Marcell) greater streamflow compared to what was expected, with an average increase of 21% (Table 3).

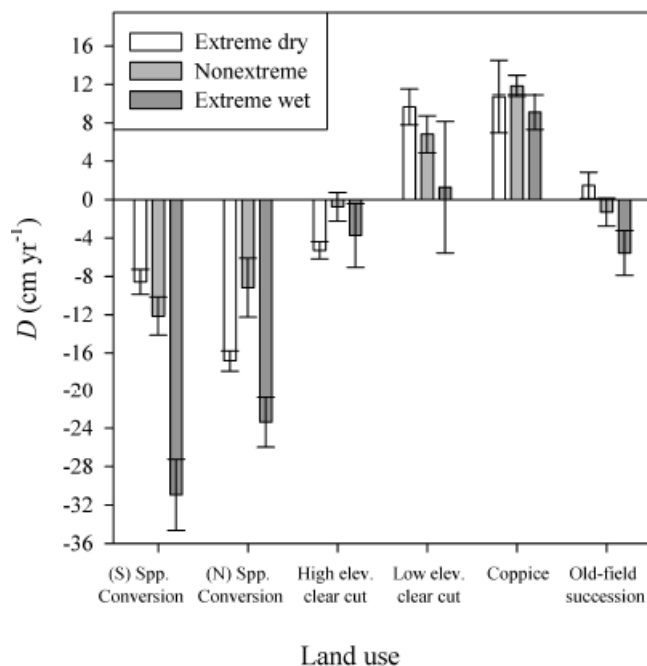
After a mature forest structure has been restored, streamflow response to the original forest management activity ranged from a decrease of 9% (Marcell) to an increase of 10% (Coweeta Coppice). Integrated across all observed years, management resulted in a more narrow range of responses, from decreases of 2% (Hubbard Brook WS4) to increases of 14% (Coweeta Coppice). The wide range in response patterns among sites over time reflects differences in treatments, soils, species and vegetation recovery, and climatic regimes among sites.

In addition, many of the watersheds exhibited a significant interaction between resulting vegetation and climate on streamflow (parameter  $d$ , Table 2), suggesting a different response among management treatments to future climatic conditions. The potential implications are explored in the next section using results from the southern Appalachians (Coweeta).

### Interactions with climate

The results presented above provide useful information on how streamflow responds to a range of management treatments across observed climatic conditions. As noted previously, the frequency of extreme rainfall events is likely to increase in the future. To better illustrate how resulting vegetation from management treatments responds differently to climate than that in unmanaged catchments, we extracted streamflow responses during extreme precipitation event years and separated these into three categories: responses in extreme dry, non-extreme, and extreme wet event years using the standard precipitation index or SPI (methods for characterizing precipitation extremes are presented in Ford *et al.*, 2011b). We then examined differences in response among the categories. Here, we present results from managed catchments in the southern Appalachians (Coweeta) as an example of the potential effects that forest management can have on vegetation, and in turn how that vegetation responds to extreme climate event years differently than unmanaged catchments. We specifically focus on Coweeta due to the large number of different management treatments conducted within the same basin.

Results indicate that forest management can influence streamflow response to climate extremes (Fig. 7; Ford *et al.*, 2011b). The most dramatic interactions were observed on the



**Fig. 7** Mean streamflow response ( $D$ ) to forest management activities (Ford *et al.*, 2011b). Response data are binned into SPI categories: extreme wet years, extreme dry years, and non-extreme years. Negative  $D$  values indicate that streamflow has been reduced by the treatment and positive values indicate that streamflow has been increased. Error bars show standard error.

hardwood to pine conversion treatments, where during extreme wet years streamflow was substantially lower than expected, compared to the expected response during non-extreme years. During extreme wet years, streamflow response was lower by an additional ~2 000 000 and ~700 000 m<sup>3</sup> ha<sup>-1</sup> year<sup>-1</sup> on the S and N facing pine watersheds, respectively, relative to the streamflow response on the deciduous hardwood watershed. Our interpretation is that the higher *ET* in pine stands results in greater soil water storage relative to the hardwood stands. In other words, higher *ET* in the conifer stands means that the soils are less saturated and have a greater capacity to absorb and store rainfall in wet years. Conversion of hardwood stands to coniferous stands may be a viable management option to mitigate the impacts of higher precipitation, especially in flood prone areas. Interestingly, the response to extreme dry conditions varied based on watershed aspect. On north facing stands, streamflow response was also significantly lower (relative to average years) during extreme drought, but that was not the case on south facing stands. During dry years, the difference between hardwood and pine was an additional ~800 000 m<sup>3</sup> ha<sup>-1</sup> year<sup>-1</sup> lower on the N facing watershed. If streamflow is lower during extreme drought, then converting hardwood stands to conifers could exacerbate the impacts of increased drought frequency on streamflow.

Responses in other treated watersheds were more subtle, but revealed some interesting patterns. For example, on the high elevation clearcut, streamflow response was significantly lower during extreme dry conditions indicating that cutting exacerbated drought impacts. However, this pattern was not observed on the low elevation clearcut. However, annual streamflow response was never greater on the cut watersheds during extreme wet years indicating that cutting does not exacerbate the effects of greater precipitation on streamflow response.

## CONCLUSIONS

Land managers on both public and private land ownerships in rural and urban landscapes are faced with the challenge of how to manage today's forests to adapt to and mitigate the effects of future climate change on a variety of ecosystem services. Management activities implemented on forest ecosystems today will influence ecosystem responses to future climate conditions. Natural resource managers require new insights into management implications and potentially new approaches to silviculture and other management practices. The network of US Forest Service EFR's provides a unique and immensely valuable resource to examine how management can be used to mitigate (or adapt to) the effects of climate change on water resources. Results of our analyses suggested that managed forests respond differently to precipitation variability than control forests. The most dramatic interactions were observed on the hardwood to pine conversion treatments, where during extreme wet years streamflow was substantially lower (e.g. an additional 20 cm<sup>-1</sup> year<sup>-1</sup> lower) when compared to the expected response during non-extreme years. On the other hand, if streamflow is lower during extreme drought, then converting hardwood stands to conifers could exacerbate the impacts of increased drought frequency on streamflow. These results suggest that the trade-off between managing forests for opposite extremes need to be carefully considered.

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