# Assessing streamflow sensitivity of forested headwater catchments to disturbance and climate change in the central Appalachian Mountains region, USA

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# Assessing streamflow sensitivity of forested headwater catchments to disturbance and climate change in the central Appalachian Mountains region, USA

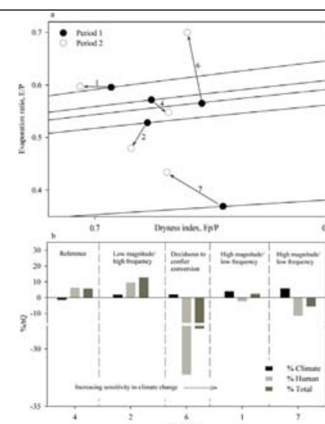
David Young[a,1], Nicolas Zégre[a,*], Pamela Edwards[b], Rodrigo Fernandez[a]

[a] School of Forestry and Natural Resources, West Virginia University, 322 Percival Hall, Morgantown, WV 26506, USA
[b] US Forest Service, Northern Research Station, PO Box 404, Parsons, WV 26287, USA

## HIGHLIGHTS

- Forest disturbance important for setting streamflow sensitivity to climate change
- Sensitivity to climate change greater in disturbed than undisturbed catchments
- Disturbance masked or amplified climate change depending on magnitude and frequency

## GRAPHICAL ABSTRACT

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## ABSTRACT

Forest headwater catchments are critical sources of water, but climate change and disturbance may threaten their ability to produce reliable and abundant water supplies. Quantifying how climate change and forest disturbances individually and interactively alter streamflow provides important insights into the stability and availability of water derived from headwater catchments that are particularly sensitive to change. We used long-term water balance data, forest inventory measurements, and a multiple-methods approach using Budyko decomposition and paired catchment models to assess how climate change and forest disturbances interact to alter streamflow in five headwater catchments located along a disturbance gradient in the Appalachian Mountains, USA. We found that disturbance was the dominant driver of streamflow changes; disturbed catchments were more sensitive to climate change than the undisturbed catchment; and disturbance was an important factor for a catchment's sensitivity to climate change, principally through changes in species composition and xylem anatomy. Streamflow sensitivity to climate change increased with increasing proportion of diffuse porous species, suggesting that not all disturbances are equal when it comes to streamflow sensitivity to climate change. Climate change effects were masked by disturbance in catchments with high magnitude/low frequency disturbances and amplified in a catchment with low magnitude/high

* Corresponding author.
*E-mail address:* nicolas.zegre@mail.wvu.edu (N. Zégre).

[1] Hydrologist (retired).

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frequency disturbance. Furthermore, critical assumptions of Budyko decomposition were assessed to evaluate the efficacy of applying decomposition to the headwater scale. Our study demonstrates the efficacy and usefulness of applying decomposition to scales potentially useful to resource managers and decision makers. Our study contributes to a more thorough understanding about the impacts of climate change on disturbed headwater catchments that will help managers to better prepare for and adapt to future changes.


## 1. Introduction

Forest headwater catchments play a vital role in provisioning fresh water services to ecosystems and downstream communities (Lowe and Likens, 2005; Caldwell et al., 2016; Duan et al., 2018). Worldwide, headwaters provide between 60 and 80% of the world's fresh water resources (Payne et al., 2002), with about a third of the world's largest cities receiving a significant proportion of drinking water from forests (Dudley and Stolton, 2003). Despite humanity's dependence on forested mountain catchments for fresh water, climate change and human activities threaten water supplies by altering the amount of precipitation (P) that is partitioned in to streamflow (Q) and evapotranspiration (E) (Caldwell et al., 2016). Understanding how climate change and human activities alter streamflow (ΔQ) from forested headwater catchments is critical for developing policies and practices that ensure future water resources reliability and security from these sources.

Streamflow from forest headwater catchments is sensitive to disturbance and climate change due to the small contributing areas and shallow soils of these catchments (Campbell et al., 2011) and the dominant role that trees play in partitioning precipitation (P) into evapotranspiration (E) and streamflow (Q) (Rodriguez-Iturbe and Porporato, 2004). Forest harvesting generally increases Q over the short-term by decreasing canopy interception and E (Bosch and Hewlett, 1982; Hornbeck et al., 1993; Campbell and Doeg, 1989; Stednick, 1996; Brown et al., 2005). As forests regrow, Q can return to similar pre-disturbance levels. However, because canopy interception and tree water use differs widely among species (e.g., Wullschleger et al., 2001; Oren and Pataki 2001; Ford et al., 2011a,b), changes in post-disturbance forest composition can alter Q over the long-term (Jones et al., 2012). The effects of climate change on Q largely depends on how changes in atmospheric conditions alter the amount, timing, and distribution of P. Q increases in recent decades throughout the United States (Lins and Slack, 1999; McCabe and Wolock, 2002) have been linked to increases in P (Karl and Knight, 1998; Krakauer and Fung, 2008; Renner and Bernhofer, 2012) that are theoretically consistent with water cycle intensification due to climate warming (Trenberth, 1999; Huntington, 2006). A warming climate also provides more energy for E, potentially decreasing Q. Alternatively, elevated atmospheric $CO_2$ can decrease leaf stomatal conductance and transpiration (Ehleringer and Cerling, 1995; Beerling, 1997; Prentice and Harrison, 2009), potentially increasing Q (Ainsworth and Rogers, 2007; Betts et al., 2007; Warren et al., 2011). How climate change and forest disturbance interact and influence ΔQ is poorly understood. Quantifying this interaction could provide important insights into the stability of Q derived from forested headwater catchments that will be useful for developing practices for ecosystem management under expected future changes (Ford et al., 2011b; Vose et al., 2012) and for decreasing model uncertainty in current predictive hydrologic models (Burt et al., 2015; Kelly et al., 2016).

Causes of hydrological change are commonly separated into climatic drivers, largely described by changes in P and potential evapotranspiration (Ep), and non-climatic drivers, such as human activities (Zhang and Schilling, 2006; Wang and Cai, 2010); vegetation response to past disturbances (Jones, 2011); water diversion and the construction of reservoirs (Döll et al., 2009), and side effects of $CO_2$ fertilization (Roderick et al., 2015). Approaches for quantitatively separating the impacts of climate ($\Delta Q_C$) and non-climate factors (e.g. human activities, $\Delta Q_H$) on streamflow generally fall into three categories: empirical statistics, hydrologic modeling, and elasticity-based methods that include the Budyko framework (Wang, 2014; Wu et al., 2017). Empirical statistics are used to establish a relationship between Q and climate variable(s) of interest (e.g. P, Ep), requiring long-term hydrometeorological data that may or may not include a period of negligible human activity (Wang, 2014). These methods can include time-trend analyses (Zhao et al., 2010; Zhang et al., 2011; Zhao et al., 2014), the double-mass curve method (Gao et al., 2016), and regression-based approaches (Schade and Shuster, 2005; Zhao et al., 2010,2014; Kelly et al., 2016), including the paired catchment approach (Bosch and Hewlett, 1982; Stednick, 1996; Box and Jenkins, 1976). Hydrological modeling uses mathematical models to quantitatively predict ΔQ induced by changes in climate and/or human activity (e.g. Ma et al., 2010; Li et al., 2014; Kim et al., 2014). Climate elasticity methods evaluate the effects of climate change on Q by estimating the elasticities of macro-climate processes represented nominally by P and Ep (Schaake, 1990; Dooge, 1992; Wu et al., 2017). Hence, the Budyko framework (Budyko, 1974) is frequently used to study climate elasticity of streamflow in larger catchments ($>10^2$ km$^2$) (e.g., Donohue et al., 2007, 2011; Wang and Hejazi 2011; Patterson et al., 2013).

Increasingly, however, Budyko-based elasticity approaches are being applied to smaller spatial scales to assess water balance sensitivity to changes (e.g., Jones et al., 2012; Tetzlaff et al., 2013; Creed et al., 2014), but its efficacy at finer spatial scales largely remains untested. Complicating understanding of the effects of climate change on streamflow is that past and present human activities can mask or amplify ΔQ attributed to climate change (Patterson et al., 2013;Jones et al., 2012). Furthermore, most attribution studies have focused on larger spatial scales ($>10^2$ km$^2$) (Patterson et al., 2013; Wang and Hejazi, 2011) or on undisturbed, headwater-scale reference catchments (Creed et al., 2014; Patterson et al., 2013), often using just one method. A multiple-method approach can reduce uncertainty and increase confidence in attributing impacts of climate change and human activities on ΔQ (Wu et al., 2017).

An in-depth understanding of all of the natural and anthropogenic factors that influence partitioning P into E and Q is required to understand the individual and interactive effects of climate change and human activities on Q (Chawla and Mujumdar, 2015; Wu et al., 2017). Expanding our understanding of how climate change interacts with different types of forest management practices is critical for developing sound policies that balance the needs of society for water, timber, and other forest ecosystem services.

In this study, we used two approaches to quantify of the effects of climate change and forest disturbances on long-term hydrology for five experimental headwater catchment: Budyko decomposition (Wang and Hejazi, 2011) and the paired catchment approach (Hewlett, 1971; Bosch and Hewlett, 1982). The catchments are part of the Fernow Experimental Forest, one experimental forest of numerous long-term studies in the USA established by the USDA Forest Service to study the effects of forest management on hydrology (water quantity and water quality), ecology, and timber (Ice and Stednick, 2004a,b). Here we leverage the power of long-term hydrology and climate records from the Fernow to (1) quantify trends in climate and water balance components over a sixty-year period; (2) quantify individual and interactive effects of human activities and climate change on Q across a gradient of disturbances; (3) and evaluate the efficacy of applying Budyko decomposition to the headwater-scale.

## 2. Materials and methods

### 2.1. Study area

The USDA Forest Service Fernow Experimental Forest (39°03′ N, 79°67′ W) is located in the highlands region of the central Appalachian Mountains in north central West Virginia (Fig. 1). This region of the mid-Atlantic USA is headwaters to both the Potomac River which drains east to Washington, D.C. and the Chesapeake Bay, and the Ohio River, which drains west to the Gulf of Mexico and is the largest tributary of the Mississippi River (Brooks et al., 2018). Runoff generated from this relatively small (62,038 km$^2$), heavily forested, and sparsely-populated (approximately 1.8 million people) state provides precipitation-driven Q directly to ~9 million people, or about 3% of the US population (Bureau, 2009).

The long-term historical climate of the Fernow is cool, humid, and continental. Historical twentieth century annual air temperature averages 9.3 °C while monthly temperatures range from −2.8 °C in January to 20.4 °C in July (Adams et al., 2012). Precipitation is relatively evenly distributed throughout the year, averaging 1450 mm, but is highly variable in late summer and early fall due to tropical storms (Smith et al., 2011). Elevation, topography, and prevailing winds, combined with air masses that originate in the north polar continental or south Gulf maritime regions, produce frequent storm events and variable temperatures (Weedfall and Dickerson, 1965). In the winter, frontal storm systems bring cold temperatures and frequent rain and snowfall that result in limited and transient snowpacks. Summers are warm and humid, dominated by local and regional convective storms. Forests are mixed deciduous hardwoods dominated by northern red oak (*Quercus rubra*), sugar maple (*Acer saccharum*), black cherry (*Prunus serotina*), yellow-poplar (*Liriodendron tulipifera*), white oak (*Q. alba*), chestnut oak

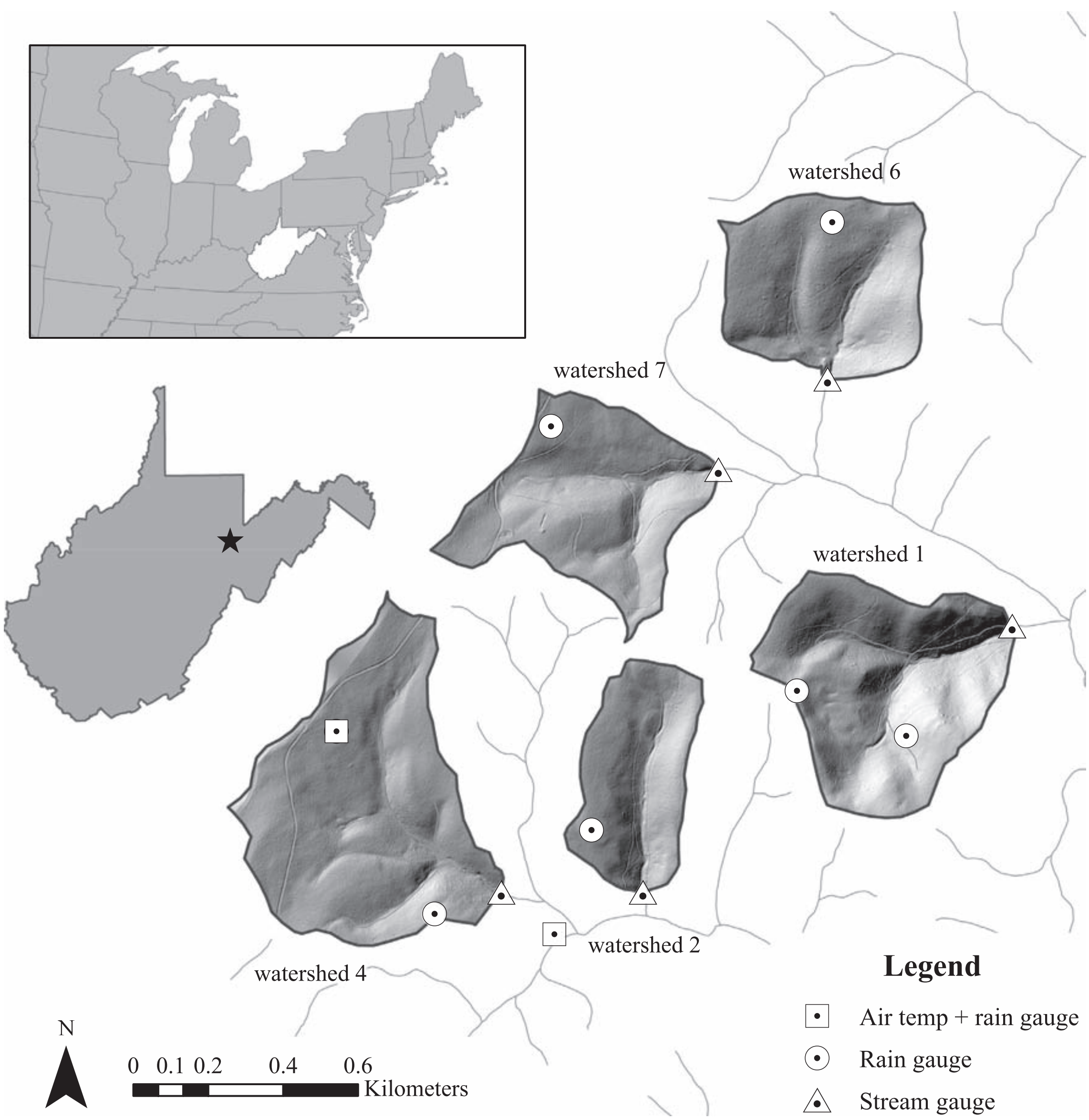


**Fig. 1.** Site map of the Fernow Experimental Forest (39°03′N, 79°67′W), located near Parsons, WV USA. Shown are stream gauges and climate stations of experimental catchments (watersheds 1, 2, 6, 7) and the reference catchment (watershed 4) examined in this study.

(*Q. prinus*), and red maple (*A. rubrum*) and other mixed hardwood and herbaceous species (Schuler, 2004;Adams et al., 2012).

Hillslopes at the Fernow are steep (mean slope = 16%); elevations range between 706 m and 843 m; and soils are shallow (<1 m) and well-drained (Adams et al., 2012). Streamflow is dominated by base flow, and storm flow is flashy, rapidly responding to P (Adams et al., 2012). The water year (WY) at the Fernow is considered to extend from May 1 to April 30 due to consistent and dependable soil saturation in the spring; the growing season extends from May through October (Kochenderfer et al., 1990; Adams et al., 2012).

### 2.2. Disturbance history

Forests at the Fernow are representative of the highly dynamic eastern US deciduous forests that have long legacies of disturbances over the last 200 years. These include forest harvesting, agriculture, natural gas development, and insect defoliation, to name a few. The Elklick Run watershed is the dominant watershed of the Fernow and was initially harvested in 1903 and 1911 during railroad logging era (Trimble, 1977). Forest fires have largely been absent from the forest over the last century and much of the watershed was not farmed (Adams and Kochenderfer, 2014). In the 1930s, chestnut blight (*Cryphonectria parasitica*) reduced the standing volume of timber by 25% (Weitzman, 1949). When the paired catchment studies at the Fernow were initiated in the 1950s, catchments were occupied by an even-aged, 35- to 45-year old mixed mesophytic forests, dominated by oak and maple species (Schuler, 2004).

Following pre-disturbance calibration periods, experimental headwater catchments were harvested using different forest practices to study the effects of contemporary forest management practices on water quantity and water quality (Table 1). Watershed 4 was not treated and was retained as a long-term reference catchment. This study focused on four of the treated catchments, watersheds 1, 2, 6, and 7 and the reference catchment, watershed 4 (Fig. 1). Catchments were selected based the gradient of disturbance and overlap and completeness of their hydro-climatic records (55–60 years).

Treatments in watershed 1 removed all trees greater than 15.2 cm in diameter at breast height (dbh, measured at 1.37-m off the ground) using clear-cut harvest techniques with no soil or water protection measures. Following disturbance, the forest was left to regenerate naturally. In watershed 2, repeated diameter-limit harvest of all trees > 43.2 cm were used periodically (~10-year recurrence), collectively disturbing about 30% of the catchment area over the entire study period. Beginning in the 1960s, half of watersheds 6 and 7 were clear-cut and treated with herbicides to inhibit natural growth (Patric and Reinhart, 1971). A few years later, a second clear-cut entry removed the remaining half of each catchment and herbicides were applied across entire catchments for another two years. Watershed 6 was replanted with Norway spruce (*Picea abies*) that remains as the dominant land cover today, whereas watershed 7 was left to recover naturally to mixed hardwoods.

### 2.3. Data collection

Precipitation has been measured in watersheds 1, 2, and 4 since 1951 and in watersheds 6 and 7 since 1956 using a network of continuous weighing and standard rain gauges and two climate stations (Fig. 1) (Edwards and Wood, 2011a,b). Precipitation is spatially weighted across catchments using the Thiessen polygon approach (Edwards and Wood, 2012). Air temperature has been measured at two climate stations, one located at the top of watershed 4 and the other located southwest of watershed 2 (Fig. 1). The daily air temperature data set contained gaps between 1989 and 2010. When three or fewer successive values were missing, data were reconstructed using a cubic spline interpolation method (Fritsch and Carlson, 1980). When greater than three successive observations were missing, air temperature was reconstructed using linear regression between air temperature measured at the nearby (<2 km) NOAA Parsons 1 NE climate station (GHCND: US00466867) using a moving 20-day window before and after missing observations (Stoy et al., 2006).

Potential evapotranspiration was estimated using a temperature-based version of the Priestly-Taylor model that is part of the EcoHydrology package in R (R Core Team, 2013). In place of radiation data, this version uses temperature-based radiation approximations using daily minimum and maximum air temperature, latitude, and day of year. This model was chosen for its simple data requirements and strong performance for estimating Ep in humid environments compared to using measured radiation (Archibald and Walter, 2014). Long-term annual E for each catchment was estimated using the annual water balance, $\bar{E} = \bar{P} - \bar{Q}$, assuming no change in storage (soil water, groundwater) for the Fernow water year (WY, May 1 to April 30).

Streamflow has been recorded at the outlet of each catchment continuously, starting as early as 1951 using a stage recorder and 120° V notch weir, and is expressed on a per unit area basis (mm d$^{-1}$) (Edwards and Wood, 2011c). Watershed 2 has a nearly decade-long gap in daily streamflow data that was reconstructed using the HBV-light hydrologic model (Seibert and Vis, 2012) using daily precipitation and potential evapotranspiration estimated using the Priestly-Taylor model. The period of missing streamflow (01 April 1979 to 04 May 1988) started three months after the January 1978 harvest that removed trees from 4.7 ha using a 43.2-cm diameter limit cut and ended as the May–July 1988 harvest started. To replicate similar land cover conditions that included the disturbance effects, the model calibration period was set using the period between harvests from 01 January 1989 to 31 December 1996. The model was calibrated using a Monte Carlo optimization approach based on 100,000 simulations. The final model was selected based on the highest Nash Sutcliffe efficiency (NSE) (Nash and Sutcliffe, 1970) which was 0.69. NSE values between 0.6 and 0.7 are typical of hydrological modeling studies in the Appalachian Mountains region of the US due to the complex topography and atmospheric forcing (e.g., (Ferrari et al., 2009,Merriam et al., 2017; Robison and Scanlon, 2018,Tao and Barros, 2018). NSE values between 0.5 and 0.7 are often considered satisfactory for model performance evaluation, while above 0.7 are considered good, particularly in studies where rainfall-runoff modeling is not the main focus (Moriasi et al., 2007; Gassman et al., 2007), such as this one.

Minimum, maximum, and mean air temperature, E, Ep, and Q data were aggregated from daily values to the annual time step using the R package HydroTSM (R Core Team, 2013). Trend analyses was performed on all climate and water balance components using the non-parametric Mann-Kendall statistical test using the R package ‘ trend’ (R Core Team, 2013). The Mann-Kendall test requires serial independence (Helsel and Hirsch, 1992) that was tested for using the Ljung-Box test (Ljung and Box, 1978) and partial auto correlation function plots (R Core Team, 2013). Serial correlation was not detected in any of the annual time series. The slope of each trend was calculated as the median of all possible pair-wise slopes (Sen, 1968; Helsel and Hirsch, 1992). Significance was assessed at $\alpha = 0.10$ level.

Forest stand structure data was extracted from publicly available datasets provided by the USDA Forest Service Northern Research Station. Stand structure was inventoried using standard forest inventory techniques by either conducting a 100% inventory of all trees greater than 12.7 cm dbh or by extrapolating stand data from growth plot measurements (Schuler, 2004; Schuler and McGill, 2007; Schuler et al., 2016). For the 100% inventory, each tree was tallied by species and 2.54-cm diameter class. To determine stand structure from growth plot data, each tree on every plot was assigned to a 5.08-cm diameter class based on the method described above and tallied by species. An area factor was calculated from the total area in all

**Table 1**
Characteristics of five experimental headwater catchments located at the Fernow Experimental Forest near Parsons, West Virginia, USA. Table includes forest harvesting treatment descriptions and calibration models used in the paired catchment study analysis.

| Watershed | Year of first complete flow record<br>WY[a] | Size<br>ha | Mean elevation<br>m | Mean slope<br>% | Aspect | Treatment | Approx. disturbance recurrence[b]<br>yrs. | Calibration model[d] | $B0_0$ p-value | $B1_1$ p-value | $R2^2$ | Std. error of estimate<br>mm |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | 1952 | 30.1 | 739 | 22 | NE | Clearcut to 15.2-cm d.b.h. except culls,<br>no soil or water resource protection (1957–58) | 60 | $QTrt,1 = -14.9 + 0.95QRef,1$ | 0.685 | <0.0001 | 0.99 | 18.2 |
| 2 | 1952 | 15.5 | 773 | 17 | S | 43.2-cm diameter limit cut (1958)<br>Repeat treatment on 10.8 ha (1972)<br>Repeat treatment on 4.7 ha (1978)<br>Repeat treatment on 10.8 ha (1988)<br>Repeat treatment on 4.7 ha (1997)<br>Repeat treatment on 10.8 ha (2004) | 10 | $QTrt,1 = -5.63 + 1.06QRef,1$ | 0.747 | <0.00001 | 0.998 | 9.3 |
| 4 | 1952 | 38.7 | 819 | 13 | ESE | None - Reference | 0 | – | – | – | – | – |
| 6 | 1957 | 22.3 | 781 | 13 | S | Clearcut lower half on 11.2 ha (1964) 28[c]<br>Lower half herbicided (1965–69)<br>Clearcut upper half on 11.1 ha (1967–68)<br>Entire watershed herbicided (1968–69)<br>Planted Norway spruce (1973)<br>Entire watershed herbicided (1975, 1980) | | $QTrt,1 = -46.1 + 0.87QRef,1$ | 0.651 | 0.002 | 0.87 | 24 |
| 7 | 1957 | 24.2 | 804 | 14 | E | Clearcut upper half on 12.1 ha (1963–64)<br>Upper half herbicided (1964–69)<br>Clearcut lower half on 12.1 ha (1966–67)<br>Entire watershed herbicided (1967–69) | 28[c] | $QTrt,1 = 24.8 + 1.19QRef,1$ | 0.805 | 0.0005 | 0.93 | 23.8 |

[a] Denotes Fernow Water Year (May 1 to April 30).
[b] N / n, where N = total number of years; n = number of disturbance events.
[c] Clear-cut + herbicide treatments were treated as single disturbance event for each entry, given that herbicide were applied following clear-cut harvest.
[d] Calibration model between longterm Q from treatment catchment and reference catchment watershed 4.

growth plots and total watershed area, and was used to extrapolate the tally from the sampled area to the total stand area.

## 2.4. Analyses

### 2.4.1. Budyko decomposition for quantifying ΔQ attributed to forest harvesting and climate change

Budyko decomposition is an elasticity-based method (Wang, 2014) based on Fu's (1981) formulation of the Budyko framework (Budyko, 1974) that includes the adjustable landscape parameter, $w$, to account for non-climatic factors that influence E such as slope, vegetation, geology, and soils (Padrón et al., 2017). The $w$-parameter controls the shape of the curve and is calibrated for each catchment using long-term P and Ep:

$$\frac{E}{P} = 1 + \frac{E_p}{P} - \left[1 + \left(\frac{E_p}{P}\right)^w\right]^{\frac{1}{w}} \tag{1}$$

Budyko decomposition is commonly used to separate the effects of climate change and human activities on ΔQ between two time periods Wang and Hejazi, 2011; Patterson et al., 2013. It is predicated on the assumption that climate change will result in movement along the curve (e.g., changes in aridity), while changes in other factors (e.g., human activities) will result in movement away from the curve. The directional change assumption is widely used but unsubstantiated, and therefore assessed in this study.

The contributions of climate change and other factors are not absolute but rather calculated relative to a baseline period (Wu et al., 2017). The two periods can be defined using formal statistical methods such as change point analyses (e.g., Pettitt test), double-mass curve test, and trend analysis (e.g., Mann-Kendall test) or by human designation. In this last case, anthropogenic effects on ΔQ are assumed to be weaker in the first period while the second period represents the post-change period where human activities and climate change are assumed to have measurable effects on ΔQ (Wang, 2014; Wu et al., 2017). Another approach is to simply split the long-term hydrological time series data into two equal periods (Patterson et al., 2013). Initially change point analyses was used to identify time periods. However, harvesting treatments occurred early in study period (Table 1) and hence, baseline periods were short (between 5 and 7 years) relative to post-change periods (greater than fifty years) and too short to capture long-term changes in climate. Since the goal of this research was to study how varying degrees of disturbances affected the sensitivity of Q to climate change, the long-term data record was split into two equal time periods, consisting of either 27 or 30 years, depending on records available in each catchment (Table 1).

The first step in Budyko decomposition is to calculate a unique Budyko curve for each catchment. This was accomplished by plotting the coordinate position of the long-term average E/P and Ep/P for the first time period (Period 1, $\phi_1$) and calibrating $w$ for each catchment using Eq. (1). Second, the coordinate position of the long-term average E/P and Ep/P for the second period (Period 2, ($\phi_2$) was plotted.

ΔQ attributed to climate change ($\Delta Q_C$) and to human activities ($\Delta Q_H$), which in our case were different types of forest harvesting treatments, were calculated as the distance between $\phi_1$ and $\phi_2$. The climate change component of the evaporation ratio, $\frac{E_2'}{P_2}$, which is the reference point from which the magnitude of climate change and harvesting activities contribute to ΔQ, was calculated for the second period using the calibrated $w$-parameter from the first period:

$$\frac{E_2'}{P_2} = 1 + \frac{E_{p2}}{P_2} - \left[1 + \left(\frac{E_{p2}}{P_2}\right)^w\right]^{\frac{1}{w}} \tag{2}$$

And finally, $\Delta Q_H$ and $\Delta Q_C$ were calculated as

$$\Delta Q_H = P_2\left(\frac{E_2'}{P_2} - \frac{E_2}{P_2}\right) \tag{3}$$

and

$$\Delta Q_C = P_2\left(1 - \frac{E_2'}{P_2}\right) - Q_1 \tag{4}$$

where $Q_1$ is streamflow from the first period. The relative changes in ΔQ attributed to forest harvesting and climate change between the two periods were calculated as

$$\%\Delta Q_H = \frac{\Delta Q_H}{Q_1} \tag{5}$$

and

$$\%\Delta Q_C = \frac{\Delta Q_C}{Q_1} \tag{6}$$

### 2.4.2. The paired catchment approach for quantifying ΔQ attributed to harvesting and climate change

The paired catchment approach (Hewlett, 1971; Bosch and Hewlett, 1982) was first used at Wagon Wheel Gap, Colorado, in 1910 (Bates, 1921) and since has become a dominant method for detecting and quantifying the effects of forest management practices on hydrology at the headwater-scale (Bosch and Hewlett, 1982; Campbell and Doeg, 1989; Stednick, 1996; Brown et al., 2005;Zégre et al., 2010). It provides an unparalleled framework for de-convolving the complex relationship between human activities and ecosystem responses to change at small scales (Andréassian, 2004; Moran et al., 2008; Jones et al., 2012). It is based on establishing statistical models of measured Q between two or more catchments during a pre-disturbance calibration period. Following the calibration, disturbance treatments (i.e., harvesting) are applied to one catchment (referred to as the treatment catchment) while the other catchment remains unchanged (reference catchment). A central tenant of the approach is that the relationship between paired catchments is statistically stable over time (Zégre et al., 2010). Stationarity is more often assumed (Zégre et al., 2010), particularly when the calibration period is short relative to the post-disturbance period. While additional explanatory variables can be included to account for variations overtime (e.g., seasonality, regrowth, forest succession, productivity) (Zégre et al., 2010; Ford et al., 2011b; Vose et al., 2012; Kelly et al., 2016). Notwithstanding, the current study only included annual Q from the reference catchment which is by far the norm in paired catchment literature (e.g., Dye and Croke, 2003). Furthermore, where common practice is to compare only hydrology from pre-disturbance calibration period to post-disturbance hydrology, here we split the long-term data record in to two periods following the Budyko decomposition approach. Hence, the first period includes hydrology from both undisturbed and disturbed states.

Regression is the standard approach for developing the calibration model and for detecting change following disturbance. Pre-disturbance calibration models were developed for each catchment using mean annual Q from watershed 4, the reference catchment, and mean annual Q for each treated catchment:

$$Q_{Trt,1} = \beta_0 + \beta_1 Q_{Ref,1} \tag{7}$$

where $Q_{Ref,1}$ and $Q_{Trt,1}$ were measured streamflow from the reference catchment, watershed 4, and each treated catchment during the calibration period ($t = 1$) and $\beta$'s were the fitted regression coefficients.

Predicted Q, which represents Q in the absence of disturbance ($\hat{Q}$) during the post-disturbance period ($t = 2$), was predicted using the calibration model:

$$\hat{Q}_{Trt,2} = \beta_0 + \beta_1 Q_{Ref,2} \tag{8}$$

The effects of forest harvesting on Q were then estimated by calculating the residual differences between observed and reconstructed Q:

$$\Delta Q_{Veg} = \overline{Q_{Trt,t}} - \overline{\hat{Q}_{Trt,t}} \tag{9}$$

Forest harvesting effects on Q were considered significant at the $\alpha = 0.05$ level when $> 5\%$ of the post-disturbance residuals exceeded the 95th percentile prediction limits (Harr et al., 1979; Zégre et al., 2010).

While the effects of climate change on Q ordinarily are not quantified using paired catchment models (e.g., Schade and Shuster, 2005), it can be estimated by calculating the difference between reconstructed streamflow predicted from paired catchment models between the two periods:

$$\Delta Q_{Cpaired} = \hat{Q}_{t2} - \hat{Q}_{t1} \tag{10}$$

*2.4.3. Evaluating the efficacy of applying Budyko decomposition to the headwater-scale*

To assess the efficacy of applying Budyko decomposition to the headwater-scale, we first compared estimates of ΔQ from decomposition to estimates from paired catchment models. Second, we validated the directional change assumption that provides the basis of separating and quantifying the contributions of harvesting and climate change to ΔQ. Our validation approach was based on reconstructed E ($E_{Recon}$). In order to represent hydrologic conditions in the absence of disturbance, $E_{Recon}$ was calculated using the water balance by subtracting $\hat{Q}$ calculated from paired catchment models, from P. New baseline Budyko curves were calculated for each catchment using $E_{Recon}$ in place of observed E in Eq.(1) to determine the coordinate positions of $\phi$ in the absence of forest harvesting.

Validation of directional assumptions was conducted by plotting the coordinate positions of $\phi_1$ and $\phi_2$ that included the effects of disturbance, and $\phi_{1_{Recon}}$ and $\phi_{2_{Recon}}$, which had the effects of disturbance removed.The vertical change assumption for harvesting was validated using the first period. Because $\phi_{1_{Recon}}$ and $\phi_1$ shared the same time period and climate (Ep/P), the effects of harvesting can be quantified as the vertical distance between $\phi_{1_{Recon}}$ and $\phi_1$ (Fig. 2). The horizontal change assumption for climate change was validated by calculating the horizontal distance between $\phi_{1_{Recon}}$ and$\phi_{2_{Recon}}$. After removing the effects of harvesting, movement between $\phi_{1_{Recon}}$ and $\phi_{2_{Recon}}$ would be expected to be along the curve (Fig. 2).

The relative performance of decomposition was conducted by comparing ΔQ estimates to paired catchment models for three cases:

*Case 1. Comparison of $\Delta Q_H$ and $\Delta Q_C$ contributions between two periods based on paired catchment models.* ΔQ attributed to both harvesting and climate change between the two periods based on Budyko decomposition was compared to $\Delta Q_H$ and $\Delta Q_C$ based on the paired catchment approach.

*Case 2. Comparison of $\Delta Q_H$ in Period 1.* Because $\phi_{1,Recon}$ and $\phi_1$ share a similar time period and climate (Ep/P), ΔQ attributed to harvesting using Budyko ($\Delta Q_{H_1(Budyko)}$) was calculated following Eq.(5), but using data for only the first period:

$$\Delta Q_{H_1(Budyko)} = P_1\left(\frac{E_{Recon,1}}{P_1} - \frac{E_1}{P_1}\right) \tag{11}$$

where $P_1$ is precipitation for Period 1; $\frac{E_{Recon,1}}{P_1}$ is the reconstructed evaporation ratio, and $\frac{E_1}{P_1}$ is the observed evaporation ratio for Period 1 that includes the effects of disturbance. This was compared to ΔQ estimated from the paired catchment models ($\Delta Q_{H_1(paired)}$) using observed and reconstructed streamflow for Period 1.

*Case 3. Comparison of $\Delta Q_C$ in Period 2.* ΔQ attributed to climate change in the second period was estimated using Budyko ($\Delta Q C(Budyko)$) by substituting $E_{Recon,1}$ into Eq.(10) to remove the effects of disturbance, and solving Eqs.(8) and(10). This was compared to $\Delta Q_{C(paired)}$) estimated from the paired catchment models (Eq.(4)) using reconstructed Q for Period 1 and Period 2.

## 3. Results

### *3.1. Changes in forest species composition*

Thirty-seven species of mostly hardwood species of trees were identified during stand inventories conducted between 1957 and 2009 (Supplementary Table 1). Oak species dominated basal area in all of the catchments during initial inventories conducted between 1957 and 1963 (Fig. 3). During the most recent inventories, sugar maple, yellow-poplar, and red maple increased in basal area in watershed 1; red maple increased in watershed 2; oaks, black cherry, and sugar maple increased in watershed 4; Norway spruce increased in watershed 6; and yellow-poplar, black cherry, and sweet birch increased in watershed 7. Disturbed catchments that remained as mixed hardwoods (watersheds 1, 2, and 7) were dominated in the most recent inventories by species with diffuse-porous xylem anatomy, accounting for 56 to 95% of relative basal areas (Fig. 3). Species with ring-porous xylem anatomy dominated the reference catchment, accounting for 51% of basal area in the most recent inventory. With the conversion of watershed 6 to Norway spruce following clearcutting and herbicide application to suppress competition from native hardwoods, tracheid xylem anatomy accounted for 100% of catchment basal area. Species with semi-ring porous xylem anatomy decreased across all catchments (Fig. 3).

### *3.2. Changes in long-term climate and water balance components*

Relative changes in water balance components between the two periods varied across catchments (Table 2). ΔP ranged from 0.2 to 4 %; ΔQ ranged from −29 to 12 %; and ΔE ranged from −7 to 26 %. Trend analysis conducted on the long-term annual time series (Supplementary Figures 1 and 2) indicated that the climate at the Fernow Experiential Forest has changed significantly over the sixty year study period (Supplementary Figs. 1 and 2). Average annual air temperature increased significantly (p-value = 0.04) by 0.01 °C $yr^{-1}$ or by 0.6 °C over the study period. Minimum annual air temperature increased significantly (p-value = 0.07) by 0.02 °C $yr^{-1}$. Changes in maximum annual air temperature were not significant (p-value = 0.7). Despite what appeared to be increasing trends in mean annual precipitation in all the catchments except for watershed 4, trends were not significant (p-value = 0.24 –0.99). Trends in E were significant at $\alpha = 0.05$ for watersheds 2 (p-value < 0.001), 6 (p-value < 0.0001), and 7 (p-value < 0.001) and at $\alpha = 0.10$ for watersheds 1 (p-value = 0.07) and 4 (p-value = 0.08). Trends in

Q were significant only in watersheds 2 (p-value < 0.001) and 6 (p-value = 0.001).

### 3.3. Effects of forest harvesting and climate change on ΔQ between two periods

#### 3.3.1. Budyko decomposition

Long-term mean evaporation ratios, E/P, and *w*-parameters during the first period varied widely across catchments likely due to variations in initial forest composition and experimental harvest treatments. During the first period E/P ranged between 0.37 (standard deviation = ±0.08) and 0.60 (±0.05) while during the second period E/P ranged between 0.43 (±0.08) and 0.69 (±0.08). Variation in Ep/P were smaller across catchments and periods (Fig. 2; Table 2) than E/P indicating that catchments share a similar climate characterized as energy-limited. This was expected given the close geographic proximity of catchments. Meanwhile *w*-parameter values ranged from 1.53 to 2.93 during the first period and 1.75 and 5.04 during the second period (Table 2).

From Period 1 to Period 2, $\phi$ moved left and away from baseline Budyko curves in the treated catchments, and right and away from baseline Budyko curve in the reference catchment (Fig. 4a). $\Delta Q_{total}$ between the two periods ranged from −29 to 15 %(Fig. 4b; Tables 2 and 3). ΔQ attributed to climate change ($\Delta Q_{C(Budyko)}$) ranged between −2 and 6% across all catchments while contributions of forest harvesting ($\Delta Q_{H(Budyko)}$) ranged between −32 and 11 %. Greater variation in ($\Delta Q_{H(Budyko)}$) reflected the different type, frequency, and intensity of treatments across catchments (Table 1). In all catchments except for watershed 2, $\Delta Q_{total}$ were similar between water balance differences (Table 2) and decomposition. Estimates of $\Delta Q_{total}$ based on water balance differences between the two periods was 12 %(Table 2) while $\Delta Q_{total}$ based on Budyko was equal to 15 %(Table 3a).

#### 3.3.2. Paired catchment approach

Paired catchment models indicated strong Q relationships the reference and treatment catchments (Table 1). p-Values for slope coefficients ranged between <0.00001 and 0.002 while coefficient

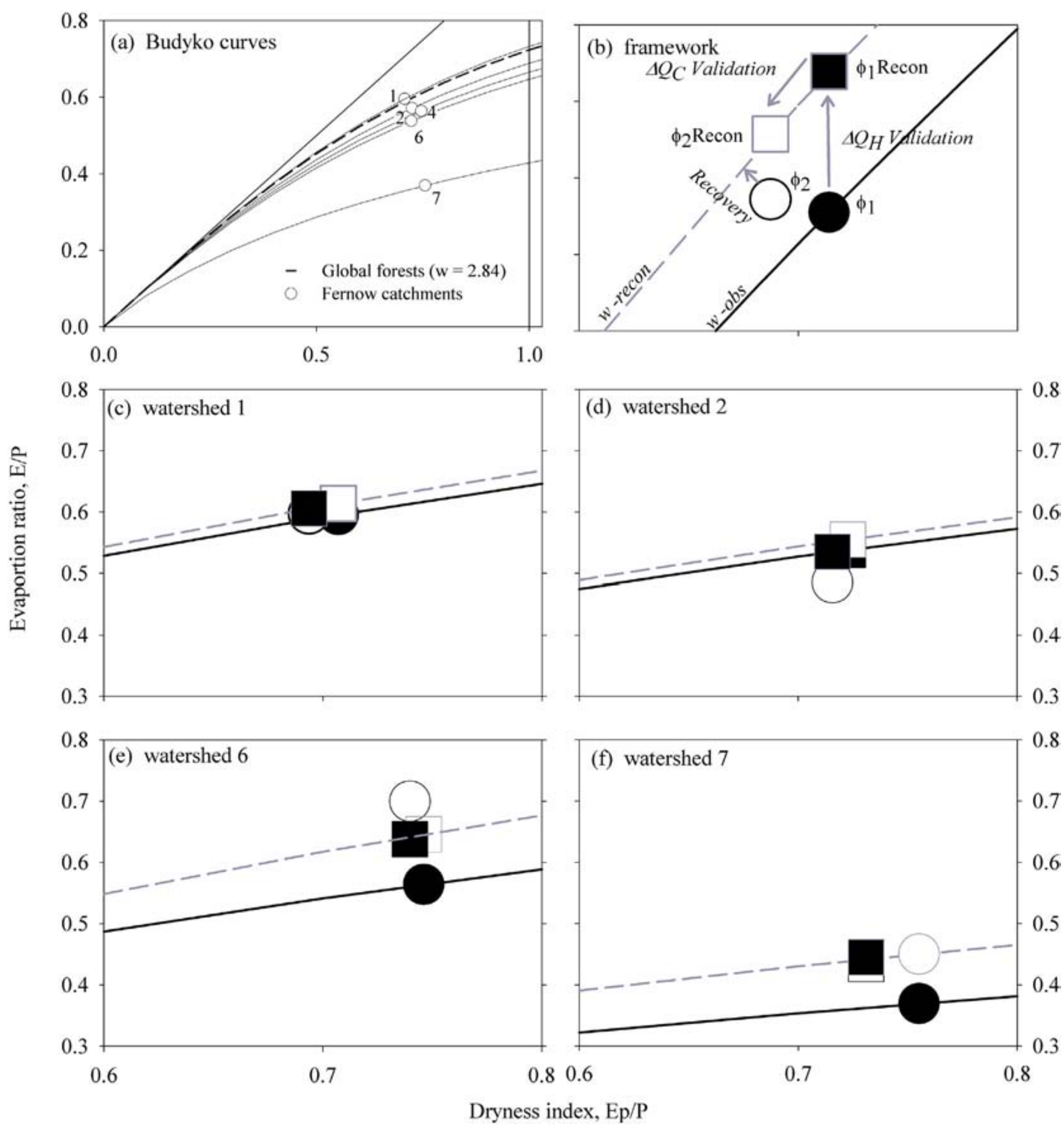


**Fig. 2.** (a) Budyko curves of headwater catchments of the USDA Fernow Experimental Forest, West Virginia and global forests following Zhang et al. (2004) that show the relationship between evaporation ratio (E/P) and dryness index (Ep/P) for Period 1. All catchments, denoted by open circles, were energy-limited. *w*-parameters values ranged from 1.53 to 2.93. (b) Framework for assessing directional change assumptions of Budyko decomposition. (c–f) Results for each catchment.

of determination ($R^2$) ranged from 0.87 to >0.99. Standard error of estimated Q ranged from 9.3 to 24.0 mm.

ΔQ attributed to forest harvesting based on the paired catchment approach ($\Delta Q_{H(paired)}$) between the two periods ranged from −33 to 7 %(Table 3a), reflecting the different type, intensity, and recurrence of forest management disturbances across catchments (Table 1, Fig. 5). ΔQ attributed to climate change between the two periods based on this approach ($\Delta Q_{C(paired)}$) increased streamflow between 4 and 5%.

### 3.4. Validation of Budyko decomposition applied to the headwater-scale

#### 3.4.1. Evaluating the directional change assumptions of Budyko decomposition

E/P based on reconstructed E ($E_{Recon}$) ranged from 0.45 to 0.65 for Periods 1 and 2, while values for reconstructed $w$ ranged from 1.78 to 3.34 for the first period and 1.82 to 3.27 for the second period (Table 2). The larger evaporation ratios indicate that, in the absence of forest disturbance, a larger proportion of P would have been partitioned to E rather than Q.

The assumption that harvesting would result in vertical movement away from baseline curves was assessed by comparing the coordinate location of $\phi_1$ that included harvesting effects, to the coordinate location of $\phi_{1,Recon}$ that removed the effects of disturbance. In all catchments, $\phi_1$ moved away from baseline curves, providing evidence that the vertical change assumption was valid in our study (Fig. 2 c–f). The assumption that climate change would produce movement along the curve was assessed by first removing the effects of disturbance in Periods 1 and 2, and comparing the coordinate locations of $\phi_{1,Recon}$ and $\phi_{2,Recon}$. In all catchments, $\phi$ moved horizontally along the reconstructed baseline Budyko curves, providing evidence that the climate change assumption was also valid in our study (Fig. 2 c–f).

#### 3.4.2. Case 1. Comparison of two-period $\Delta Q_H$ and $\Delta Q_C$ based on Budyko decomposition and paired catchment approach

The combined effects of harvesting and climate change on ΔQ ranged between −29 and 15% based on Budyko and −28 to 12% based on the paired catchment approach (Table 3a). The contribution of harvesting were similar for each approach (Table 3a), averaging −9%, although $\Delta Q_H$ based on Budyko were generally greater. $\Delta Q_C$ based on the paired catchment approach averaged 5% compared to 4% for Budyko (Table 3a).

#### 3.4.3. Case 2. $\Delta Q_H$ in Period 1

Based on paired catchment analysis, forest harvesting significantly increased Q in all the catchments during the first period, with between 12 and 90% of the post-disturbance residuals exceeding the 95th percentile prediction limits (Table 3b). The effects of forest harvesting on ΔQ based on decomposition ($\Delta Q_{H1(Budyko)}$ and paired catchment ($\Delta Q_{H1(paired)}$) approaches were identical for watersheds 1 and 2 where Q increased by 5% following harvesting. Harvesting effects were nearly identical for both approaches for the other two catchments. Streamflow increased by 22% and 23% in watershed 6, and by 14% and 15% in watershed 7, using the paired catchment and decomposition approaches, respectively (Table 3b). In each case, $\Delta Q_{H1(Budyko)}$ were contained within the 95% confidence limits of paired catchment models, indicating that decomposition performed equally to the paired catchment approach for estimating harvesting effects on ΔQ in each catchment.

#### 3.4.4. Case 3. Comparison of $\Delta Q_C$ in Period 2

The effects of climate change on ΔQ in the second period based on both approaches were also similar, with Q increasing between 2 and 5% across the treatment catchments (Table 3c). $\Delta Q_{C2(paired)}$ were generally larger than $\Delta Q_{C2(Budyko)}$ except for watershed 7, where the opposite was true.

## 4. Discussion

### 4.1. Drivers of streamflow change: climate, forest harvesting, and forest structure

The horizontal and vertical movement of $\phi$ away from baseline Budyko curves between the first and second periods indicated that

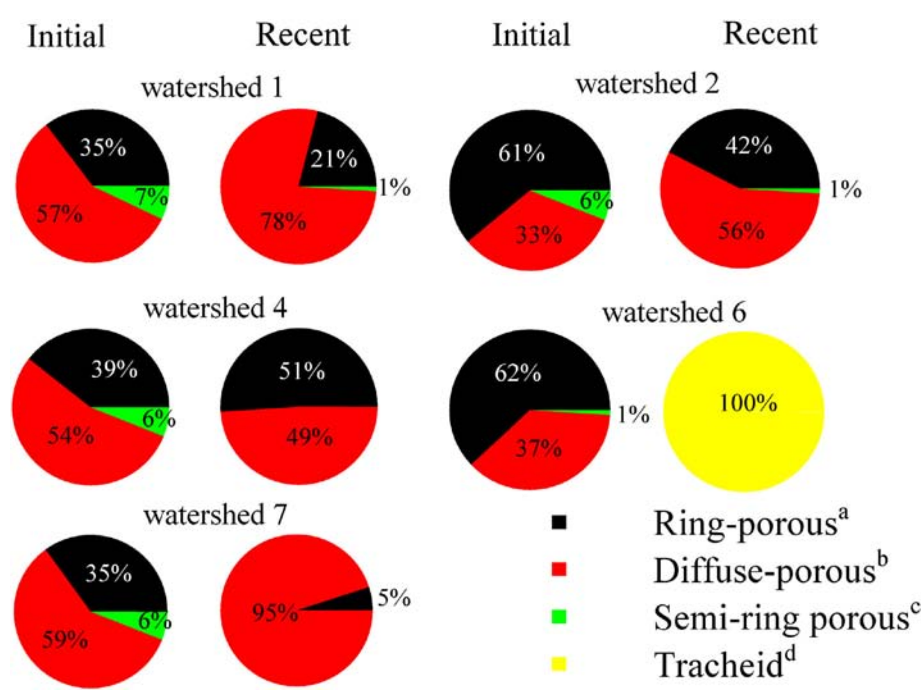


**Fig. 3.** Proportion of catchment basal area classified by xylem anatomy (ring-porous, diffuse-porous, semi-ring porous, and tracheid) for five experimental catchments at the Fernow Experimental Forest. The initial period represents first inventories conducted between 1957 and 1963 while the recent period represents the most recent inventories conducted between 1997 and 2009.

**Table 2**
Observed and reconstructed water balance and Budyko components for the five headwater experimental catchments across two periods. P = precipitation; Q = streamflow; E = evapotranspiration; E/P = evaporation ratio; Ep/P = dryness index; $w$ = landscape parameter.

| Watershed | Period | Years | Water balance components | | | Budyko components | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | P (mm) | Q (mm) | E (mm) | E/P (sd) | Recon. E/P | EP/P (sd) | w | Recon. w |
| 1 | 1 | 1952–1981 | 1484 | 606 | 878 | 0.60 (0.05) | 0.61 | 0.71 (0.09) | 2.93 | 3.20 |
| | 2 | 1982–2012 | 1529 | 622 | 906 | 0.60 (0.05) | 0.61 | 0.69 (0.09) | 3.06 | 3.20 |
| | | Δtotal | 45 | 16 | 28 | – | – | – | – | – |
| | | %Δ | 3 | 2.7 | 3 | – | – | – | – | – |
| 2 | 1 | 1952–1981 | 1452 | 691 | 761 | 0.53 (0.06) | 0.55 | 0.72 (0.09) | 2.23 | 2.39 |
| | 2 | 1982–2012 | 1483 | 777 | 705 | 0.48 (0.05) | 0.54 | 0.72 (0.10) | 2.00 | 2.30 |
| | | Δtotal | 31 | 86 | −56 | – | – | – | – | – |
| | | %Δ | 2 | 12 | −7 | – | – | – | – | – |
| 4 | 1 | 1952–1981 | 1449 | 626 | 823 | 0.57 (0.08) | – | 0.72 (0.09) | 2.56 | – |
| | 2 | 1982–2012 | 1451 | 662 | 789 | 0.54 (0.07) | – | 0.73 (0.10) | 3.06 | – |
| | | Δtotal | 3 | 36 | −34 | – | – | – | – | – |
| | | %Δ | 0.2 | 6 | −4 | – | – | – | – | – |
| 6 | 1 | 1957–1985 | 1411 | 618 | 793 | 0.56 (0.07) | 0.65 | 0.75 (0.09) | 2.40 | 3.34 |
| | 2 | 1986–2012 | 1438 | 440 | 998 | 0.69 (0.08) | 0.64 | 0.74 (0.11) | 5.04 | 3.27 |
| | | Δ total | 27 | −178 | 205 | – | – | – | – | – |
| | | %Δ | 2 | −29 | 26 | – | – | – | – | – |
| 7 | 1 | 1957–1985 | 1398 | 883 | 515 | 0.37 (0.08) | 0.45 | 0.76 (0.10) | 1.53 | 1.78 |
| | 2 | 1986–2012 | 1454 | 831 | 623 | 0.43 (0.06) | 0.45 | 0.73 (0.10) | 1.75 | 1.82 |
| | | Δ total | 56 | −52 | 108 | – | – | – | – | – |
| | | %Δ | 4 | −6 | 21 | – | – | – | – | – |

Q at the Fernow has responded to both climate change and forest harvesting. Changes in climate at the Fernow are consistent with other larger regional studies (Hayhoe et al., 2007; Pitchford et al., 2011; Fernandez and Zegre, 2019) and similar to other small catchment studies located throughout the eastern USA. The 0.01 °C yr$^{-1}$ warming trend places the Fernow just below temperature changes at other mountain sites such as Coweeta to the south (western North Carolina), and Hubbard Brook to the north (New Hampshire), where climate has warmed between 0.017 and 0.029 °C yr$^{-1}$ ((Campbell et al., 2011); Ford et al., 2011b) Laseter et al., 2012. While the increasing trends in P at the Fernow were not significant, the average increase of 1.03 mm yr$^{-1}$ falls within the observed range of P changes at Coweeta (−2.38 to 2.24 mm yr$^{-1}$) and Hubbard Brook (~1 to 3.5 mm yr$^{-1}$) (Campbell et al., 2011; Ford et al., 2011b; Laseter et al., 2012).

The leftward movement of $\phi$ along Budyko curves in the treatment catchments indicated a general shift in climate towards a decreasing E/P. In this case, $\Delta Q_C$ was positive since a decrease in the E/P translates to an increase in Q that averaged 4.3% across catchments. The rightward movement of $\phi$ in the reference catchment indicates a shift towards an increasing E/P, which has also been shown throughout the region (Fernandez and Zegre, 2019). In this case, $\Delta Q_C$ was negative since an increase in Ep/P translated to an 2% decrease in Q.

$\Delta Q_C$ at the Fernow were similar, albeit smaller, to $\Delta Q_C$ reported for other regions in the USA. In the water-limited southwest and Central Plains regions, for example, climate change increased Q by 21 to >100 %, whereas in the energy-limited southeastern and mid-Atlantic region, climate change increased Q by 14 and 35 %, respectively (Wang and Hejazi, 2011; Patterson et al., 2013. The smaller $\Delta Q_C$ in our study were attributed to the lower rate of warming in the higher elevations of the central Appalachian Mountains compared to lower lying regions of the mid-Atlantic and the southeastern USA where over the late 20th century climate has warmed by 0.74 °C yr$^{-1}$ (Patterson et al., 2012, 2013).

(Wang and Hejazi (2011) and Patterson et al. (2013)) found that climate change dominated changes in Q throughout the USA, whereas human activities in the form of forest harvesting dominated changes at the Fernow. The dominance of climate change in these other studies was not entirely unexpected since they focused primarily on relatively undisturbed meso-scale ($10^2$–$10^4$ km$^2$) catchments where the proportion of area disturbed to total catchment area was relatively small (Wang and Hejazi, 2011; Patterson et al., 2013). Harvesting in the Fernow catchments ranged between 30 and 100 %, exceeding the 15 to 20% disturbance threshold typically expected for detectable changes in Q due to harvesting (Bosch and Hewlett, 1982; Stednick, 1996). Because vegetation co-evolves with climate and landscape over time (Rodriguez-Iturbe, 2000; Rodriguez-Iturbe and Porporato, 2004), forest ecosystems can adjust resource allocation to adapt to the new climate conditions (Jones et al., 2012). Harvesting, on the other hand, abruptly disturbs canopy interception and transpiration and over the long term, influences the water balance by altering post-harvest forest structure and species composition. With the gradual warming trend, lack of significant changes in precipitation, and the comparatively large changes in forest cover, it stands to reason that ΔQ was more sensitive to harvesting than to climate change over different time scales.

Over the short term, harvesting increased Q in the Fernow catchments by decreasing E and canopy interception(Hornbeck et al., 1993). The magnitude of ΔQ were commensurate with the extent and frequency of disturbance (Table 1). Over the long term, ΔQ varied by catchment (Fig. 4b) likely reflecting both the gradient of disturbances across catchments and the resultant disturbance-driven shifts in forest composition. Stand development largely has favored diffuse porous in watersheds 1, 2, and 7 and tracheid xylem species in watershed 6 over the ring porous species that occupied larger proportions of basal areas early in the study (Fig. 3; Supplementary Table 1). Diffuse porous and tracheid species generally use more water than ring and semi-ring porous species (Wullschleger et al., 2001;

Wullschleger et al., 2001; Stoy et al., 2006; Ford et al., 2011a,b; Meinzer et al., 2013) so changes in Q could be attributed to changes in functional xylem structure due to the shifts in species composition (Fig. 3). Forest mesophication, the shift of forest towards mesophytic species, is well documented throughout the eastern United States (Nowacki and Abrams, 2008; Schuler et al., 2016) and has been linked to catchment-scale changes in E and Q at the Coweeta Hydrological Laboratory (Caldwell et al., 2016). At the Fernow, the decreases Q in watershed 6 have been attributed to forest conversion from hardwood species to Norway spruce (Adams et al., 2012; Hornbeck et al., 1993) that have accounted for 100% of basal area since 1983 (Supplementary Table 1). The increases in diffuse porous species basal areas are also commensurate with Q decreases in watersheds 1 and 7. However, despite similar basal area increases in watershed 2, Q has increased. In this case, the periodic removal of basal area due to repeated harvests likely played a larger role driving ΔQ than changes in water use due to species shifts (Fig. 3; Supplementary Table 1). Taken collectively, these results show that hydrological effects of harvesting at the Fernow are persistent despite three to five decades of regrowth thereby updating previous conclusions that the hydrological impacts of harvesting are relatively short-lived (Hornbeck et al., 1993).

Another possible explanation for the long-term water balance changes across the Fernow could be climate-mediated changes in growing season length and forest productivity. While the experimental approach and data used in this study cannot disentangle the effects of climate change on these factors, recent studies speak to the potential (Alfieri et al., 2015;Hwang et al.;Gaertner et al., 2019). In the forests of the central Appalachian Mountains region, growing season has increased on average by 22 days and E changed between −2 and 30 mm over the previous three decades (Gaertner et al., 2019). The lengthening growing season was attributed to a suite of climatic variables that included air temperature, vapor pressure deficit, wind, and humidity. Clearly further work is necessary to better understand the interaction between forest ecosystem, climate, and water balance dynamics at the headwater scale. Despite limited inference, our study contributes to a more thorough understanding about the impacts of climate change on disturbed headwater catchments that will help managers to better prepare for and adapt to future changes.

### 4.2. Not all disturbances are equal when it comes to climate change

Decomposition analysis showed that harvesting largely masked the effects of climate change-driven ΔQ (Fig. 4c), which has been discussed elsewhere (Patterson et al., 2013;Jones et al., 2012). However, our results also showed that not all disturbances are equal when it comes to streamflow sensitivity to climate change. Disturbance masked the effects of climate change in catchments that had low frequency/high magnitude disturbances (watersheds 1, 6, 7), or no recent disturbances, i.e., watershed 4. On the other hand, climate change amplified ΔQ in watershed 2. In this catchment, disturbances occurred relatively frequently, occurring on average every nine years. However, disturbances were generally smaller in magnitude than treatments in the other catchments. Furthermore, watershed 2 had an equal number of disturbance treatments in the second period as the first, whereas disturbances in the other treated catchments were focused during the first period (Table 1).

Interestingly, Q in watershed 2 was the least sensitive to changes in climate of the treatment catchments, despite the climate-driven amplification of Q (Fig. 4c). The lower sensitivity of Q to climate change in this catchment is thought to be due to disturbance-driven forest succession that, until recently, and for the majority of period 2, favored ring porous oaks. Across the Fernow catchments that we studied, Q sensitivity to climate change increased with the increase of basal area for diffuse porous species (Pearson's R = 0.75, p-value = 0.1).

Our catchment-scale results are corroborated by studies at the tree-scale which show diffuse porous species are more sensitive to variations in climate with respect to water and carbon dynamics (Ford et al., 2011b;Brzostek et al., 2014;Caldwell et al., 2016). These traits, in addition to geographic differences of water and energy availability and legacy disturbances, are important factors for catchment elasticity and adaptation of rainfall partitioning to climate change (Jones et al., 2012;Creed et al., 2014).

### 4.3. Can Budyko decomposition be applied at the headwater scale?

Our validation exercises generally support the efficacy of applying Budyko decomposition at the headwater-scale in the Fernow catchments. Notwithstanding, what was classified as $\Delta Q_H$ in the reference catchment was unexpected, given the lack of human activities in the catchment, and points to limitations of the approach. Budyko decomposition is predicated on the assumption that ΔQ are driven only by changes in climate (ΔP and ΔEp) and human activities. Despite the inclusion of the landscape parameter, *w*, decomposition does not consider changes in *w* as a driver of ΔQ. In this sense, decomposition is analogous to two-component hydrograph separation approaches that separate storm flow into only pre event- and event-water components (McDonnell et al., 1990), despite cognizance that runoff generation is more complex than just these two components (Kendall and McDonnell, 1998). The simplified analytical structure of decomposition may also explain the differences in $\Delta Q_{total}$ calculated between the water balance changes between the two periods (Table 2) and Budyko decomposition between the two periods (Table 3a). While absolute and relative $\Delta Q_{total}$ were similar between approaches in watersheds 1, 4, 6, and 7 after considering round-off error, $\Delta Q_{total}$ differed by 3% in watershed 2. The reason for this was not clear but it could be due to either model error associated with reconstruction of the missing streamflow data using the HBV model or to the relatively frequent harvesting treatments in watershed 2 that occurred throughout the study period. Given that $\Delta Q_C$ estimates are similar across catchments (Table 3a), the larger $\Delta Q_{total}$ associated with decomposition could be due to inadequacy of decomposition for capturing frequent disturbances in the estimation of $\Delta Q_H$. Notwithstanding, more research is necessary to understand what is mechanistically driving these differences.

Hydrological changes in the reference catchment could be due to a myriad of factors that are unaccounted for in decomposition, and other Budyko-based approaches (e.g. Roderick and Farquhar (2011)). These include changes in forest structure due to past natural and anthropogenic disturbance (Jones, 2011; Jones et al., 2012), changes in forest productivity, elevated atmospheric $CO_2$ concentrations, and acidic deposition. Forest harvesting early in the 20th century (Trimble, 1977) and chestnut blight in the 1930s (Weitzman, 1949), for example, significantly altered forest succession and species composition. By the start of the Fernow watershed study in the late 1950s, diffuse porous species accounted for 54% of the basal area, while ring porous oaks accounted for 34 %(Fig. 3; Supplementary Table 1). Overtime, stand development has favored oaks, with basal area increasing to 49% of catchment area by 2009, while diffuse porous basal areas has remained relatively constant. Over the same period, forest productivity also has decreased in the >100-year-old mixed hardwood forest. Forest productivity, represented in terms of net periodic annual increment, has decreased from around $4\,m^3\,ha^{-1}\,yr^{-1}$ in 1950 to around $1\,m^3\,ha^{-1}\,yr^{-1}$ in 2010 (Schuler et al., 2016).

In a similarly aged hardwood reference catchment at Hubbard Brook Experimental Forest, long-term declines in E were thought to be related to declines in productivity, accelerated tree mortality, and biomass decreases of other classes of vegetation (e.g.,

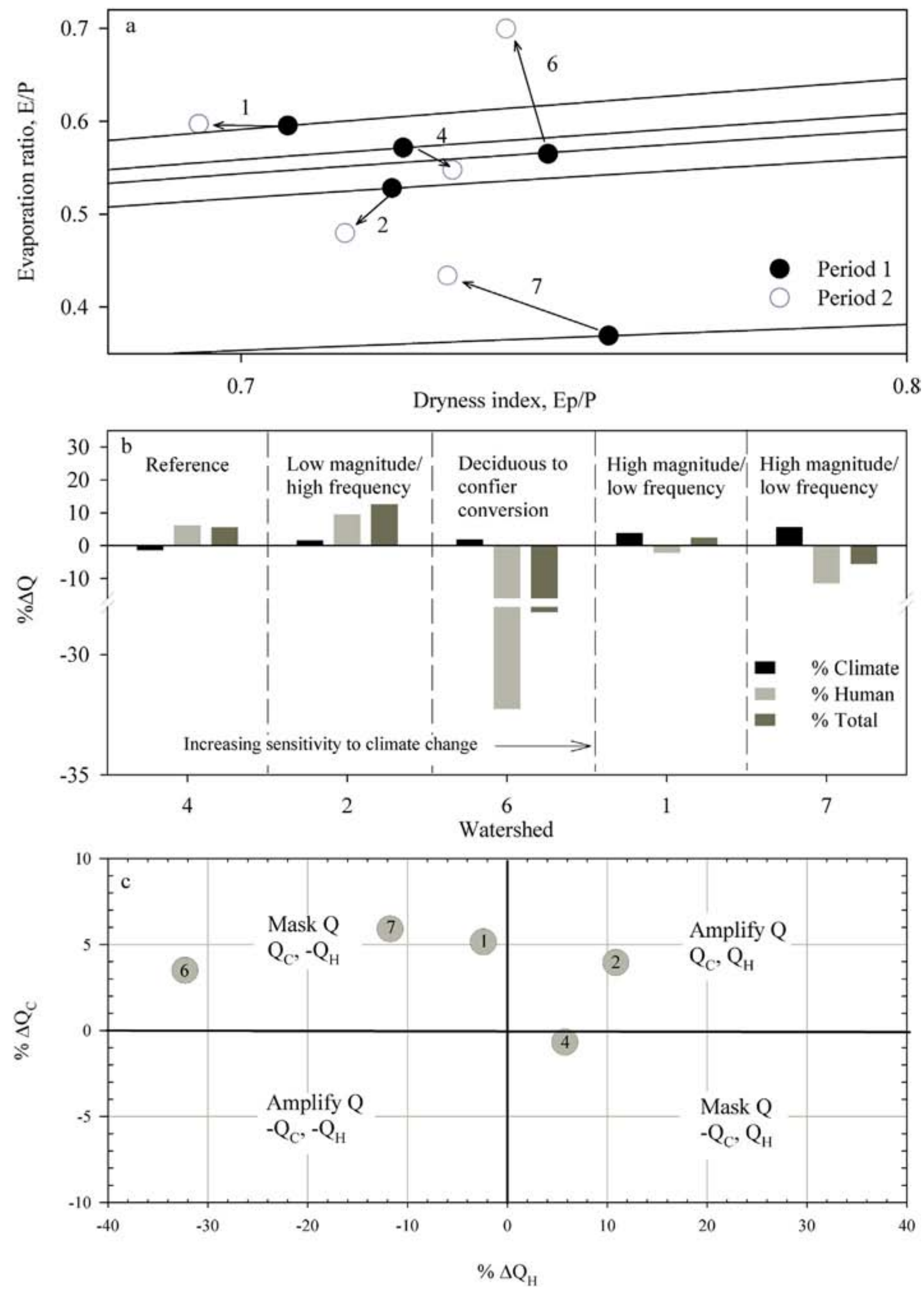


**Fig. 4.** (a) Two-period Budyko decomposition results for the five headwater catchments at the Fernow Experimental Forest. (a) Horizontal and vertical changes of $\phi$ from Period 1 to Period 2 for each catchment denoted by watershed number. (b) Changes in streamflow attributed to climate change and human activities, with catchments ranked by increasing sensitivity to climate change. (c) Streamflow changes attributed to climate change ($\Delta Q_C$) and forest harvesting ($\Delta Q_H$)for headwater catchments at the Fernow. Streamflow changes due to climate change were masked/off-set in five catchments that had low frequency/high magnitude disturbances. Climate change effects were amplified in one catchment that had high frequency/low magnitude disturbances.

seedlings, herbs) (Campbell et al., 2011). Alternatively, increases in atmospheric $CO_2$ concentrations can decrease stomatal conductance, increase water use efficiency (WUE), and decrease E (Ehleringer and Cerling, 1995;Beerling, 1996;Herrick et al., 2004). While the association between $CO_2$ and WUE has been shown at the continental (Betts et al., 2007; Gedney et al., 2006) and stand scales (Warren et al., 2011;Leuzinger and Korner, 2010) and is consistent with a roughly increasing WUE for forests worldwide (Peñuelas et al., 2011;van der Sleen et al., 2015), Huntington (2008) argued that continental-scale increases in Q is the result of water cycle intensification and not increasing $CO_2$ concentrations. Increases in Q and decreases in E in our reference catchment could also be related to acid deposition from the combustion of fossil fuels. Thomas et al. (2013) found that acid deposition in the region reduced stomatal conductance and WUE of eastern red cedar (*Juniperus virginiana L.*). Following enactment of the Clean Air Act of 1970, however, stomatal conductance and WUE recovered. Given the monotonic changes in water balance components over time, and no evidence of reversal at the Fernow, it is unlikely that the changes were related to acidic deposition.

The results from our headwater-scale application of Budyko corroborate findings from recent global assessments that were conducted at larger spatial scales. Global studies show that hydrological change is more complex than just corresponding changes in P, Ep, and landscape factors (Gudmundsson et al., 2016;Berghuijs et al., 2017;Padrón et al., 2017). Budyko-based approaches for quantifying and attributing hydrological change are overly simplified, neglecting

**Table 3**
The efficacy of applying Budyko decomposition to the headwater scale was assessed using the validation framework shown in Fig. 3 for three cases: a) Comparison of $\Delta Q_H$ and $\Delta Q_C$ between two periods; b) Comparison of $\Delta Q_H$ during Period 1; and c) Comparison of $\Delta Q_C$ during Period 2. Evaluation of directional change assumptions for decomposition and validation were based on using reconstructed streamflow and evapotranspiration to represent conditions in the absence of forest harvesting.

*a) Case 1: comparison of QH and QC contributions to Q between two periods*

| Watershed | Budyko decomposition | | | | | Paired catchment | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | $\Delta Q_{H(Budyko)}$ mm | (Eqs. (3), (5)) % Δ | $\Delta Q_{C(Budyko)}$ mm | (Eqs. (4), (6)) % Δ | $\Delta Q_{tot(Budyko)}$ % Δ | $\Delta Q_{H(paired)}$ mm | (Eq. (9)) % Δ | $\Delta Q_{C(paired)}$ mm | (Eq. (10)) % Δ | $\Delta Q_{tot(paired)}$ % Δ |
| 1 | −15 | −2 | 31 | 4 | 2 | −15 | −2 | 30 | 5 | 3 |
| 2 | 67 | 11 | 27 | 4 | 15 | 49 | 7 | 45 | 5 | 12 |
| 4 | 40 | 6 | −4 | −2 | 5 | –[a] | –[a] | – | – | – |
| 6 | −200 | −32 | 22 | 4 | −29 | −209 | −33 | 5 | 5 | −28 |
| 7 | −104 | −12 | 52 | 6 | −6 | −83 | −9 | 27 | 4 | −5 |

*b) Case 2: comparison of QH in Period 1*

| Watershed | Budyko decomposition | | Paired catchment | | |
|---|---|---|---|---|---|
| | $\Delta Q_{H1(Budyko)}$ | (Eqs. (3), (5)) | | $\Delta Q_{H1(paired)}$ | (Eq. (9)) |
| | mm | %Δ | Proportion of residuals >95% PL[a] | ΔQ mm (95% CL)[b] | % Δ |
| 1 | 27 | 5 | 12 | 27 (17,37) | 5 |
| 2 | 33 | 5 | 71 | 33 (25,41) | 5 |
| 6 | 115 | 23 | 90 | 114 (90,138) | 22 |
| 7 | 113 | 15 | 86 | 107 (82,133) | 14 |

*c) Case 3: comparison of QC in Period 2*

| Watershed | Budyko decomposition | | Paired catchment | |
|---|---|---|---|---|
| | $\Delta Q_{C(Budyko)}$[c] mm | (Eqs. (4), (6)) %Δ | $\Delta Q_{C(paired)}$ mm | (Eq. (10)) %Δ |
| 1 | 24 | 4 | 30 | 5 |
| 2 | 13 | 3 | 33 | 5 |
| 6 | 10 | 2 | 31 | 5 |
| 7 | 38 | 5 | 27 | 4 |

[a] Significant ($\alpha = 0.05$) when % post-disturbance residuals >95% prediction limits (PL).
[b] Streamflow changes, (95% confidence limits around mean change).
[c] $\Delta Q_C$based on reconstructed Q to remove disturbance effects.

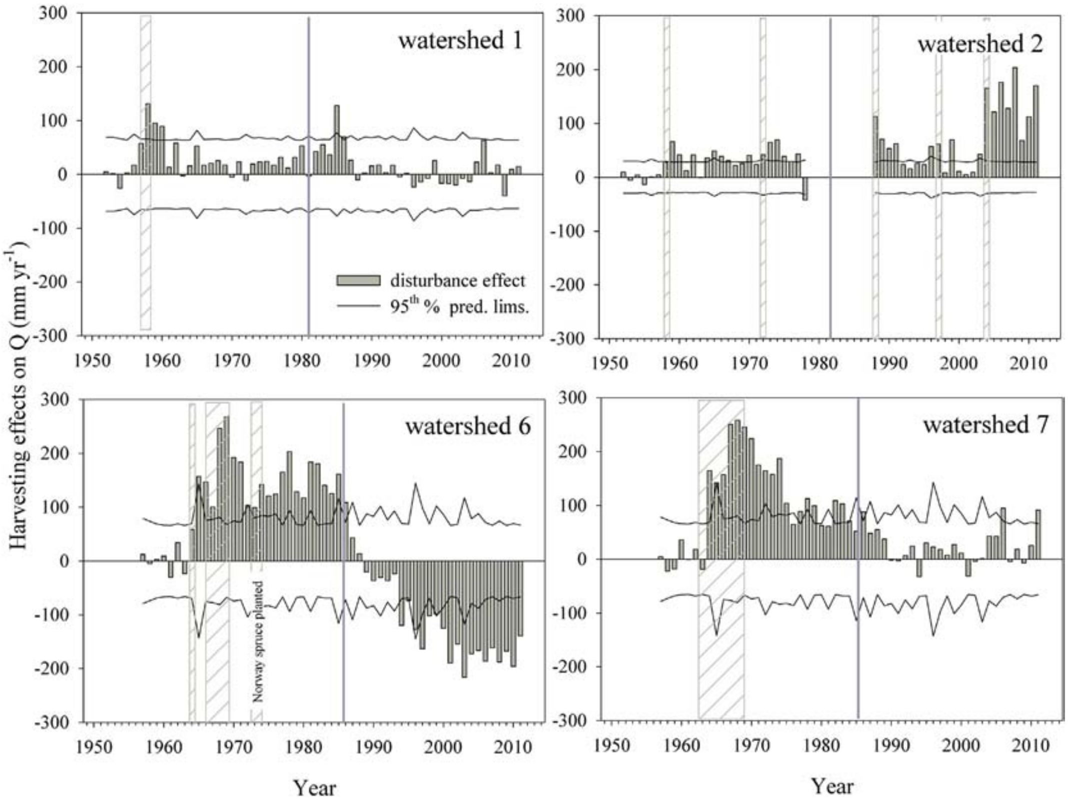


**Fig. 5.** Forest harvesting effects on annual streamflow and 95th percentile prediction limits based on paired catchment analysis. Harvesting effects were quantified as the residual difference between observed and reconstructed streamflow based on pre-disturbance calibration models between treated catchments and the reference catchment. Hashed areas represent periods of experimental disturbance treatments in each catchment (see Table 1 for treatment descriptions). The solid vertical lines separate Period 1 from Period 2 in accordance with Budyko decomposition.

other important factors such as storminess (Milly, 1994), snow dynamics (Berghuijs et al., 2014), seasonality (Petersen et al., 2012), slope (Padrón et al., 2017), and vegetation response to legacy and contemporary disturbances (Jones, 2011; Jones et al., 2012). This is especially true for Budyko decomposition that quantifies $\Delta Q_H$ and $\Delta Q_C$ using the climate change component of the evaporation ratio, E/P (Eq. (2)). And while other Budyko-based approaches (e.g., Roderick and Farquhar, 2011) include changes in the landscape parameter, hydrological change using an elasticity approach is still black box: climatic factors are described only by ΔP and ΔEp, and non-climatic factors are described only by changes in the landscape parameter (Fu, 1981;Choudhury, 1999).

### 4.4. All models are wrong, but a multi-model approach is useful

The similar results found between Budyko decomposition and the paired catchment approach in our study corroborate previous studies that employ multiple-approach methods for attributing long-term hydrological changes to climatic and non-climatic factors (e.g. Ning et al. (2018), Wu et al. (2017), Zhao et al. (2014)). For example, Zhao et al. (2014) found consistent results between ΔQ attribution using a Budyko approach and linear regression across 18 stream gaging stations in the middle reaches of the Yellow River basin of the Loess Plateau in China. Wu et al. (2017) compared ten commonly used attribution methods in a case study in the Yahne River basin, also on the Loess Plateau. Wu et al. (2017) found that Budyko-based elasticity methods and hydrological modeling using the SWAT model (Arnold et al., 1998) produced similar estimates of hydrological changes attributed to climate change and human activities. However, estimates based on empirical statistical methods, which in this study included linear regression and double mass curve analysis, differed. Statistical approaches over-estimated the relative contribution of human activities where $\Delta Q_H$ ranged between 82 and 87 %. $\Delta Q_H$ estimated using elasticity and hydrologic modeling ranged between 43 and 54 %. Most recently, Ning et al. (2018) compared the performance of four Budyko-based methods (total differential, complementary, extrapolation, and decomposition) for attributing long-term hydrological change in thirteen catchments also located in Loess Plateau. They found similar results for total differential, complementary, and decomposition approaches but errors were greatest using the extrapolation method.

The multiple method approach also highlighted potential limitations and benefits of each approach. For example, the paired catchment approach was useful for assessing the efficacy of applying Budyko decomposition to the headwater-scale. Despite its limitations, our quantitative analysis of performance and assessment of directional change assumptions of Budyko decomposition provides evidence and efficacy for the first time for applying the approach to the headwater-scale. And by applying decomposition to watershed 4, we were able to quantify how streamflow of a reference catchment of a long-term, on-going paired catchment study has changed overtime. This is important for two reasons. The first is that the hydrological behavior of a reference catchment must be stationary over time (Andréassian, 2004). The second, and related to the first, is that the relationship between reference and treatment catchment is statistically stable overtime (Zégre et al., 2010). Despite being central to the paired catchment study design, stationarity and stability are more often assumed and seldom tested for in contemporary paired catchment studies (Zégre et al., 2010). Our study, along with other recent studies of reference catchments located throughout North

America (Jones et al., 2012;Creed et al., 2014;Guillen et al.), show that the hydrology of reference catchments is indeed changing overtime, responding to both climate and legacy land use disturbances. In order to minimize the rate of type I error and type II error that can potentially obfuscate the actual effects of forest management in paired catchment models (Zégre et al., 2010), climate and non-climatic drivers of non-stationarity and stability should be included. Ford et al. (2011b), Vose et al. (2012), and Kelly et al. (2016) are examples of studies that included additional explanatory variables that account for other sources of variations (e.g., precipitation, time since disturbance, vegetation recovery). Despite their inherent limitations of each approach, taken collectively they are useful tools for developing more robust approaches for attribution studies.

## 5. Conclusion

In this study, we used long-term water balance, stand vegetation data, and a multiple method approach to better understand how climate change alters streamflow across a gradient of forest disturbances in headwater catchments located at the Fernow Experimental Forest in the eastern United States. Forest harvesting was the dominant driver of streamflow changes across catchments and an important factor for a catchment's sensitivity to climate change. While disturbed catchments were more sensitive to climate change, not all disturbances were equal when it came to catchment sensitivity to climate change. Disturbance largely overshadowed the effects of climate change in catchments with high magnitude/low frequency disturbance and amplified climate change effects in a catchment with low magnitude/high frequency disturbances. Our study adds to a growing body of evidence that climate change and forest disturbances interact in complex ways, sometimes amplifying, offsetting, or masking changes in streamflow.

This study also highlights the benefits of using a multiple method approach for studying climate and disturbance effects on hydrology. Importantly, we revealed that streamflow from a headwater reference catchment, which is often assumed to be stationary in paired catchment studies, has in fact changed overtime. Furthermore, results from the multiple method approach provides for the first time, the efficacy of applying Budyko-based elasticity approaches to the headwater scale - a scale potentially useful to resource managers and decision makers than macro-scale applications of Budyko since management decisions tend to be more place-based. Given the similarity of Budyko-based formulations (Fu, 1981;Choudhury, 1999;Sposito, 2017), we expect our results to broadly be applicable to other Budyko-based elasticity approaches applied to the headwater scale.

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2019.07.188.

## Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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## References

Adams, M.B., Edwards, P.J., Ford, W.M., Schuler, T.M., Thomas-Van Gundy, M., Wood, F., 2012. Fernow Experimental Forest: Research History and Opportunities.

Adams, M.B., Kochenderfer, J.N., 2014. Recovery of Central Appalachian forested watersheds. Comparison of Fernow and Coweeta. In: Swank, W.T., Webster, J.R. (Eds.), Long-Term Response of a Forest Watershed Ecosys- tem. Clearcutting in the Southern Appalachians (The Long-term Ecological Research Network Series). Oxford University Press, Oxford, UK.

Ainsworth, E.A., Rogers, A., 2007. The response of photosynthesis and stomatal conductance to rising [$CO_2$]: mechanisms and environmental interactions. Plant Cell Environ. 30, 258–270.

Alfieri, L., Burek, P., Feyen, L., Forzieri, G., 2015. Global warming increases the frequency of river floods in Europe. Hydrol. Earth Syst. Sci. 19, 2247–2260. HESS.

Andréassian, V., 2004. Waters and forests: from historical controversy to scientific debate. J. Hydrol. 291, 1–27.

Archibald, J.A., Walter, M.T., 2014. Do energy-based pet models require more input data than temperature-based models? An evaluation at four humid fluxnet sites. JAWRA J. Am. Water Resour. Assoc. 50, 497–508.

Arnold, J.G., Srinivasan, R., Muttiah, R.S., Williams, J.R., 1998. Large area hydrologic modeling and assessment part i: model development1. JAWRA J. Am. Water Resour. Assoc. 34, 73–89.

Bates, C., 1921. First results in the streamflow experiment, Wagon Wheel Gap, Colorado. J. For. 19, 402–408.

Beerling, D.J., 1996. Ecophysiological responses of woody plants to past $CO_2$ concentrations. Tree Physiol. 16, 389–396. https://doi.org/10.1093/treephys/16.4.389.

Beerling, D.J., 1997. Carbon isotope discrimination and stomatal responses of mature Pinus sylvestris l. Trees exposed in situ for three years to elevated $CO_2$ and temperature. Acta Oecol. 18, 697–712.

Berghuijs, W.R., Larsen, J.R., van Emmerik, T.H.M., Woods, R.A., 2017. A global assessment of runoff sensitivity to changes in precipitation, potential evaporation, and other factors. Water Resour. Res. 53, 8475–8486.

Berghuijs, W.R., Woods, R.A., Hrachowitz, M., 2014. A precipitation shift from snow towards rain leads to a decrease in streamflow. Nat. Clim. Chang. 4, 583.

Betts, R.A., Boucher, O., Collins, M., Cox, P.M., Falloon, P.D., Gedney, N., Hemming, D.L., Huntingford, C., Jones, C.D., Sexton, D.M.H., Webb, M.J., 2007. Projected increase in continental runoff due to plant responses to increasing carbon dioxide. Nature 448, 1037–1041. https://doi.org/10.1038/nature06045.

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

Box, G., Jenkins, G., 1976. Time Series Analysis Forecasting and Control. Holden-Day, San Francisco.

Brooks, R.P., Limpisathian, P.W., Gould, T., Mazurczyk, T., Sava, E., Mitsch, W.J., 2018. Does the Ohio River flow all the way to New Orleans? JAWRA J. Am. Water Resour. Assoc.(pp.n/a–n/a).

Brown, A.E., Zhang, L., McMahon, T.A., Western, A.W., Vertessy, R.A., 2005. A review of paired catchment studies for determining changes in water yield resulting from alterations in vegetation. J. Hydrol. 310, 28–61.

Brzostek, E.R., Dragoni, D., Schmid, H.P., Rahman, A.F., Sims, D., Wayson, C.A., Johnson, D.J., Phillips, R.P., 2014. Chronic water stress reduces tree growth and the carbon sink of deciduous hardwood forests. Glob. Chang. Biol. 20, 2531–2539.

Budyko, M., 1974. Climate and Life. Academic, San Diego, California.

Bureau, U.C., 2009. National Population Estimates: Characteristics.

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

Caldwell, P.V., Miniat, C.F., Elliott, K.J., Swank, W.T., Brantley, S.T., Laseter, S.H., 2016. Declining water yield from forested mountain watersheds in response to climate change and forest mesophication. Glob. Chang. Biol. 22, 2997–3012.

Campbell, I., Doeg, T., 1989. Impact of timber harvesting and production on streams: a review. Aust. J. Mar. Freshwater Res. 40, 519–539.

Campbell, J.L., Driscoll, C.T., Pourmokhtarian, A., Hayhoe, K., 2011. Streamflow responses to past and projected future changes in climate at the Hubbard Brook Experimental Forest, New Hampshire, United States. Water Resour. Res. 47, W02514.

Chawla, I., Mujumdar, P.P., 2015. Isolating the impacts of land use and climate change on streamflow. Hydrol. Earth Syst. Sci. 19, 3633–3651. HESS.

Choudhury, B., 1999. Evaluation of an empirical equation for annual evaporation using field observations and results from a biophysical model. J. Hydrol. 216, 99–110.

Creed, I.F., Spargo, A.T., Jones, J.A., Buttle, J.M., Adams, M.B., Beall, F.D., Booth, E.G., Campbell, J.L., Clow, D., Elder, K., Green, M.B., Grimm, N.B., Miniat, C., Ramlal, P., Saha, A., Sebestyen, S., Spittlehouse, D., Sterling, S., Williams, M.W., Winkler, R., Yao, H., 2014. Changing forest water yields in response to climate warming: results from long-term experimental watershed sites across North America. Glob. Chang. Biol.

Döll, P., Fiedler, K., Zhang, J., 2009. Global-scale analysis of river flow alterations due to water withdrawals and reservoirs. Hydrol. Earth Syst. Sci. 13, 2413–2432. HESS.

Donohue, R.J., Roderick, M.L., McVicar, T.R., 2007. On the importance of including vegetation dynamics in Budyko's hydrological model. Hydrol. Earth Syst. Sci. 11, 983–995. HESS.

Donohue, R.J., Roderick, M.L., McVicar, T.R., 2011. Assessing the differences in sensitivities of runoff to changes in climatic conditions across a large basin. J. Hydrol. 406, 234–244.

Dooge, J.C., 1992. Sensitivity of runoff to climate change: a Hortonian approach. Bull. Am. Meteorol. Soc. 73, 2013–2024.

Duan, K., Sun, G., Caldwell, P.V., McNulty, S.G., Zhang, Y., 2018. Implications of upstream flow availability for watershed surface water supply across the conterminous United States. JAWRA J. Am. Water Resour. Assoc. 54, 694–707. https://doi.org/10.1111/1752-1688.12644.

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

Dye, P.J., Croke, B.F.W., 2003. Evaluation of streamflow predictions by the IHACRES rainfall-runoff model in two South African catchments. Environ. Model. Softw. 18, 705–712.

Edwards, P.J., Wood, F., 2011, 17 August 2017a. Fernow Experimental Forest daily air temperature. Technical Report. U.S. Department of Agriculture, Forest Service, Northern Research Station.

Edwards, P.J., Wood, F., 2011, 17 August 2017b. Fernow Experimental Forest daily precipitation. Technical Report. U.S. Department of Agriculture, Forest Service, Northern Research Station.

Edwards, P.J., Wood, F., 2011, 17 August 2017c. Fernow experimental forest daily streamflow. Techical Report. U.S. Department of Agriculture, Forest Service, Northern Research Station.

Edwards, P.J., Wood, F., 2012. Fernow Experimental Forest Daily Streamflow.

Ehleringer, J.R., Cerling, T.E., 1995. Atmospheric $CO_2$ and the ratio of intercellular to ambient $CO_2$ concentrations in plants. Tree Physiol. 15, 105–111.

Fernandez, R., Zegre, N., 2019. Seasonal changes in water and energy balances over the Appalachian region and beyond throughout the twenty-first century. J. Appl. Meteorol. Climatol. 58, 1079–1102.

Ferrari, J.R., Lookingbill, T.R., McCormick, B., Townsend, P.A., Eshleman, K.N., 2009. Surface mining and reclamation effects on flood response of watersheds in the central Appalachian Plateau region. Water Resour. Res. 45, https://doi.org/10.1029/2008WR007109.

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

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

Fritsch, F.N., Carlson, R.E., 1980. Monotone piecewise cubic interpolation. SIAM J. Numer. Anal. 17, 238–246.

Fu, B.P., 1981. On the calculation of the evaporation from land surface [in Chinese]. Sci. Atmos. Sin. 5, 23–31.

Gaertner, B.A., Zegre, N., Warner, T., Fernandez, R., He, Y., Merriam, E.R., 2019. Climate, forest growing season, and evapotranspiration changes in the central Appalachian Mountains, USA. Sci. Total Environ. 650, 1371–1381.

Gao, G., Fu, B., Wang, S., Liang, W., Jiang, X., 2016. Determining the hydrological responses to climate variability and land use/cover change in the Loess Plateau with the Budyko framework. Sci. Total Environ. 557-558, 331–342.

Gassman, P.W., Sadeghib, A.M., Srinivasan, R., 2007. The soil and water assessment tool: historical development, applications, and future research directions. Trans. ASABE 50, 1211–1250.

Gedney, N., Cox, P.M., Betts, R.A., Boucher, O., Huntingford, C., Stott, P.A., 2006. Detection of a direct carbon dioxide effect in continental river runoff records. Nature 439, 835–838. https://doi.org/10.1038/nature04504.

Gudmundsson, L., Greve, P., Seneviratne, S.I., 2016. The sensitivity of water availability to changes in the aridity index and other factors a probabilistic analysis in the Budyko space. Geophys. Res. Lett. 43, 6985–6994.

Guillen, L. A., Zegre, N., Fernandez, R., Evaluating the stability of headwaterPlease supply the year of publication.reference catchments from long-term paired watershed studies: an eastern perspective. in: American Geophysical Union Fall Meet. Suppl., Final Paper H33L-2236.

Harr, D.R., Fredriksen, R., Rothacher, J., 1979. Changes in streamflow following timber harvest in Southwestern Oregon. Techical Report. PNW- 249 USDA For. Serv. Res. Pap.

Hayhoe, K., Wake, C., Huntington, T., Luo, L., Schwartz, M., Sheffield, J., Wood, E., Anderson, B., Bradbury, J., DeGaetano, A., Troy, T., Wolfe, D., 2007. Past and future changes in climate and hydrological indicators in the US Northeast. Clim. Dyn. 28, 381–407.

Helsel, D., Hirsch, R.M., 1992. Statistical Methods in Water Resources. Elsevier, Amsterdam.

Herrick, J.D., Maherali, H., Thomas, R.B., 2004. Reduced stomatal conductance in sweetgum (Liquidambar styraciflua) sustained over long-term $CO_2$ enrichment. New Phytol. 162, 387–396.

Hewlett, J.D., 1971. Comments on the catchment experiment to determine vegetal effects on water yield. Water Resour. Bull. 7, 376–381.

Hornbeck, J.W., Adams, M.B., Corbett, E.S., Verry, E.S., Lynch, J.A., 1993. Long-term impacts of forest treatments on water yield: a summary for Northeastern USA. J. Hydrol. 150, 323–344.

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

Huntington, T.G., 2008. $CO_2$-induced suppression of transpiration cannot explain increasing runoff. Hydrol. Process 22, 311–314.

Hwang, T., Martin, K. L., Vose, J. M., Wear, D., Miles, B., Kim, Y., Band, L. E. Nonstationary hydrologic behavior in forested watersheds is mediated by climate-induced changes in growing season length and subsequent vegetation growth. Water Resour. Res.54 , 5359–5375.

Ice, G.G., Stednick, J.D., 2004a. A Century of Forest and Wildland Watershed Lessons. Society of Ameican Foresters, Bethesda, Maryland.

Ice, G.G., Stednick, J.D., 2004b. Forest watershed research in the United States. Forest History Today Spring/Fall.

Jones, J., 2011. Hydrologic responses to climate change: considering geographic context and alternative hypotheses. Hydrol. Process.(pp.n/a–n/a) N2r.

Jones, J.A., Creed, I.F., Hatcher, K.L., Warren, R.J., Adams, M.B., Benson, M.H., Boose, E., Brown, W.A., Campbell, J.L., Covich, A., Clow, D.W., Dahm, C.N., Elder, K., Ford, C.R., Grimm, N.B., Henshaw, D.L., Larson, K.L., Miles, E.S., Miles, K.M., Sebestyen, S.D., Spargo, A.T., Stone, A.B., Vose, J.M., Williams, M.W., 2012. Ecosystem processes and human influences regulate streamflow response to climate change at long-term ecological research sites. BioScience 62, 390–404.

Karl, T.R., Knight, R.W., 1998. Secular trends of precipitation amount, frequency, and intensity in the United States. Bull. Am. Meteorol. Soc. 79, 231–241.

Kelly, C.N., McGuire, K.J., Miniat, C.F., Vose, J.M., 2016. Streamflow response to increasing precipitation extremes altered by forest management. Geophys. Res. Lett. 43, 3727–3736.

Kendall, C., McDonnell, J.J., 1998. Isotope Tracers in Catchment Hydrology. Elsevier, New York.

Kim, Y., Band, L.E., Song, C., 2014. The influence of forest regrowth on the stream discharge in the North Carolina Piedmont Watersheds. JAWRA J. Am. Water Resour. Assoc. 50, 57–73.

Kochenderfer, J.N., Edwards, P.J., Helvey, J.D., 1990. Land management and water yield in the Appalachians. Watershed Planning and Analysis in Action.

Krakauer, N.Y., Fung, I., 2008. Mapping and attribution of change in streamflow in the conterminous United States. Hydrol. Earth Syst. Sci. 12, 1111–1120. HESS.

Laseter, S.H., Ford, C.R., Vose, J.M., Swift, L.W., Jr, 2012. Long-term temperature and precipitation trends at the Coweeta Hydrologic Laboratory, Otto, North Carolina, USA. Hydrol. Res. 43, 890–901.

Leuzinger, S., Korner, C., 2010. Rainfall distribution is the main driver of runoff under future $CO_2$-concentration in a temperate deciduous forest. Glob. Chang. Biol. 16, 246–254.

Li, F., Zhang, G., Xu, Y.J., 2014. Separating the impacts of climate variation and human activities on runoff in the Songhua River Basin, Northeast China. Water 6, 3320–3338.

Lins, H.F., Slack, J.R., 1999. Streamflow trends in the United States. Geophys. Res. Lett. 26, 227–230.

Ljung, G., Box, G.E.P., 1978. On a measure of lack of fit in time series models. Biometrika 66, 67–72.

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

Ma, H., Yang, D., Tan, S.K., Gao, B., Hu, Q., 2010. Impact of climate variability and human activity on streamflow decrease in the Miyun Reservoir Catchment. J. Hydrol. 389, 317–324.

McCabe, G.J., Wolock, D.M., 2002. A step increase in streamflow in the conterminous United States. Geophys. Res. Lett. 29, 2185.

McDonnell, J.J., Bonell, M., Stewart, M.K., Pearce, A.J., 1990. Deuterium variations in storm rainfall: implications for stream hydrograph separation. Water Resour. Res. 26, 455–458.

Meinzer, F.C., Woodruff, D.R., Eissenstat, D.M., Lin, H.S., Adams, T.S., McCulloh, K.A., 2013. Above- and belowground controls on water use by trees of different wood types in an eastern US deciduous forest. Tree Physiol. 33, 345–356.

Merriam, E. R., Fernandez, R., Petty, J. T., Zegre, N. Can brook trout survive climate change in large rivers? If it rains. Sci. Total Environ. 607-608,1225–1236, 2017.

Milly, P.C.D., 1994. Climate, soil water storage, and the average annual water balance. Water Resour. Res. 30, 2143–2156.

Moran, M.S., Peters, D.P.C., McClaran, M.P., Nichols, M.H., Adams, M.B., 2008. Long-term data collection at usda experimental sites for studies of ecohydrology. Ecohydrology 1, 377–393.

Moriasi, D.N., Arnold, J.G., Van Liew, M.W., Bingner, R.L., Harmel, R.D., Veith, T., 2007. Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. Trans. ASABE 50, 885–900.

Nash, J., Sutcliffe, J., 1970. River flow forecasting through conceptual models. Part I: a discussion of principles. J. Hydrol. 10, 282–290.

Ning, T., Li, Z., Feng, Q., Liu, W., Li, Z., 2018. Comparison of the effectiveness of four Budyko-based methods in attributing long-term changes in actual evapotranspiration. Sci. Rep. 8, 12665–12665. 30140034[pmid] PMC6107678[pmcid] 10.1038/s41598-018-31036-x[PII] Sci Rep.

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

Oren, R., Pataki, D., 2001. Transpiration in response to variation in microclimate and soil moisture in southeastern deciduous forests. Oecologia 127, 549–559.

Padrón, R.S., Gudmundsson, L., Greve, P., Seneviratne, S.I., 2017. Large-scale controls of the surface water balance over land: insights from a systematic review and meta-analysis. Water Resour. Res. 53, 9659–9678.

Patric, J., Reinhart, K., 1971. Hydrologic effects of deforesting two mountain watersheds in West Virginia. Water Resour. Res. 7, 1182–1188.

Patterson, L.A., Lutz, B., Doyle, M.W., 2012. Streamflow changes in the South Atlantic, United States during the mid- and late 20th century. J. Am. Water Resour. Assoc. 48, 1126–1138.
Patterson, L.A., Lutz, B., Doyle, M.W., 2013. Climate and direct human contributions to changes in mean annual streamflow in the South Atlantic, U.S.. Water Resour. Res.(pp.n/a–n/a).
M., E., Payne, K., Warrington, S., 2002. High stakes: the future for mountain societies. Technical Report. Panos Institute.
Peñuelas, J., Canadell, J.G., Ogaya, R., 2011. Increased water-use efficiency during the $20^{th}$ century did not translate into enhanced tree growth. Glob. Ecol. Biogeogr. 20, 597–608.
Petersen, T., Devineni, N., Sankarasubramanian, A., 2012. Seasonality of monthly runoff over the continental United States: causality and relations to mean annual and mean monthly distributions of moisture and energy. J. Hydrol. 468-469, 139–150.
Pitchford, J., Wu, C., Lin, L., Petty, J., Thomas, R., Veselka, W., Welsch, D., Zégre, N., Anderson, J., 2011. Climate change effects on hydrology and ecology of wetlands in the Mid-Atlantic Highlands. Wetlands (pp.1-13).
Prentice, I.C., Harrison, S.P., 2009. Ecosystem effects of $CO_2$ concentration: evidence from past climates. Clim. Past 5, 297–307.
R Core Team, 2013. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. http://www.R-project.org/.
Renner, M., Bernhofer, C., 2012. Applying simple water-energy balance frameworks to predict the climate sensitivity of streamflow over the continental United States. Hydrol. Earth Syst. Sci. 16, 2531–2546. HESS.
Robison, A.L., Scanlon, T.M., 2018. Climate change to offset improvements in watershed acid-base status provided by clean air act and amendments: a model application in Shenandoah National Park, Virginia. J. Geophys. Res.: Biosci. 123, 2863–2877.
Roderick, M.L., Farquhar, G.D., 2011. A simple framework for relating variations in runoff to variations in climatic conditions and catchment properties. Water Resour. Res. 47, n/a n/a.
Roderick, M.L., Greve, P., Farquhar, G.D., 2015. On the assessment of aridity with changes in atmospheric $CO_2$. Water Resour. Res. 51, 5450–5463.
Rodriguez-Iturbe, I., 2000. Ecohydrology: a hydrologic perspective of climate-soil-vegetation dynamies. Water Resour. Res. 36, 3–9.
Rodriguez-Iturbe, I., Porporato, A., 2004. Ecohydrology of Watercontrolled Ecosystems: Soil Moisture and Plant Dynamics. Cambridge University Press, Cambridge.
Schaake, J.C., 1990. From climate to flow. . John Wiley, New York, pp. 177–206. Cited By :1 Export Date: 23 January 2019.
Schade, T.G., Shuster, W.D., 2005. Paired watershed study of land-use and climate change impact on small streams. In: Walton, R. (Ed.), World Water and Environmental Resources Congress. ASCE., https://doi.org/10.1061/40792(173)484.
Schuler, T., McGill, D., 2007. Long-term assessment of financial maturity, diameter-limit selection in the central appalachians. Technical Report. NRS-2.
Schuler, T.M., 2004. Fifty years of partial harvesting in a mixed mesophytic forest: composition and productivity. Can. J. For. Res. 34, 985–997.
Schuler, T.M., Thomas-Van Gundy, M., Brown, J.P., Wiedenbeck, J.K., 2016. Managing appalachian hardwood stands using four management practices: 60-year results. For. Ecol. Manag.
Seibert, J., Vis, M.J.P., 2012. Teaching hydrological modeling with a user-friendly catchment-runoff-model software package. Hydrol. Earth Syst. Sci. 16, 3315–3325. HESS.
Sen, P.K., 1968. Estimates of the regression coefficient based on Kendall's tau. J. Am. Stat. Assoc. 63, 1379–1389.
Smith, J.A., Baeck, M.L., Ntelekos, A.A., Villarini, G., Steiner, M., 2011. Extreme rainfall and flooding from orographic thunderstorms in the central Appalachians. Water Resour. Res. 47, W04514.
Sposito, G., 2017. Understanding the Budyko equation. Water 9, 236.
Stednick, J.D., 1996. Monitoring the effects of timber harvest on annual water yield. J. Hydrol. 176, 79–95.
Stoy, P.C., Katul, G.G., Siqueira, M.B.S., Juang, J.-Y., Novick, K.A., McCarthy, H.R., Christopher Oishi, A., Uebelherr, J.M., Kim, H.-S., Oren, R.A.M., 2006. Separating the effects of climate and vegetation on evapotranspiration along a successional chronosequence in the southeastern US. Glob. Chang. Biol. 12, 2115–2135.
Tao, J., Barros, A.P., 2018. Multi-year atmospheric forcing datasets for hydrologic modeling in regions of complex terrain Ú methodology and evaluation over the integrated precipitation and hydrology experiment 2014 domain. J. Hydrol.: Reg. Stud. 22.
Tetzlaff, D., Soulsby, C., Buttle, J., Capell, R., Carey, S.K., Laudon, H., McDonnell, J., McGuire, K., Seibert, J., Shanley, J., 2013. Catchments on the cusp? Structural and functional change in northern ecohydrology. Hydrol. Process. 27, 766–774.
Thomas, R.B., Spal, S.E., Smith, K.R., Nippert, J.B., 2013. Evidence of recovery of juniperus Virginiana trees from sulfur pollution after the clean air act. Proc. Natl. Acad. Sci. 110, 15319–15324.
Trenberth, K., 1999. Conceptual framework for changes of extremes of the hydrological cycle with climate change. Climatic Change 42, 327–339.
Trimble, G.R., 1977. A history of the Fernow Experimental Forest and the Parsons Timber and Watershed Laboratory. Techreport. Gen. Tech.Rep. NE-28.
van der Sleen, P., Groenendijk, P., Vlam, M., Anten, N.P.R., Boom, A., Bongers, F., Pons, T.L., Terburg, G., Zuidema, P.A., 2015. No growth stimulation of tropical trees by 150 years of $CO_2$ fertilization but water-use efficiency increased. Nat. Geosci. 8, 24–28.
Vose, J.M., Ford, C.R., Laseter, S., Dymond, S., Sun, G., Adams, M.B., Sebestyen, S., Campbell, J., Luce, C., Amatya, D., Elder, K., Heartsill Scalley, T., 2012. Can Forest Watershed Management Mitigate Climate Change Effects on Water Resources.
Wang, D., Cai, X., 2010. Comparative study of climate and human impacts on seasonal baseflow in urban and agricultural watersheds. Geophys. Res. Lett. 37.
Wang, D., Hejazi, M., 2011. Quantifying the relative contribution of the climate and direct human impacts on mean annual streamflow in the contiguous United States. Water Resour. Res. 47, W00J12.
Wang, X., 2014. Advances in separating effects of climate variability and human activity on stream discharge: an overview. Adv. Water Resour. 71, 209–218.
Warren, J.M., Pötzelsberger, E., Wullschleger, S.D., Thornton, P.E., Hasenauer, H., Norby, R.J., 2011. Ecohydrologic impact of reduced stomatal conductance in forests exposed to elevated $CO_2$. Ecohydrology 4, 196–210.
Weedfall, R., Dickerson, W., 1965. Climate of Canaan Valley and Blackwater Falls State Park, West Virginia. Techreport,Station Rep. 43.West Virginia University Agricultural Experiment Station.
Weitzman, S., 1949. The fernow experimental forest. Techreport. U.S. Department of Agriculture, Northeastern Forest Experimental Station, Upper Darby, PA.
Wu, J., Miao, C., Zhang, X., Yang, T., Duan, Q., 2017. Detecting the quantitative hydrological response to changes in climate and human activities. Sci. Total Environ. 586, 328–337.
Wullschleger, S.D., Hanson, P.J., Todd, D.E., 2001. Transpiration from a multi-species deciduous forest as estimated by xylem sap flow techniques. For. Ecol. Manag. 143, 205–213.
Zégre, N., Skaugset, A.E., Som, N.A., McDonnell, J.J., Ganio, L.M., 2010. In lieu of the paired catchment approach: hydrologic model change detection at the catchment scale. Water Resour. Res. 46, W11544.
Zhang, L., Hickel, K., Dawes, W.R., Chiew, F.H.S., Western, A.W., Briggs, P.R., 2004. A rational function approach for estimating mean annual evapotranspiration. Water Resour. Res. 40, W02502.
Zhang, Y., Guan, D., Jin, C., Wang, A., Wu, J., Yuan, F., 2011. Analysis of impacts of climate variability and human activity on streamflow for a river basin in northeast China. J. Hydrol. 410, 239–247.
Zhang, Y.K., Schilling, K.E., 2006. Increasing streamflow and baseflow in Mississippi River since the 1940s: effect of land use change. J. Hydrol. 324, 412–422.
Zhao, F., Zhang, L., Xu, Z., Scott, D.F., 2010. Evaluation of methods for estimating the effects of vegetation change and climate variability on streamflow. Water Resour. Res. 46, W03505.
Zhao, G., Tian, P., Mu, X., Jiao, J., Wang, F., Gao, P., 2014. Quantifying the impact of climate variability and human activities on streamflow in the middle reaches of the Yellow River Basin, China. J. Hydrol. 519, 387–398.