Forest Ecology and Management 529 (2023) 120718

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

A global synthesis of hydrological sensitivities to deforestation and forestation

Yiping Hou [a], Xiaohua Wei [a,*], Mingfang Zhang [b], Irena F. Creed [c], Steven G. McNulty [d], Silvio F.B. Ferraz [e]

[a] *Department of Earth, Environmental and Geographic Sciences, University of British Columbia (Okanagan Campus), 1177 Research Road, Kelowna, British Columbia V1V 1V7, Canada*

[b] *School of Resources and Environment, University of Electronic Science and Technology of China, Chengdu 611731, China*

[c] *Department of Physical and Environmental Sciences, University of Toronto, 1265 Military Trail, Toronto, Ontario M1C 1A4, Canada*

[d] *USDA Southeast Regional Climate Hub, P.O. Box 12254, Research Triangle Park, NC 27709-2254, United States*

[e] *Department of Forest Sciences, University of Sao Paulo, Sao Paulo 13400-970, Brazil*

ARTICLE INFO

Keywords:
Hydrological sensitivity
Deforestation
Forestation
Inter-annual and intra-annual climate
Watershed property
Water retention capacity
LAI

ABSTRACT

Hydrological sensitivity to forest change, defined as hydrological response intensity (%) per unit of forest cover change (%), is essential for understanding the magnitude of possible hydrological consequences caused by forest disturbance (e.g., deforestation, wildfire, and insect infestation) or forestation (e.g., reforestation and afforestation). This synthesis estimated and compared hydrological sensitivities (HS_f) of annual streamflow to deforestation and forestation based on quantitative analyses of 311 watersheds across the globe. The roles of climate (both inter-annual and intra-annual) and watershed properties (e.g., topography-related water retention capacity, site condition, watershed size, forest type, and soil type) in HS_f were assessed in deforestation and forestation groups, respectively. The key findings are: (1) hydrological sensitivities to forestation are significantly larger than those to deforestation, with an average value of 1.24% and 0.91% change in annual streamflow following 1% forestation and deforestation, respectively; (2) annual climate dryness (defined by PET/P at the annual scale) is the primary contributor to HS_f to deforestation and forestation, with a relative importance of 75.5% and 60.6%, respectively, but intra-annual synchronicity of water and energy (i.e., greater matching in the timing of maximum P and maximum PET at the monthly scale) produces a significant impact on HS_f to forestation; (3) leaf area index (LAI) has a contrasting effect on HS_f to deforestation (negative response) versus forestation (positive response); (4) water retention index (I_R) has a negative role in HS_f, demonstrating that watersheds with larger water retention capacities are less hydrologically sensitive, particularly in the forestation group; (5) contrast to our general expectation, hydrological sensitivities to forestation are significantly greater in larger watersheds; and (6) hydrological responses are more sensitive to deforestation in watersheds with pure forest types and are more sensitive to forest cover change in Lithosols-dominated watersheds. Our findings suggest that hydrological effects between deforestation and forestation are not simply reversed and demonstrate that hydrological sensitivities are significantly influenced by climate and watershed properties. Hydrological sensitivities and their contributing drivers must be considered in protecting water and other aquatic properties.

1. Introduction

Forests cover nearly one-third of the global landmass and play an essential role in regulating hydrological processes and, by extension, ecological functions and services, such as water supply, water purification, biodiversity, and carbon sequestration (Creed et al., 2016; Clerici et al., 2019; Liu et al., 2021; Zhang and Wei, 2021). However, forests are experiencing substantial forest management activities, for example, deforestation, reforestation, afforestation, conversion in response to agricultural intensification and expansion, and urbanization, to meet the needs of economic development and environmental protection. For example, the Global Forest Resources Assessment (2020) reported that global forests have decreased since 1990, with most forest harvesting activities in South America and Africa (FAO, 2020; Keenan et al., 2015).

* Corresponding author.
E-mail address: adam.wei@ubc.ca (X. Wei).

https://doi.org/10.1016/j.foreco.2022.120718
Received 24 December 2021; Received in revised form 24 November 2022; Accepted 4 December 2022
Available online 16 December 2022

In comparison, the most significant forest cover gain has been observed in Asia, primarily attributed to China and India's large-scale ecological restoration and plantation programs (FAO, 2020; Jones et al., 2018). These changes have stimulated a growing interest in assessing how hydrological processes respond to forest cover change induced by forest management activities, particularly globally (Creed and van Noordwijk, 2018; Creed et al., 2019; Li et al., 2017; Villarini and Wasko, 2021; Zhang et al., 2017; Zhang and Wei, 2021).

Most studies suggest that deforestation increases annual streamflow and increases the size and frequency of floods (Goeking and Tarboton, 2020), while forestation (afforestation and reforestation) has the opposite impact (Farley et al., 2005; Filoso et al., 2017; Jackson et al., 2005). Despite a general consistency in the direction of hydrological responses, there are significant variations in the magnitude of hydrological responses (Zhang et al., 2017; Zhang and Wei, 2021). For example, based on a global review, Zhang et al. (2017) found that 1.7 to 100% forest cover loss resulted in annual streamflow increases from 0.4 to 599.1%, while 0.7 to 100% forest cover gain resulted in annual streamflow changes from 0.7 to 167.7%. These large variations in hydrological responses suggest that hydrological sensitivities to forest change are likely related to the scale of the investigation, climate, type and severity of forest disturbance, and watershed properties (Zhang and Wei, 2021). As far as we know, hydrological sensitivities to deforestation or forestation remain poorly studied and synthesized, particularly at the global scale.

Forests go through a disturbance-recovery cycle. Following a disturbance (e.g., harvesting or wildfire), recovery or forestation occurs through natural regeneration or plantation. It is unclear whether deforestation and forestation produce similar magnitudes of hydrological response or if the hydrological impacts caused by deforestation can be reversed through forestation? Very few studies have directly compared the difference in hydrological responses or sensitivities between deforestation and forestation. For example, Liu et al. (2015) found that there was an increase of 113 mm/yr in annual streamflow due to deforestation, but a reduction of 51 mm/yr caused by forestation in the Meijiang Watershed, China. Swift and Swank (1981) found that two consecutive clear-cuts increased annual streamflow by 65% in 23 years and 40 % in 12 years, respectively in WS13 at the Coweeta Long-Term Experimental Forest, while streamflow recovery took 31 and 49 years, respectively. Despite being limited in their geographic scope, these studies indicate different hydrological sensitivities between deforestation and forestation. A synthesis of this topic at the global scale would help address this knowledge gap.

Hydrological sensitivity to forest change can be defined as the intensity of hydrological responses per unit of forest cover change. The concept is the opposite of hydrological resistance (Mitchell et al., 2016; Creed et al., 2014). A hydrologically sensitive watershed will experience significantly more overland flow and evapotranspiration (ET) changes than a hydrologically insensitive watershed. Additionally, a hydrologically sensitive watershed may require a more extended period to fully recover hydrological functions (Creed et al., 2014). Hydrological sensitivity can be affected by climate and watershed properties (e.g., tree species, land cover characteristics, topography, and landscape pattern) (Hou et al., 2021; Zhang et al., 2017; Zhang and Wei, 2021). For example, hydrological sensitivities to forest change were more significant in watersheds situated in semi-arid and arid regions compared to watersheds situated in humid regions (Hou et al., 2021; Peña-Arancibia et al., 2019). Further, hydrological sensitivities to forest change were smaller in watersheds with mature trees, mixed forest types, diverse landforms, and gentle slopes compare to watersheds with young trees, single forest types, simple landforms, and steep slopes (Creed et al., 2014; Zhou et al., 2015). For any watershed, watershed properties are a crucial factor for determining the hydrological response to external changes. Thus, understanding the contributing factors of hydrological sensitivity can help us better identify and manage forest areas with large hydrological sensitivities and associated negative consequences.

Research interests into the sensitivity of hydrological responses to forest change are growing. Recent theoretical analyses examined hydrological sensitivity associated with watershed characteristics and climate variability using the conceptual Budyko framework. For example, Zhou et al. (2015) used Fu's Budyko framework to show that the effects of land cover change on hydrological sensitivities (using runoff ratio as a proxy) were largest in watersheds with a wetness index (the ratio of precipitation to potential evapotranspiration) of 0.5 to 0.7. Further, Zhang et al. (2004) showed that the effects of watershed characteristic changes (represented as the changes in the Budyko parameter) on hydrological sensitivities (using the ratio of evapotranspiration to precipitation as a proxy) were largest in watersheds with a wetness index near 1.0. Although these analyses hardly address hydrological sensitivity to a specific land cover type such as forests, they provide valuable insights into hydrological sensitivities in a broader climate and land cover change. Watershed-based assessments have examined hydrological sensitivities by proposing a pre-defined index (Zhang et al., 2017; Li et al., 2017), applying the principle of elasticity (Kibria et al., 2016; Zheng et al., 2013), retrieving the sensitivity coefficient to the Budyko parameter (Berghuijs et al., 2017; Chen et al., 2021; Gudmundsson et al., 2016; Lv et al., 2019), and conducting hydrological modeling (Mo et al., 2021; Pomeroy et al., 2012). Despite growing research interest in the topic, to our knowledge, there remains a lack of a global review or synthesis that examines hydrological sensitivities to forest change induced by forest management activities and their contributing factors critically.

Zhang et al. (2017) proposed a hydrological sensitivity index to indicate the response intensity of annual runoff to forest cover change, and then applied this index to explore their relationships with climate, forest type, and hydrological regime in 312 watersheds around the globe. While hydrological responses to both natural and anthropogenic forest change were estimated, Zhang et al. (2017) did not evaluate the effects of forest management activities (i.e., deforestation and forestation) on hydrological sensitivities and did not answer the question that whether hydrological impacts of deforestation are simply reversed to forestation? Besides, the relative importance of integrated drivers in deforestation and forestation groups was not assessed. To fill this gap, we evaluated hydrological sensitivities to forest management activities (i.e., deforestation and forestation) and their contributing drivers using published data from non-modeling work. Our study addresses the following scientific questions: (1) Are hydrological sensitivities to deforestation and forestation the same? If not, (2) how do hydrological sensitivities to deforestation and forestation vary among climate classes? (3) How do hydrological sensitivities to deforestation and forestation vary among watershed property classes? And (4) to what extent do climate and watershed properties contribute to hydrological sensitivities to deforestation and forestation?

2. Methods

2.1. Data collection

We collected data from published papers that quantified the effects of forest cover change on hydrological processes. First, we searched *Web of Science* databases for published papers with the following terms: "hydrological" or "streamflow" or "runoff" or "runoff ratio" or "runoff coefficient" or "evapotranspiration" or "evapotranspiration ratio" AND "forest change" or "forest disturbance" or "deforestation" or "forestation" or "planting" in the title, abstract, or keywords. Among the selected papers, we then searched for those that reported: (1) forest change proportions (%); (2) the absolute or relative changes in annual streamflow caused by forest cover change; and (3) the watershed properties (e.g. slope and range in elevation). We included data from papers that applied paired watershed experiments (PWEs), conceptional frameworks (e.g., the Budyko and Tomer-Schilling frameworks), and statistical analyses of long-term data (e.g., graphic methods, trend analyses, and elasticity analyses). We focused on data for forest management activities (i.e., deforestation and forestation). Since data based on hydrological models generally have a coarse representation of forest cover change, they were excluded from the analysis. Our final data set includes 311 watersheds across the globe (Fig. 1 and Table S1), of which 218 watersheds are from deforestation activities with an average response period of 12 years (ranging from 1 to 41 years), while 93 watersheds are from forestation activities with an average response period of 18 years (ranging from 1 to 46 years). Most data on deforestation are from North America, Europe, and Australia, while most data on forestation are from Asia (Fig. 1). Detailed descriptions of data collection, auxiliary data sources, data preparations, and variable collections are provided in the Supplementary Materials (Sections S1 and S2).

2.2. Data analyses

2.2.1. Hydrological sensitivity to forest change

Hydrological sensitivity to forest change (HS_f) is defined as the response intensity (%) of streamflow per unit of forest cover change (%) (Equation (1); Zhang et al., 2017). This dimensionless index provides a unified measure for comparing hydrological response per unit of forest cover change among watersheds with different forest change proportions, climates, and watershed properties. Here, we focused on hydrological sensitivities of annual streamflow to deforestation and forestation.

$$HS_f = \left| \frac{\Delta Q_f\%}{\Delta F\%} \right| \quad (1)$$

where, $\Delta Q_f\%$ is the relative change in annual streamflow caused by forest change, which is calculated as the absolute change in annual streamflow (ΔQ_f, mm) divided by the long-term mean annual streamflow ($\overline{Q}$, mm); and $\Delta F\%$ is the proportion of forest change (%).

2.2.2. Integrated drivers of HS_f

Water retention capacity of a watershed plays an essential role in streamflow generation processes. Water retention capacity is a function of watershed properties (e.g., slope, landform complexity, site conditions, and land cover characteristics). Watersheds with steep slopes and significant elevation differences (i.e., relief) tend to have shorter water residence times, shorter flow paths, and consequently smaller water retention capacities than watersheds with gentle slopes and slight elevation differences (Jencso and McGlynn, 2011; Nippgen et al., 2011; Huang et al., 2020; Zhou et al., 2015). Here, we proposed the watershed's average water retention index (I_R, Equation (2)), calculated as a function of the watershed's average slope and range in elevation. Other topographic indices such as relief ratio, slope length factor, flow path length, and downslope distance gradient can also be used to explain topography-related water retention capacity (Hou et al., 2021; Jencso and McGlynn, 2011; Nippgen et al., 2011; Zhang et al., 2004). Unfortunately, those topographic indices were unavailable from the selected studies.

$$I_R = \frac{1}{\log(Slope) \times \log(Elev.diff.)} \quad (2)$$

where, I_R is the water retention index, and Elev.diff. is the elevation difference.

We chose long-term dryness index (DI) and leaf area index (LAI) in the growing season to represent averaged climate and site conditions that influence HS_f. DI, the ratio of mean annual potential evapotranspiration (PET) to mean annual precipitation (P), is an integrated indicator to reflect the interaction between energy and water on ET at the annual scale (Zhang et al., 2017). LAI represents the site condition related to land cover type, vegetation coverage, growth condition, and other biophysical processes (Donohue et al., 2007; Jin et al., 2021; Zhang et al., 2004). Watersheds with high LAI tend to have large evapotranspiration capacity, more layers of vegetation structure, and high vegetation cover (Khairiah et al., 2017).

2.2.3. Watershed classifications

Once hydrological sensitivity to forest change (HS_f) for each watershed was quantified, we compared HS_f among various watershed classes. Watersheds were classified in different ways. First, according to the

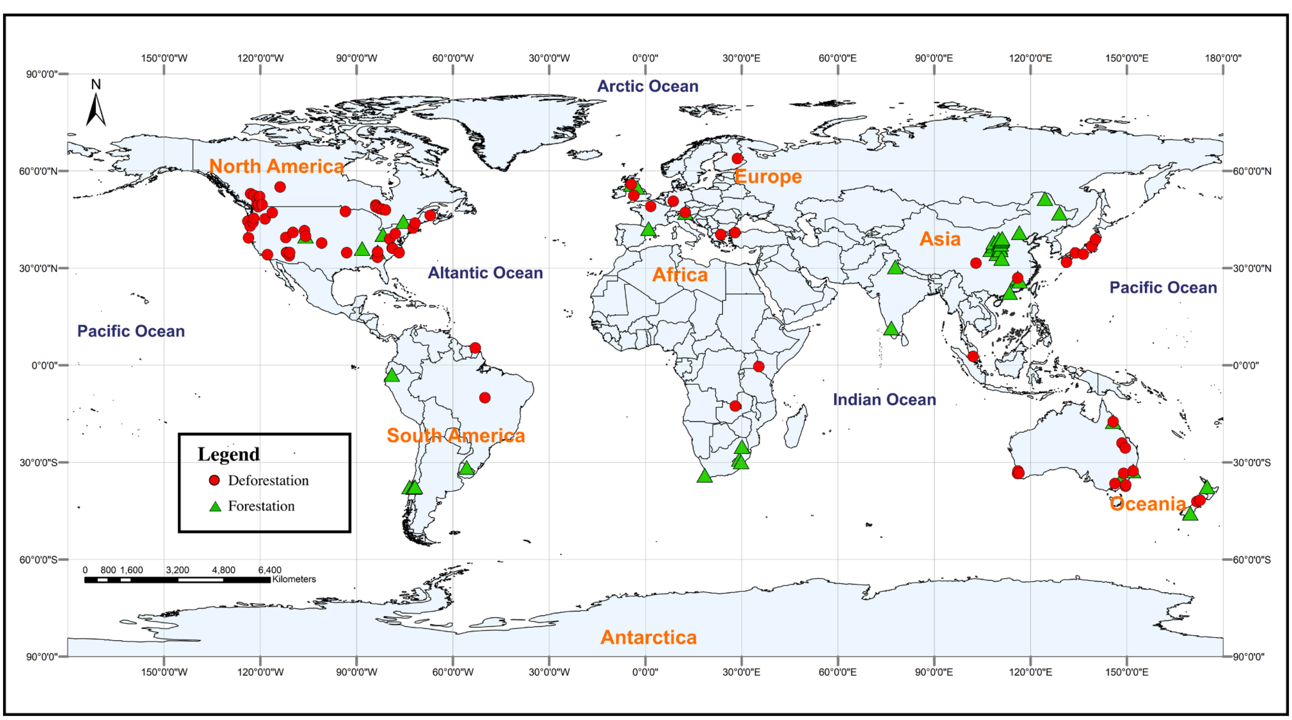

Fig. 1. Locations of the selected study sites with the red circles representing watersheds with deforestation activities and the green triangles representing watersheds with forestation activities. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

long-term DI, watersheds were classified into water-limited (WL) and energy-limited (EL) watersheds. Watersheds with a long-term DI less than 1.0 belong to energy-limited conditions, while water-limited watersheds have long-term DI>1.0 (Creed et al., 2014; Patterson et al., 2013). Second, watersheds were classified based on their intra-annual synchronicity of water supply (P) and energy demand (PET). We applied the intervals between maximum monthly P timing and maximum monthly PET timing (PfPET) to measure the matching of water supply and energy demand (Berghuijs and Woods, 2016; Shao et al., 2012). Based on calculated intervals, watersheds were grouped into synchronized systems with PfPET less than or equal to 2 and desynchronized systems with PfPET>2. Third, watersheds were classified as tropical, arid, temperate, or continental climate zones according to the Köppen-Geiger climate classification. Site locations in the Köppen-Geiger climate classifications can be found in Figure S1 in the Supplementary Materials. Finally, watersheds were classified according to the water retention index, watershed size, soil type, and forest type (Table 1, Section S2.4 and Table S1 in the Supplementary Materials).

2.2.4. *Statistical analyses*

The nonparametric Mann-Whitney *U* test was used to detect statistically significant differences in HS_f between watershed classes because there is no explicit requirement for data distribution (Aryal and Zhu, 2020; Mann and Whitney, 1947).

We used linear and nonlinear methods to evaluate the relationships between influencing drivers and HS_f as well as their relative importance. Multiple linear regression (MLR) models can indicate drivers' positive or negative roles in HS_f, and the standardized beta coefficients can indicate the relative importance of each driver in HS_f. We also used a regression tree-based machine learning model, the gradient boosting machine (GBM), to explore the nonlinear relationships and relative importance of each driver in HS_f (Giles-Hansen et al., 2021; Hallema et al., 2018). Although the GBM can describe nonlinear regression relationships, it runs as a black box with no specific indications of drivers' positive or negative roles. Thus, the combined estimation of two different methods was used to provide more robust results. MLR models were performed separately for HS_f in both deforestation and forestation groups with a significant level of 0.05. For GBM, we applied the R 'GBM' package to build deforestation and forestation model groups (R Core Team, 2016). 15-fold cross-validation repeated three times was used to tune the GBM models, and the model with the minimum root mean square error (RMSE) was selected to determine their relative importance.

Table 1
The sample size (N) across different watershed classes.

Category			Deforestation (N = 218)	Forestation (N = 93)
Climate	Inter-annual	Energy-limited (DI≤1.0)	161	54
		Water-limited (DI>1.0)	57	39
	Intra-annual	Synchronized (PfPET≤ 2)	102	26
		Desynchronized (PfPET>2)	116	67
	Köppen-Geiger classification	Tropical	11	2
		Arid	19	14
		Temperate	123	52
		Continental	65	25
Watershed property	Water retention index	I_R≤0.5	176	81
		I_R>0.5	42	12
	Watershed size	Small (<1000km^2)	197	59
		Large (>1000km^2)	21	34
	Forest type	Broadleaf	102	60
		Coniferous	105	22
		Mixed	11	11
	Soil type	Acrisols	37	17
		Podzols	33	9
		Cambisols	31	23
		Lithosols	9	18

3. Results and discussion

3.1. *Hydrological sensitivities to deforestation and forestation*

Hydrological sensitivities to forestation are significantly larger than those associated with deforestation ($p<0.001$; Fig. 2).

This result answers our first question: hydrological sensitivities to forestation differ from and are significantly larger than those to deforestation (Fig. 2). 1% forest change caused by deforestation and forestation, on average, can result in a 0.91% and 1.24% change in annual streamflow, respectively. This finding is consistent with previous research evaluating the change magnitude of streamflow to forest cover change. Wang et al. (2020) revealed that absolute changes in streamflow after forestation are much larger than deforestation. Piao et al. (2007) found that deforestation can increase annual streamflow by 8 mm/yr worldwide, while Jackson et al. (2005) found forestation can decrease annual streamflow by 227 mm/yr, with some streams drying up.

The significant difference in HS_f between deforestation and forestation groups may be due to the following several factors. First, differences in forest management operations and their associated changes in ecosystem structure and functioning could contribute to the difference in HS_f. Forest harvesting activities can partially offset hydrological changes. For example, understory vegetation may be left on the site typically exhibiting competitive release (e.g., a rapid post-disturbance growth response) that may reduce increases in streamflow. Dead materials (e.g., woody debris) may be left on the site to mitigate the increases of surface flow (Coble et al., 2020), and soil infiltration ability and soil moisture would be maintained if soil disturbance is not considerable (Peña-Arancibia et al., 2019). For example, using brush mats can significantly reduce soil compaction in harvested sites (Ring et al., 2021). These activities can maintain streamflow. In contrast, forestation, particularly afforestation (55 of 93 cases in this study), often starts from bare land or converts other land-use types (e.g., agriculture, urban) into forests where initial forest cover is less or limited (Filoso et al., 2017). Therefore, forestation activities could dramatically alter initial conditions in vegetation (e.g., type, structure, and component) and soils (e.g., infiltration, soil moisture), leading to larger HS_f. In summary, even if both processes occur under the same climate, deforestation immediately alters forest structure and some hydrological

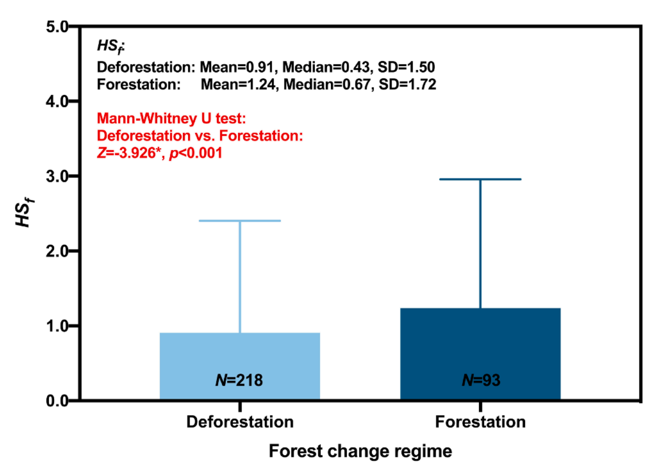

Fig. 2. A comparison of HS_f between deforestation and forestation groups with mean, median, standard deviation (SD), sample size (*N*), and the result of the Mann-Whitney *U* test (*denotes statistically significant with a *p*-value less than 0.10). The data shown are mean values with SD.

processes linked to it, but could maintain the functioning of the soil, while forestation changes the soil and forest structure over a long period of time, resulting in an ecosystem structure and functioning probably different from the original one.

Second, forestation programs often use non-native, fast-growing tree species in monocultural plantations, which causes more rapid changes in ET, and consequently, annual streamflow (Farley et al., 2005; Ferraz et al., 2021; Jackson et al., 2005; Rahmat et al., 2018). For example, 83% of forestation sites in our dataset are associated with non-native, fast-growing tree plantations. This forest composition might lead to larger HS_f. However, there are some cases where native trees are replanted, and natural regeneration of native species takes place, which may result in smaller HS_f than planting non-native, fast-growing tree species.

Third, the response time following deforestation and forestation might also contribute to differences in HS_f. Deforestation causes changes in streamflow immediately after tree removal, but rapid changes are diminished as forest recovery progresses (Brown et al., 2005; Moore et al., 2020). In comparison, streamflow responses to forestation are gradual and persist for an extended period as the site reaches a new equilibrium (Farley et al., 2005; Zhang et al., 2015). While the magnitude of hydrological change can be variable (Filoso et al., 2017), a consistent decreasing trend in hydrological response may continue for decades after forestation (Feng et al., 2016). The more extended hydrological responses from forestation suggest the hydrological sensitivities would be more significant.

Finally, the difference in climate conditions between deforestation and forestation groups might also contribute to larger HS_f in the forestation group. Forest harvesting occurs typically in areas where trees have matured, while forestation activities are implemented anywhere. Our analysis shows lower DI values (mean DI=0.88, energy-limited) in the deforestation group and higher DI values (mean DI=1.08, water-limited) in the forestation group. As Fig. 3 in the next section illustrated, hydrological sensitivities in water-limited systems are significantly larger than in energy-limited systems.

The above reasons explain the difference in HS_f between deforestation and forestation groups, suggesting that hydrological responses to deforestation and forestation are not simply reversible. Instead, deforestation and forestation activities modify forest ecosystem structures and functioning (vegetation and soil) differently in time and space (Ferraz et al., 2020), causing fundamental changes to HS_f. However, different management activities may result in variable hydrological responses, suggesting that more future research is needed.

3.2. HS_f and climate

Hydrological sensitivities to deforestation and forestation in water-limited watersheds are both significantly larger than those in energy-limited watersheds ($p<0.001$) (Fig. 3). For the deforestation group, the mean value of HS_f is 0.66 for energy-limited watersheds and 1.60 for water-limited watersheds, while for the forestation group, it is 0.61 for energy-limited watersheds and 2.11 for water-limited watersheds.

This result answers our second question: climate controls HS_f to both deforestation and forestation, with significantly larger HS_f observed in water-limited watersheds than in energy-limited ones (Fig. 3). There is also a positive relationship between DI and HS_f (Table 2).

This finding is in line with global and regional assessments (Luo et al., 2020; Peña-Arancibia et al., 2019; Zhang et al., 2017; Zhou et al., 2015). Climate directly affects water and energy inputs in watersheds and indirectly affects forest distribution, growth, and phenology (Bearup et al., 2014; Villarini and Wasko, 2021; Yang et al., 2021). For example, under water-limited conditions, forest distribution, growth, and succession are more water-dependent (Bai et al., 2020). In turn, forest characteristics can affect water flux (Asbjornsen et al., 2011). If forests are changed in water-limited systems, the close linkages between forests and water processes are disrupted, and more significant changes in hydrological processes are expected. In addition, ET/P ratios in water-limited systems are much larger than those in energy-limited systems. For example, the average ET/P ratio of two energy-limited subtropical watersheds in the Poyang Lake Basin is 0.54. In comparison, the ET/P ratio is 0.92 for four water-limited semi-arid watersheds in the Loess Plateau, China (Hou et al., 2021). Thus, a change in forest cover could lead to larger changes in annual ET and annual streamflow, and consequently, HS_f in water-limited systems.

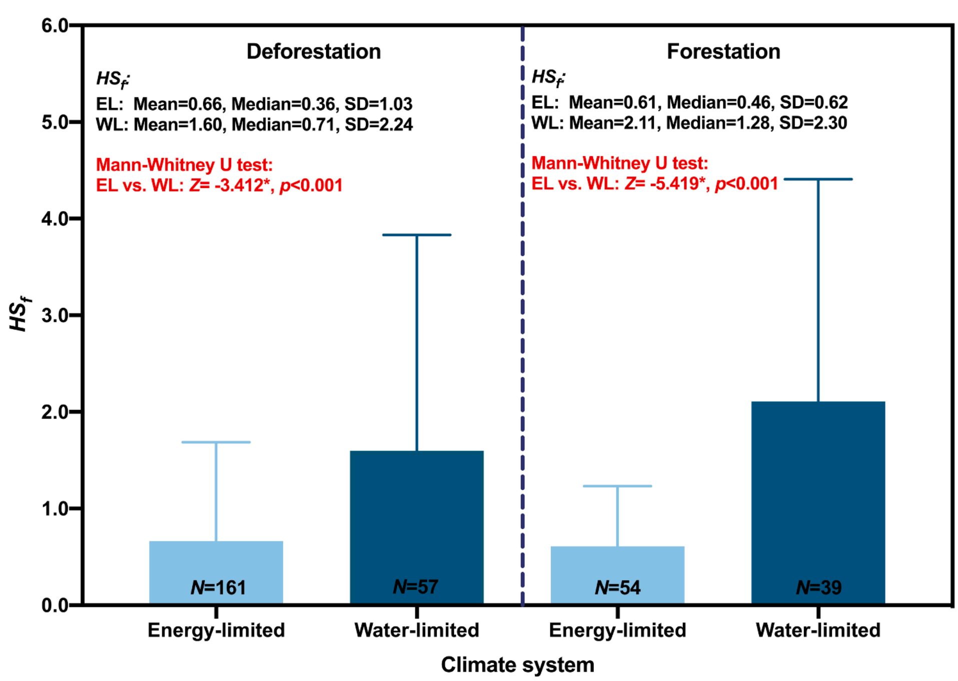

Fig. 3. Comparisons of HS_f between energy-limited (EL) and water-limited (WL) systems with mean, median, standard deviation (SD), sample size (N), and the results of Mann-Whitney U tests (*denotes statistically significant with a p-value less than 0.10) in deforestation and forestation groups. The data shown are mean values with SD.

Fig. 4 estimates HS_f to deforestation and forestation among Köppen-Geiger climate classifications. We tested differences in HS_f between tropical and other climate zones in the deforestation group but did not involve tropical watersheds in the forestation group because there are only two samples in this climate zone undergoing forestation (Fig. 4). We failed to detect any significant differences in HS_f between watersheds in the deforestation group in the Köppen-Geiger tropical, arid, temperate, and continental climate zones. Nevertheless, 1% deforestation, on average, can cause 0.78%, 1.61%, 0.87%, and 0.73% changes in annual streamflow in tropical, arid, temperate, and continental watersheds, respectively. In contrast, we detected significant differences in HS_f between watersheds in the forestation group. Hydrological sensitivities in the arid (mean HS_f=1.93) and continental (mean HS_f=1.42) zones are significantly larger than in the temperate zone (mean HS_f=1.00). Nevertheless, the largest HS_f in both deforestation and forestation groups is in the arid zone, consistent with the comparison between energy-limited and water-limited systems.

Fig. 5a shows differences in HS_f to deforestation and forestation between synchronized (PfPET$\leq$ 2) and desynchronized (PfPET$>$ 2) watersheds. In the deforestation group, there is no significant difference in HS_f between synchronized (mean HS_f=0.84) and desynchronized watersheds (mean HS_f=0.97; p>0.10). In contrast, in the forestation group, hydrological sensitivities are significantly larger in synchronized watersheds (mean HS_f=1.42) than in desynchronized watersheds (mean HS_f=0.76; p<0.10). Overall, the interval phase (PfPET) has limited impacts on HS_f in the deforestation group (r=0.008 and p=0.904; Fig. 5b), but there is a significant negative relationship between PfPET and HS_f in the forestation group (r=−0.204 and p=0.050; Fig. 5c). These results suggest that hydrological responses to forestation are more sensitive in more synchronized watersheds.

Fig. 5b and 5c show that hydrological sensitivities decrease significantly with increasing PfPET in synchronized watersheds in both deforestation and forestation groups (the purple dashed lines in Fig. 5b-5c). In contrast, hydrological sensitivities increase significantly with increasing PfPET in desynchronized watersheds in the forestation group alone (the orange dashed line in Fig. 5c). These results highlight significant and dynamic relationships between HS_f and synchronicity of PfPET, with the largest HS_f occurring at PfPET=0 (i.e., perfect matching of synchronicity between monthly energy (PET) and water (P)) in both deforestation and forestation groups.

Generally, synchronized watersheds have significantly larger HS_f than desynchronized watersheds in the forestation group, and there are negative relationships between HS_f and PfPET for both deforestation and forestation groups with the largest sensitivity at PfPET=0. This is because the intra-annual synchronicity of P and PET plays an essential role in tree growth, and thus HS_f. Water and energy availability are better matched in more synchronized watersheds, promoting and increasing tree growth and ET and leading to more significant hydrological responses when forests are altered. Berghuijs et al. (2014) found that the synchronicity of P and PET can significantly affect inter-annual precipitation partitioning by increasing annual ET and reducing annual streamflow, suggesting a smaller amount of annual streamflow in synchronized watersheds than in desynchronized watersheds.

There is a positive relationship between HS_f and PfPET in desynchronized watersheds in the forestation group (the orange dashed line in Fig. 5c). A closer look at this positive relationship indicates that the large sensitivity at PfPET=6 might cause the positive relationship. When monthly P and PET are the least matched (at PfPET=6) in desynchronized watersheds, these watersheds are likely energy-limited during winter and water-limited during summer (Feng et al., 2019). The increase in forest cover causes the most severe soil moisture deficit in the growing season, especially during the summer period, which in turn causes the large HS_f.

3.3. HS_f and watershed properties

Topography delineates flow path, water movement, water residence time, and water storage capacity, which is partly associated with water retention capacity (Li et al., 2018; Teutschbein et al., 2018; Zhang and Wei, 2021; Zhou et al., 2015). Our results show that watersheds with lower values of water retention index (i.e., I_R≤0.5) have significantly larger hydrological sensitivities to forestation. Fig. 6 shows that there is no significant difference in HS_f between low I_R (i.e., I_R≤0.5) and high I_R (i.e., I_R>0.5) watersheds in the deforestation group (p>0.10). In contrast, hydrological sensitivities are significantly larger in watersheds with low I_R than those with high I_R (i.e., p<0.10) for the forestation group. On average, 1% forestation results in 1.29% and 0.88% changes in streamflow in low and high I_R watersheds, respectively. This

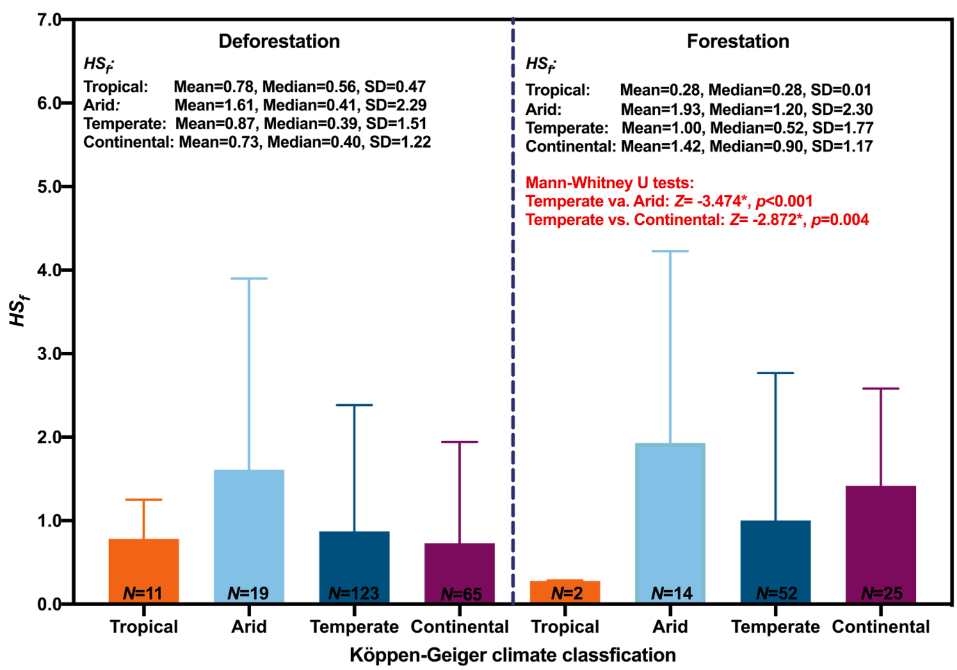

Fig. 4. Comparisons of HS_f between Köppen-Geiger climate classes with mean, median, standard deviation (SD), sample size (N), and the results of significant Mann-Whitney U tests (*denotes statistically significant with a p-value less than 0.10) in deforestation and forestation groups. The data shown are mean values with SD.

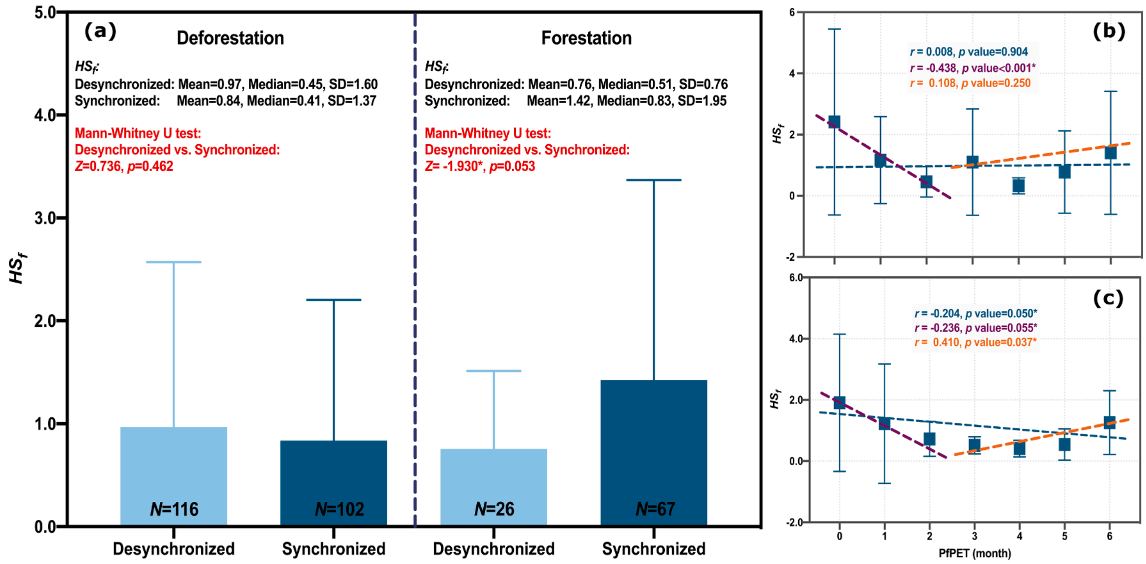

Fig. 5. (**a**) Comparisons of HS_f between watersheds with synchronized and desynchronized monthly P and PET timing in deforestation and forestation groups. The data shown are mean values with SD. In addition, the mean, median, standard deviation (SD), sample size (N), and the results of Mann-Whitney U tests (*denotes statistically significant with a p-value less than 0.10) are presented; **(b)** The relationship between HS_f and interval phase between peak monthly precipitation and peak monthly potential evapotranspiration (PfPET) in the deforestation group; and **(c)** The relationship between HS_f and PfPET in the forestation group.

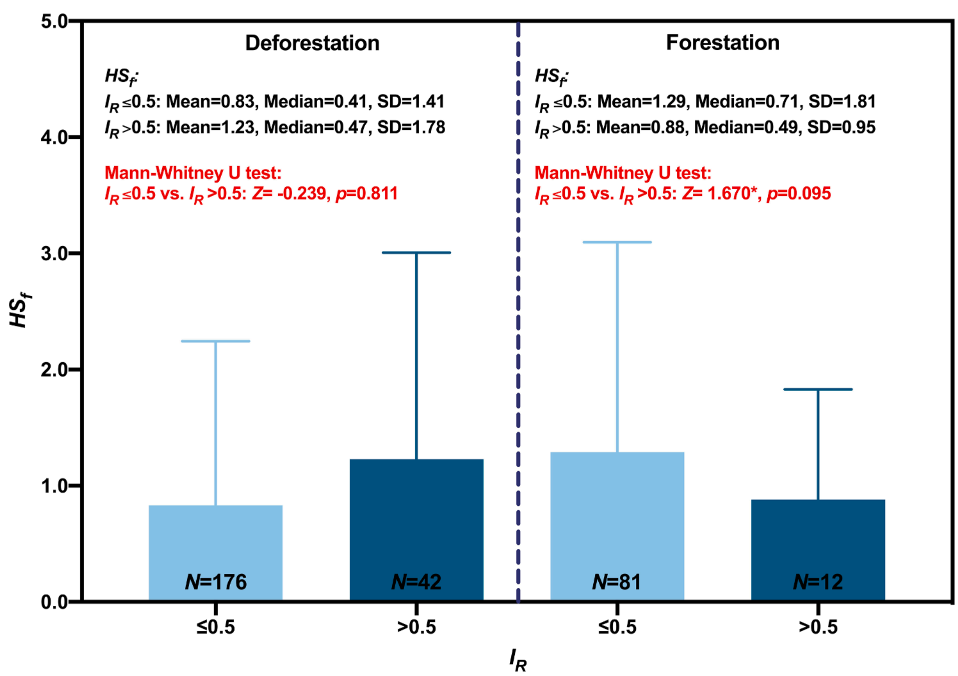

Fig. 6. Comparisons of HS_f between watersheds with low ($I_R \leq 0.5$) and high ($I_R > 0.5$) water retention index (I_R) in deforestation and forestation groups. The data shown are mean values with SD. In addition, the mean, median, standard deviation (SD), sample size (N), and the results of Mann-Whitney U tests (*denotes statistically significant with a p-value less than 0.10) are presented.

difference is likely because watersheds with lower water retention capacities tend to have poor water storage for soil infiltration and groundwater recharge (López-Ramírez et al., 2020). As a result, these watersheds are more likely to "flush" with quicker hydrological responses to forest cover change (e.g., conversion to a plantation). The non-significant difference in HS_f between low and high I_R watersheds in the deforestation group might be because the classification criterion (I_R=0.5) is subjective. However, the MLR models clearly suggest that hydrological sensitivities decrease with rising I_R for deforestation and forestation groups (Table 2). These results suggest that watersheds with greater topography-related water retention capacities are more resistant or less sensitive to forest cover change.

Our results show that hydrological sensitivities in large watersheds (mean HS_f=2.02) are significantly greater than in smaller watersheds (mean HS_f=0.78) ($p<0.001$) in the forestation group (Fig. 7). Therefore, large watersheds (i.e., watershed size>1000 km^2) are more sensitive to forestation than small watersheds (i.e., watershed size<1000 km^2). However, in the deforestation group, there is no significant difference in HS_f between small and large watersheds ($p>0.10$), with their mean values being 0.88 and 1.16 in small and large watersheds, respectively. This result contradicts commonly held perceptions that larger watersheds have larger hydrological buffering capacities and are therefore

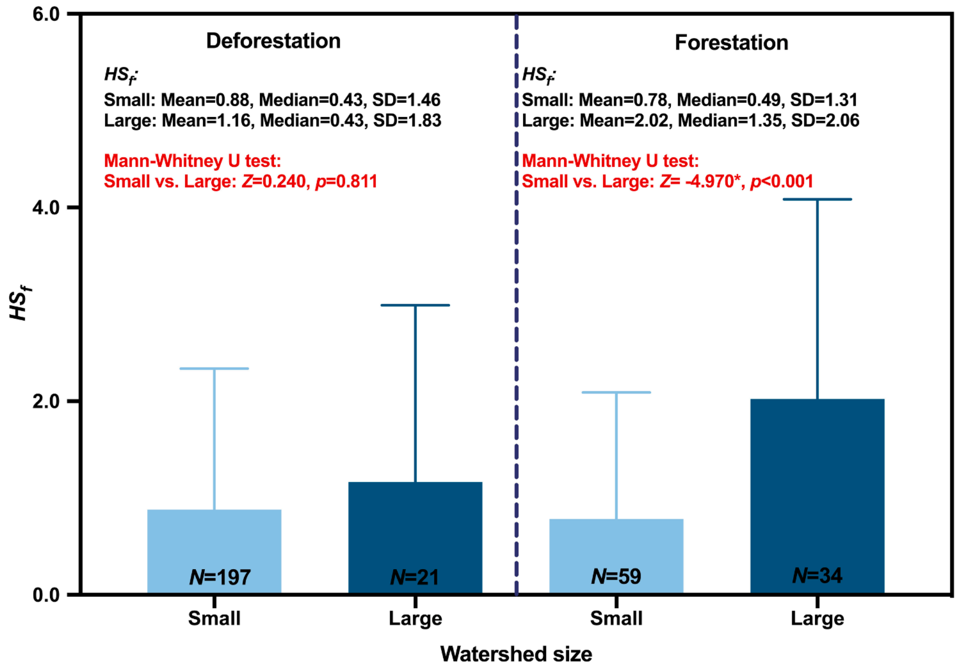

Fig. 7. Comparisons of HS_f between small (<1000 km^2) and large (>1000 km^2) watersheds with mean, median, standard deviation (SD), sample size (N), and the results of Mann-Whitney U tests (*denotes statistically significant with a p-value less than 0.10) in deforestation and forestation groups. The data shown are mean values with SD.

less sensitive to forest disturbance or forest change (Blöschl et al., 2007; Filoso et al., 2017; Huff et al., 2000; Zhou et al., 2015). Nevertheless, our result is consistent with Li (2018), who detected an amplified effect on annual streamflow changes caused by cumulative forest disturbance with increasing watershed size in the southern interior of British Columbia.

The following reasons could explain the larger hydrological sensitivities to forestation in large watersheds. First, large hydrological buffering capacities in large watersheds are commonly related to the total magnitudes of peak or low flow (Eaton et al., 2002). However, the larger hydrological sensitivities in large watersheds, which are the focus of this study, are related to variations in the total magnitude of annual streamflow caused by forest cover change. HS_f can be amplified with increasing watershed size due to interactions and possible feedback among various processes (Li, 2018). Second, the selection of study watersheds might also contribute to this contrasting result. Researchers commonly avoid complicated landforms (e.g., large lakes or wetlands) when selecting watersheds to assess hydrological responses to forest change. Therefore, these complicated landforms with large hydrological buffering capacities might not be well represented in their research

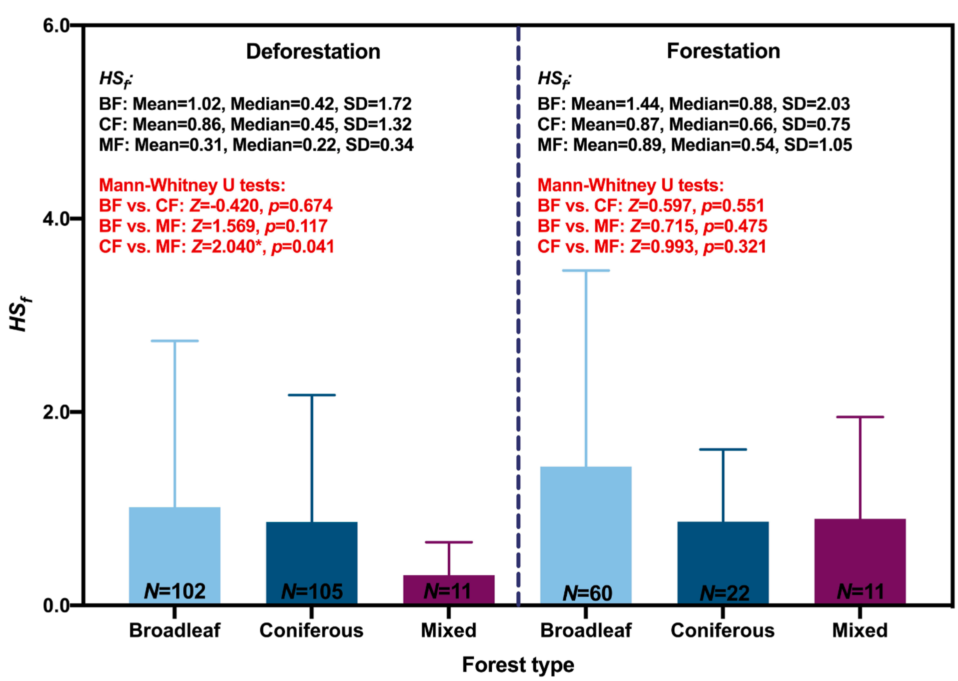

Fig. 8. Comparisons of HS_f between broadleaf (BF), coniferous (CF), and mixed forest-dominated (MF) watersheds with mean, median, standard deviation (SD), sample size (N), and the results of Mann-Whitney U tests (*denotes statistically significant with a p-value less than 0.10) in deforestation and forestation groups. The data shown are mean values with SD.

design. For this reason, larger watersheds tend to have greater stream power through the greater contributing area assuming a similar slope. This factor could probably contribute to larger hydrological sensitivities to forest change. Finally, this study used 1000 km^2 as a dividing line to compare small and large watersheds. Although this definition has often been used (England et al., 2007; Singh, 1995; Wei et al., 2013; Wei and Zhang, 2010), it is a subjective threshold that could introduce uncertainty. Therefore, we applied another watershed size threshold to classify small and large watersheds (Figure S2) and found the watershed size threshold does not affect the result.

Our results show that mixed forest-dominated watersheds have significantly smaller HS_f to deforestation than coniferous forest-dominated watersheds (Fig. 8). In contrast, differences in HS_f between coniferous and broadleaf types (BF vs CF) and between broadleaf and mixed types (BF vs MF) are not significant in the deforestation group (Fig. 8). Mean hydrological sensitivities to deforestation are 1.02, 0.86, and 0.31 in broadleaf, coniferous, and mixed forest-dominated watersheds, respectively. In comparison, hydrological sensitivities to forestation in broadleaf, coniferous, and mixed forest-dominated watersheds are not significantly different. These results demonstrate that forest types could have an important role in HS_f to deforestation.

The significantly smaller hydrological sensitivities to deforestation in mixed forest-dominated watersheds suggest that these watersheds are more hydrologically resistant to deforestation than coniferous forest-dominated watersheds. The diversity of tree species in coniferous forests is relatively small. In contrast, the diversity of tree species is large with complex, multi-layered stand structures in mixed forest-dominated watersheds (Ferraz et al., 2013). These structural and functional traits of mixed forest-dominated watersheds are expected to play a positive role in buffering hydrological responses to deforestation, and consequently reducing HS_f. Similar results were also found in other studies (Creed et al., 2014; Ellison et al., 2017; van Dijk et al., 2012; Zhang et al., 2017; Zhou et al., 2015). In the forestation group, recovery of hydrological functioning and services always takes much longer and may not be fully realized (Liu et al., 2016; Senf et al., 2019). As a result, the difference in HS_f to forestation among coniferous, broadleaf, and mixed forest-dominated watersheds is likely less pronounced.

Finally, our results show that the dominant soil type in watersheds influences HS_f. Fig. 9 shows hydrological sensitivities to deforestation and forestation in Acrisols-, Podzols-, Cambisols-, and Lithosols-dominated watersheds. In the deforestation group, Acrisols-dominated watersheds have significantly larger HS_f values than those in Podzols- and Cambisols-dominated watersheds, and Lithosols-dominated watersheds have significantly larger HS_f values than Podzols-dominated watersheds. Mean hydrological sensitivities are 0.92 for Acrisols-dominated, 0.32 for Podzols-dominated, 0.27 for Cambisols-dominated, and 0.97 for Lithosols-dominated watersheds, respectively. In the forestation group, significantly smaller hydrological sensitivities are observed in Acrisols-dominated watersheds with an average value of 0.38. Mean hydrological sensitivities to forestation in Podzols-, Cambisols-, and Lithosols-dominated watersheds are 0.69, 1.57, and 2.24, respectively.

Soil types affect soil moisture, groundwater recharge, discharge, and the interaction between surface and subsurface processes, influencing HS_f (Schoonover and Crim, 2015). For both deforestation and forestation groups, Lithosols have the largest HS_f, while Podzols have relatively smaller HS_f (Fig. 9). Lithosols typically have shallow soil layers less than 10 cm in thickness and low soil water holding capacities as they are generally located on steep slopes (Nachtergaele, 2017). Once forest change activities are implemented in Lithosols-dominated watersheds, changes in soil moisture and other hydrological processes are expected to be dramatic and quick, resulting in more severe hydrological responses to forest change. In contrast, Podzols are the typical soils of coniferous forests with coarse textures. Podzols are acidic soils with low fertility (Sanborn et al., 2011). As a result, forest change-related tree growth rates are often low in Podzols-dominated watersheds, and the related hydrological responses are less sensitive.

Our results (Figs. 6–9) answer the third question: water retention capacity and watershed size contribute to HS_f to forestation (Figs. 6–7), forest type affects HS_f to deforestation (Fig. 8), and soil type modulates HS_f to both deforestation and forestation (Fig. 9).

3.4. HS_f and the relative importance of contributing drivers

The multiple linear regression (MLR) model for the deforestation group shows that DI is positively related to HS_f, while LAI (representing

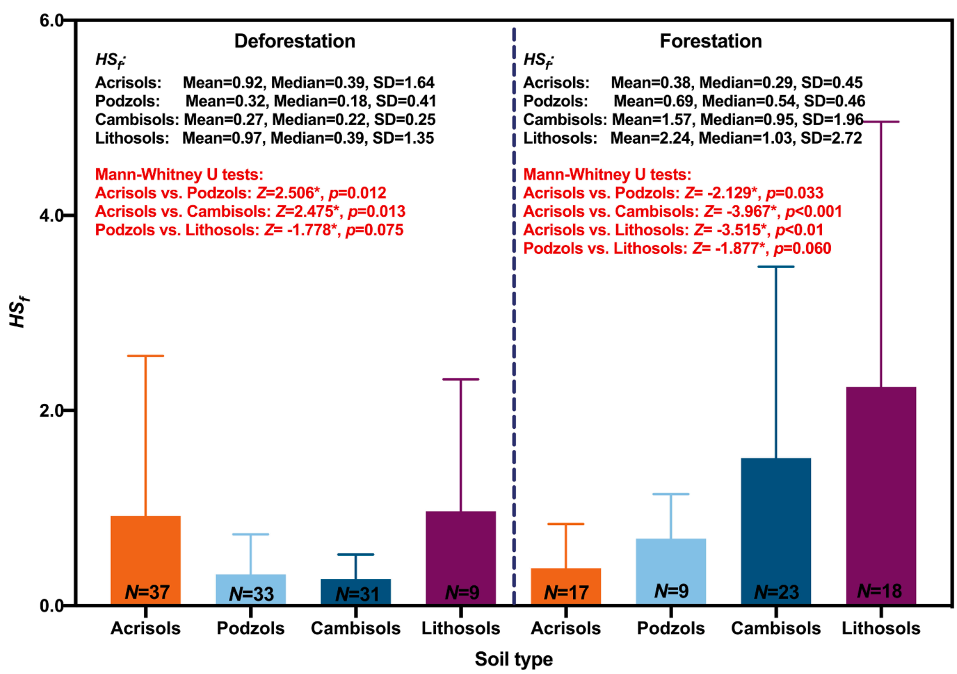

Fig. 9. Comparisons of HS_f between Acrisols-, Podzols-, Cambisols-, and Lithosols-dominated watersheds with mean, median, standard deviation (SD), sample size (N), and the results of significant Mann-Whitney U tests (*denotes statistically significant with a p-value less than 0.10) in deforestation and forestation groups. The data shown are mean values with SD.

Table 2
Results of multiple linear regression (MLR) models between HS_f and contributing drivers.

Forest change group	Constant	Integrated index	Coefficient	Standardized beta coefficients	R^2	p-value
Deforestation	0.127	DI	0.997	0.368	0.15	<0.001
		LAI	−0.030	−0.029		
		I_R	−0.013	−0.007		
Forestation	0.218	DI	1.516	0.453	0.21	<0.001
		LAI	0.059	0.045		
		I_R	−2.315	−0.180		

Note. DI, LAI, and I_R denote dryness index, leaf area index, and water retention index, respectively.

the site condition) and I_R (representing the topography-related water retention capacity) are negatively related to HS_f (Table 2). For the forestation group, MLR suggests hydrological sensitivities increase with DI while decrease with I_R. However, there is a positive relationship between HS_f and LAI (Table 2). Fig. 10 exhibits the averaged relative importance of climate and watershed properties to HS_f, estimated by the machine learning based GBM and MLR models (GBM model parameters and performance, and the relative importance of the two methods are presented in Section S4 in the Supplementary Materials). In the deforestation group, the relative importance of climate in estimating HS_f is 75.5%. In comparison, the relative importance of watershed properties is 24.5% in the deforestation group (with I_R contributing 13.7% and LAI contributing 10.8%) (Fig. 10). In comparison, in the forestation group, the relative importance of climate in estimating HS_f is 60.6 %, with I_R contributing 27.2% and LAI contributing 12.2% (Fig. 10). Climate is the most important contributing driver of HS_f in deforestation and forestation groups, while I_R and LAI are secondary drivers.

This result answers the fourth question: climate is the primary driver of hydrological sensitivities to deforestation and forestation, while watershed properties play a secondary role. Previous studies have demonstrated the roles of climate and watershed properties in terms of the total magnitude of hydrological variables. For example, Zhang et al. (2004) showed that ET is mainly driven by climate (i.e., P and PET) with a lower contribution of watershed properties. From the theoretical Budyko framework analysis, Zhou et al. (2015) suggested that climate dominates hydrological responses when the watershed characteristic parameter (m in Fu's Budyko framework) is greater than 2. Conversely, watershed properties dominate hydrological responses when the watershed characteristic parameter is less than 2. Similarly, Liu et al. (2019) showed that the relative contribution of precipitation to hydrological response is more significant than that of watershed properties at the global scale. However, the relative importance of climate and watershed properties to hydrological responses, such as HS_f, is rarely examined. Our result suggests that climate variability can alter hydrological processes and make a large contribution to HS_f. It also suggests that a framework that considers climatic aspects, watershed properties, and forest management type, extent, and intensity as determinants of the observed effects is required to understand hydrological effects of forest change (see also Ferraz et al., 2019).

LAI plays different roles in HS_f in deforestation versus forestation (Table 2). While LAI positively influences on HS_f in the forestation group, it negatively influences HS_f in the deforestation group. LAI, determined by site conditions (e.g., land cover types, forest types, and tree species), largely controls ET and gross photosynthesis (Potithep et al., 2013; Reichenau et al., 2016). Therefore, higher LAI values indicate more favorable site conditions for forest growth and vegetation regeneration. The positive role of LAI in terms of HS_f to forestation may reflect the fast growth of non-native tree species often used in forestation activities that lead to the large change in hydrological processes in these watersheds (Chi et al., 2015). In contrast, the negative role of LAI in terms of HS_f to deforestation may reflect the fast regeneration rates of native tree species that may mitigate streamflow increments after tree removal (Brown et al., 2005).

4. Uncertainties

There are some uncertainties in this study. First, we selected data from existing publications using modeling approaches across the globe. However, the representation of tropical watersheds is relatively low. According to the Köppen-Geiger classification, only 13 study watersheds are situated in the tropics. Also, we have a relatively small sample size of large watersheds (e.g., 55 of our study watersheds are above 1000 km^2). The imbalanced sample sizes among classes might cause uncertainty in our statistical analyses. Second, inconsistent quantification methods and response periods among the selected studies prevent consistent comparisons. For example, hydrological sensitivities might decrease with increasing response periods in the deforestation group while increase with increasing response periods in the forestation group. Since streamflow responses in each year were not available from selected studies, inconsistent response periods might also cause uncertainty. To understand hydrological sensitivities to deforestation and forestation, it is impossible to capture the period from reforestation or afforestation to a mature stand since some types of forests take a very long time to reach maturity and such periods would vary with climate and forest types, e.g., tropical forests may recover in 10 years, while boreal forests could take >100 years. In this study, we assumed that the case studies are based on data collected during periods of high impact on hydrological sensitivities. Third, the proposed water retention index (I_R) is only based on watershed slope and elevation difference (i.e., basin relief), which might not entirely reflect water retention capacity. Other watershed property indices are related to the water retention capacity (Li et al., 2018; Scown et al., 2015), but the selected studies' data to generate these indices were unavailable. Fourth, there are likely other factors that were not considered in this study that contribute HS_f. Finally, forest cover change is not a perfect indicator as it may not capture variations in terms of forest state, distribution, and canopy condition since these variations at the finer scale (i.e., stand-level) can also affect hydrological processes. However, forest cover at the watershed scale is a suitable indicator to reflect forest dynamics and their effects on hydrology at this scale.

5. Implications for forest and watershed management

The key findings on hydrological sensitivities to deforestation and forestation and their implications for forest management are summarized in Fig. 11. First, hydrological sensitivities to forestation differ from and are significantly larger than those to deforestation, suggesting that hydrological responses to deforestation and forestation are not simply reversible. Second, hydrological sensitivities to forest change are larger in water-limited or arid environments than in energy-limited or humid environments. This indicates that forest management activities must be customized to different climatic regions. For example, forestation operations could aggravate water shortages in arid watersheds (Feng et al., 2016), especially with synchronized energy demand and water supply. Third, watershed properties (e.g., water retention capacity, site condition, and forest type) are crucial drivers of HS_f. For example, watersheds with low water retention capacity have large HS_f because of limited soil infiltration and groundwater recharge opportunities. Therefore, forest management activities must respect the limitations imposed by the

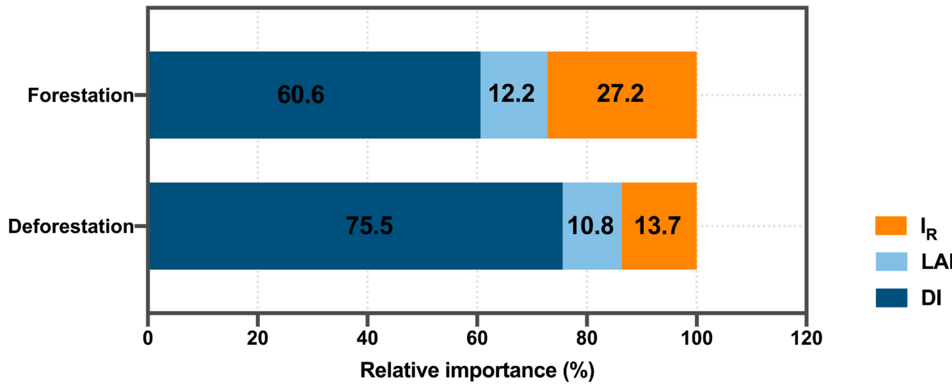

Fig. 10. The relative importance of DI, LAI, and I_R to HS_f in deforestation and forestation groups.

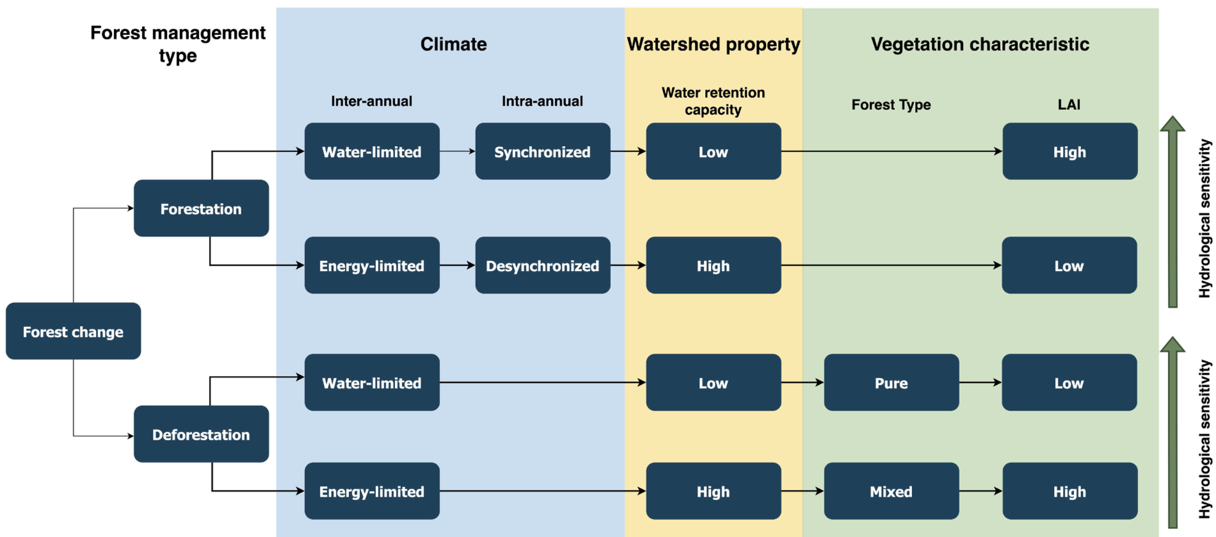

Fig. 11. A framework for managing hydrological sensitivities to deforestation or forestation.

climate and watershed characteristics. Adapting the type of tree species, the intensity of forest management, and the scale of conversions between forest cover and other vegetation types is essential to avoid adverse hydrological effects. Finally, our findings demonstrate that certain forest management activities such as deforestation should be avoided in areas with a potential for large hydrological sensitivities.

6. Conclusions

This study critically examined and compared hydrological sensitivities (HS_f) to deforestation and forestation, and their influencing factors across multiple watershed classes. We conclude that forestation results in larger HS_f than deforestation. Climate is the primary driver for influencing HS_f. For both forest management groups, arid watersheds have larger HS_f than humid watersheds. Hydrological sensitivities are larger in forestation watersheds with better matchings between water and energy at the monthly scale. Watershed properties such as site condition, water retention capacity, forest type, and soil type also contribute to HS_f. We suggest that both climate and watershed properties, including forest cover change, must be included in assessing hydrological sensitivities. Forest management decisions should account for variations in hydrological sensitivities for protecting hydrological functions and minimizing water-related environmental risks.

CRediT authorship contribution statement

Yiping Hou: Methodology, Data curation, Formal analysis, Writing – original draft. **Xiaohua Wei:** Conceptualization, Writing – review & editing, Supervision. **Mingfang Zhang:** Conceptualization, Writing – review & editing. **Irena F. Creed:** Writing – review & editing. **Steven G. McNulty:** Writing – review & editing. **Silvio F.B. Ferraz:** Writing – review & editing.

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.

Data availability

Data will be made available on request.

Acknowledgments

The authors thank the China Scholarship Council (CSC) for sponsoring Yiping Hou. We are grateful to Dr. Andrés Iroumé from Facultad de Ciencias Forestales y Recursos Naturales, Universidad Austral de Chile for sharing Geographic Information System (GIS) data, and Min Yan from Sichuan Agricultural University, China for sharing soil data set. We are also grateful to thank the editor and anonymous reviewers for their constructive suggestions and comments on this paper. This research was supported by Natural Sciences and Engineering Research Council of Canada, Discovery Grants Program (No. RGPIN-2021-02628).

Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.foreco.2022.120718.

References

Aryal, Y., Zhu, J., 2020. Effect of watershed disturbance on seasonal hydrological drought: An improved double mass curve (IDMC) technique. J. Hydrol. 585, 124746 https://doi.org/10.1016/j.jhydrol.2020.124746.

Asbjornsen, H., et al., 2011. Ecohydrological advances and applications in plant–water relations research: a review. J. Plant Ecol. 4 (1–2), 3–22. https://doi.org/10.1093/jpe/rtr005.

Bai, P., Liu, X., Zhang, Y., Liu, C., 2020. Assessing the impacts of vegetation greenness change on evapotranspiration and water yield in China. e2019WR027019 Water Resour. Res. 56 (10). https://doi.org/10.1029/2019WR027019.

Bearup, L.A., Maxwell, R.M., Clow, D.W., McCray, J.E., 2014. Hydrological effects of forest transpiration loss in bark beetle-impacted watersheds. Nat. Clim. Change 4 (6), 481–486. https://doi.org/10.1038/nclimate2198.

Berghuijs, W.R., Sivapalan, M., Woods, R.A., Savenije, H.H.G., 2014. Patterns of similarity of seasonal water balances: A window into streamflow variability over a range of time scales. Water Resour. Res. 50 (7), 5638–5661. https://doi.org/10.1002/2014WR015692.

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 (10), 8475–8486. https://doi.org/10.1002/2017WR021593.

Berghuijs, W.R., Woods, R.A., 2016. A simple framework to quantitatively describe monthly precipitation and temperature climatology. Int. J. Climatol. 36 (9), 3161–3174. https://doi.org/10.1002/joc.4544.

Blöschl, G., et al., 2007. At what scales do climate variability and land cover change impact on flooding and low flows? Hydrol. Process. 21 (9), 1241–1247. https://doi.org/10.1002/hyp.6669.

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 (1–4), 28–61. https://doi.org/10.1016/j.jhydrol.2004.12.010.

Chen, Z., Wang, W., Woods, R.A., Shao, Q., 2021. Hydrological effects of change in vegetation components across global catchments. J. Hydrol. 595, 125775 https://doi.org/10.1016/j.jhydrol.2020.125775.

Chi, X., et al., 2015. Effects of size, neighbors, and site condition on tree growth in a subtropical evergreen and deciduous broad-leaved mixed forest. China. Ecol. Evol. 5 (22), 5149–5161. https://doi.org/10.1002/ece3.1665.

Clerici, N., Cote-Navarro, F., Escobedo, F.J., Rubiano, K., Villegas, J.C., 2019. Spatio-temporal and cumulative effects of land use-land cover and climate change on two ecosystem services in the Colombian Andes. Sci. Total Environ. 685, 1181–1192. https://doi.org/10.1016/j.scitotenv.2019.06.275.

Coble, A.A., et al., 2020. Long-term hydrological response to forest harvest during seasonal low flow: Potential implications for current forest practices. Sci. Total Environ. 730, 138926 https://doi.org/10.1016/j.scitotenv.2020.138926.

Creed, I.F., et al., 2014. Changing forest water yields in response to climate warming: results from long-term experimental watershed sites across North America. Global Change Biol. 20 (10), 3191–3208. https://doi.org/10.1111/gcb.12615.

Creed, I.F., et al., 2019. Managing forests for both downstream and downwind water. Front. For. Glob. Change 2 (64). https://doi.org/10.3389/ffgc.2019.00064.

Creed, I., F., van Noordwijk, M., 2018. Forest and Water on a Changing Planet: Vulnerability, Adaptation and Governance Opportunities. International Union of Forest Research Organizations (IUFRO), Vienna, Austria.

Creed, I.F., Weber, M., Accatino, F., Kreutzweiser, D.P., 2016. Managing Forests for Water in the Anthropocene—The Best Kept Secret Services of Forest Ecosystems. Forests 7 (12), 60. https://doi.org/10.3390/f7030060.

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 (2), 983–995. https://doi.org/10.5194/hess-11-983-2007.

Eaton, B., Church, M., Ham, D., 2002. Scaling and regionalization of flood flows in British Columbia, Canada. Hydrol. Process. 16 (16), 3245–3263. https://doi.org/10.1002/hyp.1100.

Ellison, D., et al., 2017. Trees, forests and water: Cool insights for a hot world. Global Environ. Chang. 43, 51–61. https://doi.org/10.1016/j.gloenvcha.2017.01.002.

England, J.F., Velleux, M.L., Julien, P.Y., 2007. Two-dimensional simulations of extreme floods on a large watershed. J. Hydrol. 347 (1), 229–241. https://doi.org/10.1016/j.jhydrol.2007.09.034.

FAO, 2020. Global Forest Resources Assessment 2020: Main report, Rome, Itlay, 184 pp. https://doi.org/10.4060/ca9825en.

Farley, K.A., Jobbagy, E.G., Jackson, R.B., 2005. Effects of afforestation on water yield: a global synthesis with implications for policy. Global Change Biol. 11 (10), 1565–1576. https://doi.org/10.1111/j.1365-2486.2005.01011.x.

Feng, X., et al., 2016. Revegetation in China's Loess Plateau is approaching sustainable water resource limits. Nat. Clim. Change 6 (11), 1019. https://doi.org/10.1038/NCLIMATE3092.

Feng, X., Thompson, S.E., Woods, R., Porporato, A., 2019. Quantifying asynchronicity of precipitation and potential evapotranspiration in Mediterranean climates. Geophys. Res. Lett. 46 (24), 14692–14701. https://doi.org/10.1029/2019GL085653.

Ferraz, S.F.B., et al., 2021. How do management alternatives of fast-growing forests affect water quantity and quality in southeastern Brazil? Insights from a paired catchment experiment. Hydrol. Process. 35 (9), e14317.

Ferraz, S.F.B., Rodrigues, C.B., Garcia, L.G., Alvares, C.A., Lima, W.d.P., 2019. Effects of Eucalyptus plantations on streamflow in Brazil: Moving beyond the water use debate. Forest Ecol. Manag. 453, 117571 https://doi.org/10.1016/j.foreco.2019.117571.

Ferraz, S.F.B., Brancalion, P.H.S., Guillemot, J., Meli, P., 2020. On the need to differentiate the temporal trajectories of ecosystem structure and functions in restoration programs. Trop. Conserv. Sci. 13, 1–6. https://doi.org/10.1177/1940082920910314.

Ferraz, S.F.B., Lima, W.d.P., Rodrigues, C.B., 2013. Managing forest plantation landscapes for water conservation. Forest Ecol. Manag. 301, 58–66. https://doi.org/10.1016/j.foreco.2012.10.015.

Filoso, S., Bezerra, M.O., Weiss, K.C.B., Palmer, M.A., 2017. Impacts of forest restoration on water yield: A systematic review. PLoS One 12 (8), e0183210.

Giles-Hansen, K., Wei, X., Hou, Y., 2021. Dramatic increase in water use efficiency with cumulative forest disturbance at the large forested watershed scale. Carbon Bal. Manag. 16 (1), 6. https://doi.org/10.1186/s13021-021-00169-4.

Goeking, S.A., Tarboton, D.G., 2020. Forests and water yield: A synthesis of disturbance effects on streamflow and snowpack in western coniferous forests. J. For. 118 (2), 172–192. https://doi.org/10.1093/jofore/fvz069.

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 (13), 6985–6994. https://doi.org/10.1002/2016GL069763.

Hallema, D.W., et al., 2018. Burned forests impact water supplies. Nat. Commun. 9 (1), 1307. https://doi.org/10.1038/s41467-018-03735-6.

Hou, Y., et al., 2021. Quantification of ecohydrological sensitivities and their influencing factors at the seasonal scale. Hydrol. Earth Syst. Sci. 25 (3), 1447–1466. https://doi.org/10.5194/hess-25-1447-2021.

Huang, P., et al., 2020. The ecohydrological effects of climate and landscape interactions within the Budyko framework under non-steady state conditions. Catena 217, 106481. https://doi.org/10.1016/j.catena.2022.106481.

Huff, D.D., Hargrove, B., Tharp, M.L., Graham, R., 2000. Managing Forests for Water Yield: The Importance of Scale. J. For. 98 (12), 15–19. https://doi.org/10.1093/jof/98.12.15.

Jackson, R.B., et al., 2005. Trading water for carbon with biological carbon sequestration. Science 310 (5756), 1944. https://doi.org/10.1126/science.1119282.

Jencso, K.G., McGlynn, B.L., 2011. Hierarchical controls on runoff generation: Topographically driven hydrologic connectivity, geology, and vegetation. Water Resour. Res. 47 (11) https://doi.org/10.1029/2011WR010666.

Jin, Z., et al., 2021. Quantifying the impact of landscape changes on hydrological variables in the alpine and cold region using hydrological model and remote sensing data. Hydrol. Process. 35 (10), e14392.

Jones, J., et al., 2018. Forest landscape hydrology in a 'new normal' era of climate and land use change, Forest and water on a changing planet: Vulnerability, adaption and governance opportunities. A global assessment report. International Union of Forest Research Organizations (IUFRO), pp. 81–100.

Keenan, R.J., et al., 2015. Dynamics of global forest area: Results from the FAO Global Forest Resources Assessment 2015. Forest Ecol. Manag. 352, 9–20. https://doi.org/10.1016/j.foreco.2015.06.014.

Khairiah, R.N., Setiawan, Y., Prasetyo, L.B., Permatasari, P.A., 2017. Leaf Area Index (LAI) in different type of agroforestry systems based on hemispherical photographs in Cidanau Watershed. IOP Conf. Series: Earth Environ. Sci. 54, 012050 https://doi.org/10.1088/1755-1315/54/1/012050.

Kibria, K.N., Ahiablame, L., Hay, C., Djira, G., 2016. Streamflow Trends and Responses to Climate Variability and Land Cover Change in South Dakota. Hydrology 3 (1). https://doi.org/10.3390/hydrology3010002.

Li, Q., et al., 2017. Forest cover change and water yield in large forested watersheds: A global synthetic assessment. Ecohydrology 10 (4), e1838.

Li, Q., et al., 2018. Topography significantly influencing low flows in snow-dominated watersheds. Hydrol. Earth Syst. Sci. 22 (3), 1947–1956. https://doi.org/10.5194/hess-22-1947-2018.

Li, Q., 2018. The cumulative effects of forest disturbance on streamflow components and their scaling properties in nested watersheds of the southern interior of British Columbia, University of British Columbia, Kelowna, Canada. doi: https://open.library.ubc.ca/collections/ubctheses/24/items/1.0372358.

Liu, W., et al., 2015. How do climate and forest changes affect long-term streamflow dynamics? A case study in the upper reach of Poyang River basin. Ecohydrology 8, 46–57. https://doi.org/10.1002/eco.1486.

Liu, W., et al., 2016. Hydrological recovery in two large forested watersheds of southeastern China: the importance of watershed properties in determining hydrological responses to reforestation. Hydrol. Earth Syst. Sci. 20 (12), 4747–4756. https://doi.org/10.5194/hess-20-4747-2016.

Liu, J., et al., 2019. Global attribution of runoff variance across multiple timescales. J. Geophys. Res. Atmos. 124 (24), 13962–13974. https://doi.org/10.1029/2019JD030539.

Liu, N., et al., 2021. Forested lands dominate drinking water supply in the conterminous United States. Environ. Res. Lett. 16 (8), 084008 https://doi.org/10.1088/1748-9326/ac09b0.

López-Ramírez, S.M., et al., 2020. Land use change effects on catchment streamflow response in a humid tropical montane cloud forest region, central Veracruz, Mexico. Hydrol. Process. 34 (16), 3555–3570. https://doi.org/10.1002/hyp.13800.

Luo, Y., Yang, Y., Yang, D., Zhang, S., 2020. Quantifying the impact of vegetation changes on global terrestrial runoff using the Budyko framework. J. Hydrol. 590, 125389 https://doi.org/10.1016/j.jhydrol.2020.125389.

Lv, X., Zuo, Z., Ni, Y., Sun, J., Wang, H., 2019. The effects of climate and catchment characteristic change on streamflow in a typical tributary of the Yellow River. Sci. Rep. 9 (1), 14535. https://doi.org/10.1038/s41598-019-51115-x.

Mann, H.B., Whitney, D.R., 1947. On a test of whether one of two random variables is stochastically larger than the other. Ann. Math. Stat. 18 (1), 50–60. https://doi.org/10.1214/aoms/1177730491.

Mitchell, P.J., et al., 2016. An ecoclimatic framework for evaluating the resilience of vegetation to water deficit. Glob. Change Biol. 22, 1677–1689. https://doi.org/10.1111/gcb.13177.

Mo, G., Huang, Y., Yang, Q., Wang, D., Mo, C., 2021. Runoff sensitivity to climate and land-use changes: A case study in the Longtan basin, Southwestern China. J. Water Clim. Chang. 12 (4), 1059–1070. https://doi.org/10.2166/wcc.2020.196.

Moore, R.D., Gronsdahl, S., McCleary, R., 2020. Effects of forest harvesting on warm-season low flows in the Pacific Northwest: A review. Confluence: J. Watershed Sci. Manage. 4, 29. https://doi.org/10.22230/jwsm.2020v4n1a35.

Nachtergaele, F.O., 2017. Classification Systems: FAO☆, Reference Module in Earth Systems and Environmental Sciences. Elsevier. https://doi.org/10.1016/B978-0-12-409548-9.10520-2.

Nippgen, F., McGlynn, B.L., Marshall, L.A., Emanuel, R.E., 2011. Landscape structure and climate influences on hydrologic response. Water Resour. Res. 47 (12) https://doi.org/10.1029/2011WR011161.

Patterson, L.A., Lutz, B., Doyle, M.W., 2013. Climate and direct human contributions to changes in mean annual streamflow in the South Atlantic, USA. Water Resour. Res. 49 (11), 7278–7291. https://doi.org/10.1002/2013WR014618.

Peña-Arancibia, J.L., Bruijnzeel, L.A., Mulligan, M., van Dijk, A.I.J.M., 2019. Forests as 'sponges' and 'pumps': Assessing the impact of deforestation on dry-season flows across the tropics. J. Hydrol. 574, 946–963. https://doi.org/10.1016/j.jhydrol.2019.04.064.

Piao, S., et al., 2007. Changes in climate and land use have a larger direct impact than rising CO2 on global river runoff trends. Proc. Natl. Acad. Sci. U.S.A. 104 (39), 15242. https://doi.org/10.1073/pnas.0707213104.

Pomeroy, J., Fang, X., Ellis, C., 2012. Sensitivity of snowmelt hydrology in Marmot Creek, Alberta, to forest cover disturbance. Hydrol. Process. 26 (12), 1891–1904. https://doi.org/10.1002/hyp.9248.

Potithep, S., Nagai, S., Nasahara, K.N., Muraoka, H., Suzuki, R., 2013. Two separate periods of the LAI–VIs relationships using in situ measurements in a deciduous broadleaf forest. Agric. For. Meteorol. 169, 148–155. https://doi.org/10.1016/j.agrformet.2012.09.003.

R Core Team, 2016. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

Rahmat, A., Noda, K., Onishi, T., Senge, M., 2018. Runoff characteristics of forest watersheds under different forest managements. Rev. Agric. Sci. 6, 119–133. https://doi.org/10.7831/ras.6.119.

Reichenau, T.G., et al., 2016. Spatial Heterogeneity of Leaf Area Index (LAI) and Its Temporal Course on Arable Land: Combining Field Measurements, Remote Sensing and Simulation in a Comprehensive Data Analysis Approach (CDAA). PLoS One 11 (7), e0158451.

Ring, E., Andersson, M., Hansson, L., Jansson, G., Högbom, L., 2021. Logging mats and logging residue as ground protection during forwarder traffic along till hillslopes. Croat. J. For. Eng. 42, 445–462. https://doi.org/10.5552/crojfe.2021.875.

Sanborn, P., Lamontagne, L., Hendershot, W., 2011. Podzolic soils of Canada: Genesis, distribution, and classification. Can. J. Soil Sci. 91 (5), 843–880. https://doi.org/10.4141/cjss10024.

Schoonover, J.E., Crim, J.F., 2015. An introduction to soil concepts and the role of soils in watershed management. J. Contemp. Water Res. Educ. 154 (1), 21–47. https://doi.org/10.1111/j.1936-704X.2015.03186.x.

Scown, M.W., Thoms, M.C., De Jager, N.R., 2015. Measuring floodplain spatial patterns using continuous surface metrics at multiple scales. Geomorphology 245, 87–101. https://doi.org/10.1016/j.geomorph.2015.05.026.

Senf, C., Müller, J., Seidl, R., 2019. Post-disturbance recovery of forest cover and tree height differ with management in Central Europe. Landsc. Ecol. 34 (12), 2837–2850. https://doi.org/10.1007/s10980-019-00921-9.

Shao, Q., Traylen, A., Zhang, L., 2012. Nonparametric method for estimating the effects of climatic and catchment characteristics on mean annual evapotranspiration. Water Resour. Res. 48 (3) https://doi.org/10.1029/2010WR009610.

Singh, V.P., 1995. Computer Models of Watershed Hydrology. Water Resources Publications, Littleton CO.

Swift Jr, L.W., Swank, W.T., 1981. Long term responses of streamflow following clearcutting and regrowth. Hydrol. Sci. Bull. 26 (3), 245–256. https://doi.org/10.1080/02626668109490884.

Teutschbein, C., Grabs, T., Laudon, H., Karlsen, R.H., Bishop, K., 2018. Simulating streamflow in ungauged basins under a changing climate: The importance of landscape characteristics. J. Hydrol. 561, 160–178. https://doi.org/10.1016/j.jhydrol.2018.03.060.

van Dijk, A.I.J.M., Peña-Arancibia, J.L., Bruijnzeel, L.A., 2012. Land cover and water yield: inference problems when comparing catchments with mixed land cover. Hydrol. Earth Syst. Sci. 16 (9), 3461–3473. https://doi.org/10.5194/hess-16-3461-2012.

Villarini, G., Wasko, C., 2021. Humans, climate and streamflow. Nat. Clim. Change 11 (9), 725–726. https://doi.org/10.1038/s41558-021-01137-z.

Wang, S., McVicar, T.R., Zhang, Z., Brunner, T., Strauss, P., 2020. Globally partitioning the simultaneous impacts of climate-induced and human-induced changes on catchment streamflow: A review and meta-analysis. J. Hydrol. 590, 125387 https://doi.org/10.1016/j.jhydrol.2020.125387.

Wei, X., Zhang, M., 2010. Research methods for assessing the impacts of forest disturbance on hydrology at large-scale watersheds, Landscape Ecology and Forest Management: Challenges and Solutions in a Changing Globe, pp. 119–147. https://doi.org/10.1007/978-3-642-12754-0_6.

Wei, X., Liu, W., Zhou, P., 2013. Quantifying the relative contributions of forest change and climatic variability to hydrology in large watersheds: A critical review of research methods. Water 5 (2). https://doi.org/10.3390/w5020728.

Yang, D., Yang, Y., Xia, J., 2021. Hydrological cycle and water resources in a changing world: A review. Geography Sustainab. 2 (2), 115–122. https://doi.org/10.1016/j.geosus.2021.05.003.

Zhang, L., et al., 2004. A rational function approach for estimating mean annual evapotranspiration. Water Resour. Res. 40 (2) https://doi.org/10.1029/2003WR002710.

Zhang, M., et al., 2017. A global review on hydrological responses to forest change across multiple spatial scales: Importance of scale, climate, forest type and hydrological regime. J. Hydrol. 546, 44–59. https://doi.org/10.1016/j.jhydrol.2016.12.040.

Zhang, M., Wei, X., 2021. Deforestation, forestation, and water supply. Science 371 (6533), 990. https://doi.org/10.1126/science.abe7821.

Zhang, S., Yang, H., Yang, D., Jayawardena, A.W., 2015. Quantifying the effect of vegetation change on the regional water balance within the Budyko framework. Geophys. Res. Lett. 43 (3), 1140–1148. https://doi.org/10.1002/2015GL066952.

Zheng, J., Yu, X., Deng, W., Wang, H., Wang, Y., 2013. Sensitivity of Land-Use Change to Streamflow in Chaobai River Basin. J. Hydrol. Eng. 18 (4), 457–464. https://doi.org/10.1061/(ASCE)HE.1943-5584.0000669.

Zhou, G., et al., 2015. Global pattern for the effect of climate and land cover on water yield. Nat. Commun. 6 https://doi.org/10.1038/Ncomms6918.