Journal of Hydrology 580 (2020) 124259

Contents lists available at ScienceDirect

Journal of Hydrology

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

Research papers

Disentangling effects of forest harvest on long-term hydrologic and sediment dynamics, western Cascades, Oregon

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ARTICLE INFO

This manuscript was handled by Marco Borga, Editor-in-Chief, with the assistance of Daniele Penna, Associate Editor

Keywords:
Forest management
Streamflow
Sediment yield
Paired catchment

ABSTRACT

The magnitude of sediment yield following forest timber harvest is controlled by increases in both sediment supply and streamflow. Since the relation between sediment transport and streamflow typically follows a power law, small increases in streamflow may translate into large increases in sediment transport. Interpreting the geomorphic effects of streamflow increases is confounded by the fact that timber harvest influences both the hydrologic regime and sediment supply of a watershed simultaneously, making it difficult to isolate the streamflow effect alone. Here we report on a novel approach to this problem using long-term data from two paired catchments located in the H.J. Andrews Experimental Forest, Oregon, USA. We use observed streamflow from the treated (clearcut) and control watersheds to reconstruct a natural streamflow time series for the treated watershed, one that represents streamflow response in conditions prior to harvest. We combine this reconstructed natural streamflow time series with observed relations between streamflow and sediment transport to quantify the background sediment yield and disentangle the relative effects of changes in hydrology and sediment supply. Results indicate that while increases in streamflow can account for modest increases in sediment transport, this is dwarfed by the increased supply of sediment that accompanies most timber harvest. These results have broad relevance to forest timber harvest and fuel management practices worldwide and can be used to constrain or set bounds on likely effects of more modest (i.e., thinning) techniques.

1. Introduction and background

Felling and removing trees changes virtually all aspects of a forest's water and sediment budget: canopy interception and transpiration are reduced, throughfall and soil evaporation are increased, and snow accumulation and melt dynamics are changed in a complex fashion. At the same time, soils are stripped bare and exposed to erosion by various mechanisms, access roads, root strength declines, increasing potential for mass movements, and sediment may be directly delivered to streams by ground disturbance. These effects are intrinsic to forest timber harvest and fuel reduction activities, although the absolute changes to hydrologic and geomorphic processes will vary with the logging methods employed, the intensity and geography of disturbance, and the underlying hydrogeomorphic setting of the landscape.

The hydrologic and geomorphic effects of forest harvest have been the focus of scientific research for over a century, both in the U.S. and abroad (e.g., Robinson et al., 2003; Andréassian, 2004; Cosandey et al., 2005; Brown et al., 2005; Neary, 2016; McDonnell et al., 2018; Evaristo and McDonnell, 2019). Starting with the first paired catchment experiments in Wagon Wheel Gap, West Virginia, USA, and extending to investigations in paired and non-paired catchments from both temperate and tropical locales, numerous studies have documented the effects of cutting trees and related activities on a host of physical processes, including peak and low flows, annual water yields, suspended and bedload sediment yields, and channel morphology. Taken together, our understanding of the hydrogeomorphic consequences of forest harvest is probably better than for most other land use activities, with the possible exception of agriculture.

Although well-established in the literature, the paired catchment experimental design has acknowledged limitations (Grant et al., 2008; Alila et al., 2009). Even in carefully selected adjacent or nested basin pairs, there are both fundamental and evolving differences in how

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https://doi.org/10.1016/j.jhydrol.2019.124259

Available online 22 October 2019

water and sediment are received and processed due to climate, vegetation, soils, topography and geology. Pre- and post-treatment periods vary in length and do not necessarily include similar ranges and sequences of flows, and field measurements include a range of inherent changes in instrumentation along with measurement and analytical errors, making strong statistical inference challenging. Some researchers have even challenged the underlying basis for comparing control/treatment in terms of standard flow metrics, choosing instead to compare frequency relationships (Alila et al., 2009). There are also issues related to scaling of results as most paired catchments are limited to first and second order streams (Zhang et al., 2017). Still, in a conventional control-treatment analysis of harvest effects on hydrology, metrics extracted from streamflow (e.g. instantaneous peaks, annual 7-day average low flow, and total annual flow) are compared pre- and post-treatment, and the difference is interpreted as the forest treatment effect. Linear regression is easily applied to these types of data and has been widely used along with time series analysis, and auto- and cross-correlation, to explore changes in magnitude and trend (Biederman et al., 2015; Caldwell et al., 2016; Hallema et al., 2018).

1.1. Results from paired watershed studies in the Pacific Northwest

Paired watershed studies examining the effects of forest timber harvest on peak flows and water yield have been particularly contentious, with much of the attention focused on studies from the extensive forest lands in the U.S. Pacific Northwest. Despite decades of studies, and for many of the reasons cited above, consensus has not been reached out on the magnitude, persistence, and mechanisms responsible for peak flow and water yield changes following timber harvest. Studies examining the same long-term streamflow data from paired catchment studies in the H .J. Andrews Experimental Forest and other basins in western Oregon, for example, reached conflicting conclusions on the magnitude and causes of peak flow changes, setting off an extensive debate in the literature (Alila et al., 2009; Beschta et al., 2000; Grant et al., 2008; Harr, 1986; Jones, 2000; Jones and Grant, 1996, 2001a, 2001b; Thomas and Megahan, 1998, 2001).

There is, however, broad agreement that hydrologic changes following timber harvest do occur in many forest environments in the Pacific Northwest and beyond (e.g. Gomi et al., 2005; Grant et al., 2008; Moore and Wondzell, 2005). Key issues affecting the magnitude of streamflow changes include the type, intensity and spatial extent of timber harvesting, with clearcutting representing the most extreme case. While forest management practices have evolved over time and clearcutting is no longer practiced on U.S. federal land, it is still the primary timber harvest method used across the globe. The world's forests shrank by 1.3 million square kilometers between 1990 and 2015, a trend that is highly likely continue in the future (World Bank, 2016). Moreover, salvage logging has also increased, triggered by increased fire and drought related tree mortality due to climate change. Although highly location specific, salvage logging operations in many ways resemble conventional clearcutting (Thorn et al., 2018). The long-term data on extreme treatment presented here provides key insights as an end-member on processes responsible for land and water quality degradation following landscape scale disturbances.

In addition to research on the hydrologic effects of forest management, many studies have investigated sediment production and transport following forest harvest in the Pacific Northwest and other regions (e.g. Ambers, 2001; Bathurst and Iroumé, 2014; Beschta, 1978; Birkinshaw et al., 2011; Brown and Krygier, 1971; Bywater-Reyes et al., 2017, 2018; Fredriksen, 1970a,b; Grant and Wolff, 1991; Hotta et al., 2007; Lewis et al., 2001; Macdonald et al., 2003; Mersereau and Dyrness, 1972; Stednick, 2008; Stott, 2005; Swank et al., 2014; Swanson and Fredriksen, 1982). Timber harvest operations such as felling and yarding can alter hillslope hydrology and increase surface erosion by compacting soils and removing organic litter. Although clear-cutting alone can increase sediment production somewhat, studies have shown much larger increases in sediment movement on hillslopes and in channels after slash-burning (Bathurst and Iroumé, 2014; Grant and Wolff, 1991; Lewis et al., 2001; Mersereau and Dyrness, 1972). In the steep, wet landscape of the Pacific Northwest where sediment yields from small watersheds are commonly dominated by episodic mass-wasting events, the most significant changes to sediment transport processes following timber harvest may be related to increased rates of mass movement, including landslides and debris flows, caused by reduced hillslope stability due to loss of root strength (Grant and Wolff, 1991; Montgomery et al., 2000; Rice et al., 1979). Cutting riparian vegetation can also alter the sediment transport regime within a watershed by reducing wood recruitment to channels and increasing debris flow runout, thereby affecting sediment storage patterns in a basin (Lancaster et al., 2001). Though beyond the scope of this study, road-building and the construction of landings on steep slopes can also have substantial impacts on hillslope hydrology and sediment production (e.g. Tague and Band, 2001; Wemple et al., 1996, 2001).

Accelerated sediment production following changes in land-use and land cover is widely viewed as an environmental problem, causing channel bed aggradation, which increases the risk of flooding and enhances bank erosion, and impacting fish habitat by aggrading pools, raising water temperatures, and blanketing spawning gravels with fine-grained sediment (Lisle, 1989; Lisle and Hilton, 1992). Increased sediment transport rates can also affect water quality, which may adversely affect drinking water supplies, aquatic habitat, and recreational values.

1.2. Coupling hydrologic and geomorphic responses to forest harvest

Most studies typically evaluate the effects of timber harvest on streamflow and sediment transport separately, even while acknowledging that the two are intimately coupled. From a geomorphic perspective, changes in either the frequency or magnitude of streamflow will directly affect sediment transport. Specifically, sediment transport (Q_s), whether as suspended load or bedload, is typically related to streamflow (Q) by a power law:

$$Q_s = aQ^b \tag{1}$$

For both suspended and bedload transport, the b-exponent is typically greater than one, meaning that any change in streamflow will have a corresponding exponential effect on sediment transport. While both the a- and b- coefficients vary widely by landscape and channel system, reported values of b, for example, range from 1.42 to 2.96 for suspended load in rivers in different physiographic settings (Morehead et al., 2003) and 2.30 to 5.06 for bedload in gravel-bed rivers in Idaho (Emmett and Wolman, 2001).

This relationship implies that if timber harvest increases streamflow, in a transport-limited basin it will also increase basin sediment transport and yield, independent of any changes in sediment supply. Few studies have explicitly examined this coupling, however, and explored the relative or absolute role of either streamflow changes or increased sediment supply on sediment transport. For the small number of studies that have looked at the relation between streamflow and sediment transport, the literature is, again, often contradictory. A review paper on suspended sediment transport in headwater streams, for example, points to both sediment supply and increased streamflow as being responsible for elevated sediment transport following timber harvest, but acknowledges that was lack the ability to disentangle the relative effects of each (Gomi et al., 2005). Other review papers on forest harvest effects on water quality and quantity, however, treat effects on streamflow and sediment production and transport separately (Anderson and Lockaby, 2011; Croke and Hairsine, 2006; Hubbart et al., 2007; Grant et al., 2008; Moore and Wondzell, 2005; Stednick, 2008; Reiter et al., 2009). Most case-studies of forest harvest impacts report streamflow or water yield and sediment effects as if they are independent (i.e. Hotta et al., 2007; Karwan et al., 2007; Klein et al., 2012; Kreutzweiser and Capell, 2001; Lewis and Keppeler, 2007;

McBroom et al., 2008; Stott, 2005; Stott et al., 2001; Swank et al., 2001; Webb et al., 2012). Grant and Wolff (1991) used a multivariate streamflow model to predict sediment transport following timber harvest in Oregon, USA but did not separate out the effects of logging on streamflow as part of their model. Bywater-Reyes et al. (2017) used generalized least squares regression model in the Trask River Watershed (western Oregon, USA) and concluded a lithologically mediated response of forest management on suspended sediment yield; a similar analysis for the western Cascades emphasized the importance of cumulative annual streamflow and physiography (Bywater-Reyes et al., 2018). Lewis and others (2001; Lewis and Keppeler, 2007) also developed a multivariate sediment transport model for Caspar Creek in California, USA and concluded that increased streamflow volume following logging was the single most important explanatory variable for elevated suspended sediment transport.

Distinguishing between the effects of streamflow increases versus sediment supply increases caused by disturbances associated specifically with logging is important in order to target effective mitigation strategies. For example, if increased sediment flux down channels is driven primarily by increased streamflow, then mitigation strategies should focus on reducing cutover area, managing cutting pattern and block size, and increasing residual stand densities. On the other hand, if sediment supply is driven primarily by ground disturbance and increased erosion rates following timber harvest, then alternative logging strategies that limit the area of ground disruption or soil exposure and maintaining effective riparian buffers around streams to limit sediment delivery are likely to be more effective approaches to reducing basin sediment yields.

In this paper we utilize a reverse regression technique to evaluate the relative and absolute importance of increased sediment transport due to changes in streamflow versus increased sediment supply due to forest timber harvest alone. We begin by developing a reverse regression technique for reconstructing the natural streamflow time series for the treated watershed. We define natural streamflow as the expected water production from a catchment unaltered by forest management or any other disturbance. We then describe how to combine measured and reconstructed streamflow for the treated watershed with observed regressions between streamflow and sediment transport pre- and post-harvest in order to refine estimates of suspended and bedload sediment yields due to timber harvest from both changes in streamflow and changes in sediment supply. We next illustrate this approach using paired catchment data from two small and well-studied sites in the H.J. Andrews Experimental Forest in Oregon, USA. Finally, we conclude with a discussion of the broader implications of this technique for understanding the relative effects of changes in hydrology and sediment supply to sediment yield following forest management.

2. Analytical approach

Our goal is to compare total sediment flux with and without changes in streamflow using a before-after-control-impact (BACI) design method from a paired catchment study in the H.J. Andrews Experimental Forest in Oregon, USA. This requires disentangling forest timber harvest related increases in basin sediment transport due to increased streamflow from changes due to increased sediment supply (Fig. 1). Here we discuss the logic underlying our approach; in the following section we develop the approach analytically.

It is relatively straightforward to calculate total pre- and post-treatment sediment yields for the treated watershed that includes background yield and streamflow effects, using the observed pre- and post-treatment streamflow time series and corresponding sediment rating curves. The post-treatment streamflow time series reflects any changes in hydrology that have occurred as the result of timber harvest and, when combined with the post-treatment sediment rating curve, the resulting sediment yields incorporate the effects of both changes in streamflow and increased sediment supply along with the background sediment yield (Fig. 1, Scenario 1A). This result can be compared with estimates of sediment transport that reflect the increase in streamflow alone. We do this by first coupling the post-treatment streamflow time series with the observed pre-treatment sediment transport relationship to estimate background sediment yield and any increase or decrease in the sediment yield due to timber harvest driven changes in streamflow (Fig. 1, Scenario 1B). Second, we couple the reconstructed natural streamflow time series for the treated watershed during the post-treatment time period, described in following section 2.1, with the pre-treatment sediment transport relation to estimate background sediment yield transport for the treated watershed (Fig. 1, Scenario 2B). Subtracting this background sediment yield from scenario 1B therefore provides an estimate of sediment transport due solely to the change in streamflow.

2.1. Reconstructing a natural streamflow time series for the post-treatment period

Analytically we can describe relationships between measured streamflow from the control and treated catchments using the conventional linear regression method for the pre- and post-treatment time periods:

$$log(Q_{trt})_{pre} = C_1 + C_2 log(Q_{ctrl})_{pre} \tag{2}$$

$$log(Q_{trt})_{post} = C_3 + C_4 log(Q_{ctrl})_{post} + C_5 T \tag{3}$$

where Q_{trt} and Q_{ctrl} are drainage area-weighted measured streamflow in the treated and control watersheds respectively, T is time (in years) since the end of treatment, subscripts $_{pre}$ and $_{post}$ refer to the pre- or post-treatment time periods, and C_1 through C_5 are the regression coefficients. From (2) and (3) we can reconstruct two possible streamflow time series for the treated watershed in the post-treatment period using both forward and reverse regression.

2.1.1. Forward regression

The standard forward regression technique applies the pre-treatment relationship (2) to the post treatment measured streamflow in the control watershed $(Q_{ctrl})_{post}$ to reconstruct natural streamflow time series for the treated watershed $(\widehat{Q}_{trt})_{post}$ during the post-treatment period as if the watershed not been harvested.

$$log(\widehat{Q}_{trt})_{post} = C_1 + C_2 log(Q_{ctrl})_{post} \tag{4}$$

This predicted streamflow $(\widehat{Q}_{trt})_{post}$ is based on a simple extension of the pre-treatment streamflow relationship between the control and treated watersheds.

While at first glance this forward regression might seem to be the logical approach, several factors argue against this as the sole method. First, we want to reconstruct a natural streamflow time series for the treated watershed that accurately captures the flow behavior realized on the treated watershed during the post-treatment period (Fig. 2a). Reconstructed natural streamflow time series for the treated watershed using forward regression will likely resemble the flow realized in the control watershed as opposed to those in the treated watershed (Fig. 2b). Second, we wanted to account for the fact that flows in the treated watershed following harvest were non-stationary and showed a gradual return towards the pre-treatment condition as forests recovered, as noted by previous studies (Beschta et al, 2000; Jones and Grant, 1996; Thomas and Megahan, 1998). Using the stationary pre-treatment relationship between control and treated watersheds would not have captured this trend and its effect on sediment yield.

2.1.2. Reverse regression

We use the reverse regression technique to account for the fact that the streamflow behavior realized over the post-treatment period may be different from those in the pre-treatment period even in its natural state due to climate variability. We reconstructed an additional set of natural

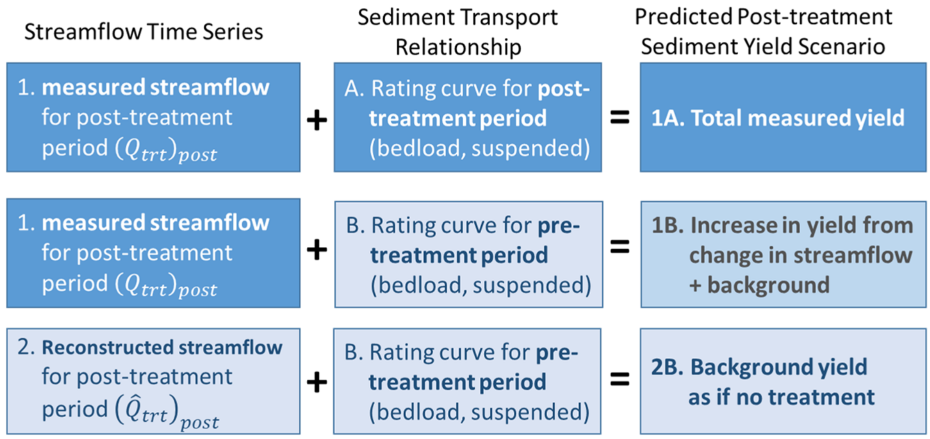

Fig. 1. Framework for evaluating post-treatment total measured sediment yield (1A), background sediment yield and sediment yield attributable to increase in streamflow (1B), and background sediment yield as if the treatment did not occur (2B).

streamflow time series based on post-treatment streamflow in the treated watershed. To do this, we create a reverse regression of the post-treatment streamflow following (3), with the treated watershed $(Q_{trt})_{post}$ as an independent variable and control watershed streamflow $(Q_{ctrl})_{post}$ as a dependent variable:

$$\log(Q_{ctrl})_{post} = C_6 + C_7 \log(Q_{trt})_{post} + C_8 T \tag{5a}$$

We use this reverse regression equation to generate alternate synthetic streamflow time series for the control watershed $(\widehat{Q}_{ctrl})_{post}$ based on $(Q_{trt})_{post}$ for the post-treatment time period.

$$\log(\widehat{Q}_{ctrl})_{post} = C_6 + C_7 \log(Q_{trt})_{post} + C_8 T \tag{5b}$$

The resulting synthetic streamflow $(\widehat{Q}_{ctrl})_{post}$ for the control

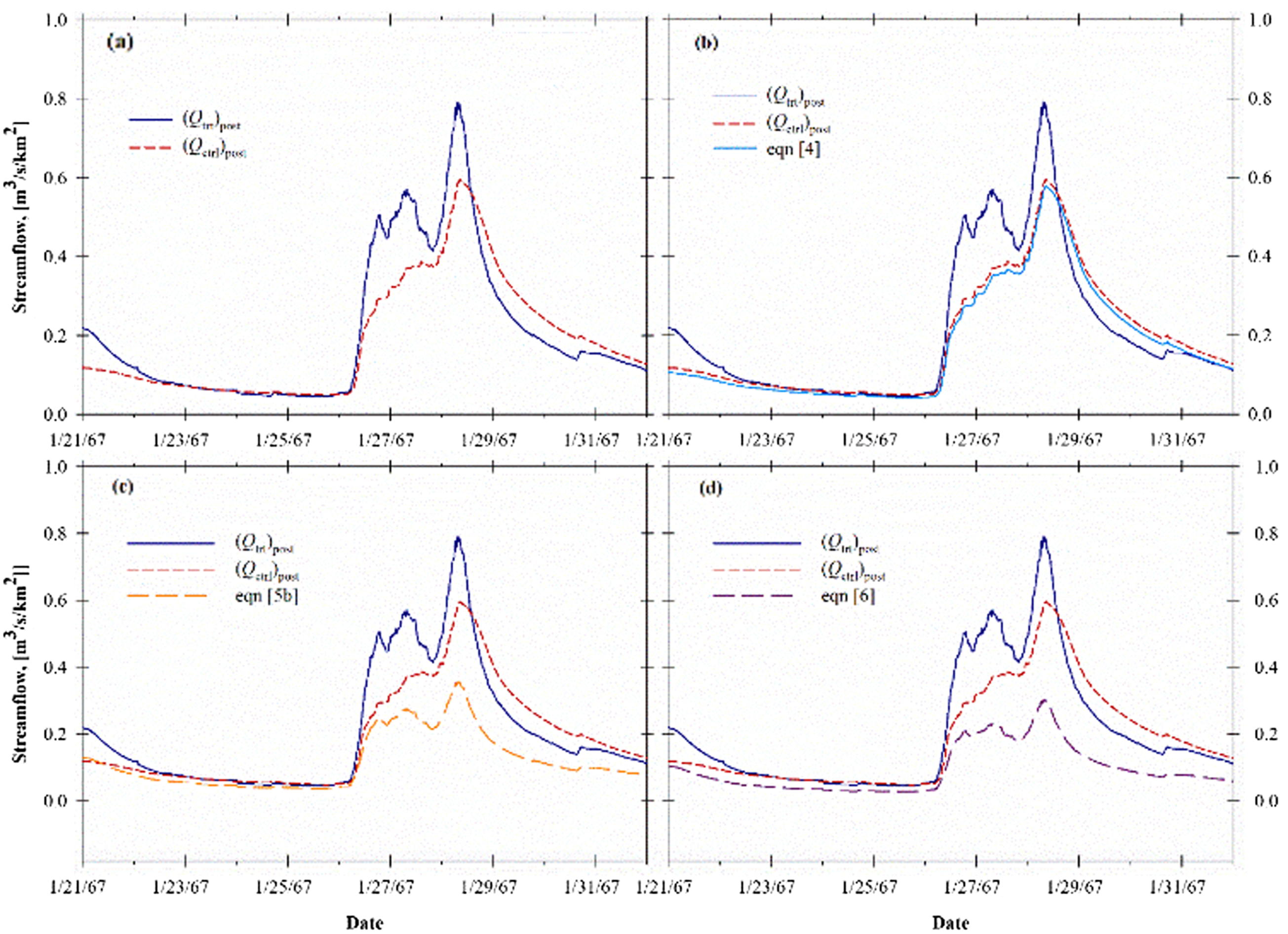

Fig. 2. Storm streamflow illustrating regression techniques for natural streamflow time series reconstruction, (a) measured unit streamflow for the control (dashed red) and treated (solid blue) watersheds, (b) reconstructed natural streamflow for the treated watershed (solid light blue) using conventional forward regression, (c) streamflow for the control watershed (dashed orange) using reverse regression as described in equation [5b] and (d) reconstructed natural streamflow for the treated watershed (dashed purple) as if the basin was not harvested. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

watershed is a function of measured post-treatment streamflow in the treated watershed $(Q_{trt})_{post}$ and time since treatment T (Fig. 2c). It is worth noting here that instead of applying the inverse solution for unknown $(\widehat{Q}_{ctrl})_{post}$ with known $(Q_{trt})_{post}$ in equation (3) we establish the reverse regression (5a) by reversing the roles of the explanatory variable and the response. Reverse regression appears to have a slight edge in terms of the width of the prediction interval (Parker et al., 2010).

We then use the transformed streamflow time series for the control watershed from (5b) in the pre-treatment period equation (2), to predict streamflow in the treated basin for the post-treatment period as if the watershed had not been harvested (Fig. 2d):

$$log(\widehat{Q}_{trt})_{post} = C_1 + C_2 log(\widehat{Q}_{ctrl})_{post} \tag{6}$$

This modeled streamflow $(\widehat{Q}_{trt})_{post}$ provides an estimate of runoff for the same time period and climatic conditions as the post treatment period. As illustrated in Fig. 2, streamflow time series derived using the reverse regression technique more closely resemble the overall shape and timing of the observed streamflow in the treated watershed than the conventional forward regression method.

3. Methods

3.1. Study area

The study watersheds are in the H. J. Andrews Experimental Forest in the western Cascades range of Oregon (Fig. 3). Bedrock consists of Tertiary volcaniclastic rocks with some isolated, basalt flows (Sherrod and Smith, 2000; Swanson and James, 1975). The steep dissected surface of the study area has been shaped by a variety of fluvial, glacial, and mass wasting processes (Swanson and James, 1975), with primary channels oriented southeast to northwest. Mass movements ranging from shallow failures that generate rapid-moving debris flows, to large massive slump-earthflow complexes are primary sources of sediment and organic material to the stream. Hillslope soils are shallow to moderately deep and moderately productive, varying with depth and topographic position (Rothacher et al., 1967). Climate is characterized by mild, wet winters and warm, dry summers (Rothacher et al., 1967). Annual precipitation averaged 2,300 mm per year for the duration of this study, with most falling as light rain between November and April. Historical mean monthly temperatures range from 0.6 °C in January to 17.8 °C in July (Smith, 2002). Rainfall intensities are rarely high enough to generate overland flow, except where ground is compacted by logging roads or skid trails. The study watersheds are located in what is termed as the "transient snow zone" between 400 and 1,100 m elevation, and can receive rain or snow (Harr, 1986). Rain-on-snow events in the transient snow zone commonly generate floods (Jennings and Jones, 2015).

Table 1
Paired-watershed characteristics for H.J. Andrews watersheds 1 and 2.

Watershed	Area,	Elevation, m		Average gradient	
	km^2	gage	maximum	channel	hillslope
WS1	0.959	457	1020	0.278	0.632
WS2	0.603	548	1070	0.364	0.611

Watersheds 1 (WS1) and 2 (WS2) are part of a paired study in which the adjacent catchments were selected for similar size, aspect, and topography (Fig. 3; Table 1). Detailed descriptions of the watersheds are provided in Grant and Wolff (1991) and Rothacher et al. (1967). The streams draining the two catchments are steep, step-pool channels, with gravel to boulder beds and occasional bedrock outcrops. The channel of WS2 (the forested control) has abundant large coarse woody debris as both individual pieces and log jams distributed along the channel. Deposits of fine sediment (sand and finer) is limited to channel margins, shallow pools behind woody debris, and bed interstices. Streams generally run quite clear except during storms. Suspended load is primarily clay products; bedload is typically a mixture of gravel, sand, and particulate organics.

WS1 was 100% clear-cut from fall 1962 through summer 1966 using a skyline suspension system to minimize soil disturbance and eliminate the need for roads within the basin (Fig. 4). Logging debris was then broadcast burned in October 1966 and removed from the streams. Although the original specifications were to remove only wood introduced to the channel by logging while leaving the natural debris, most of the pre-logging legacy wood in the lower part of the watershed

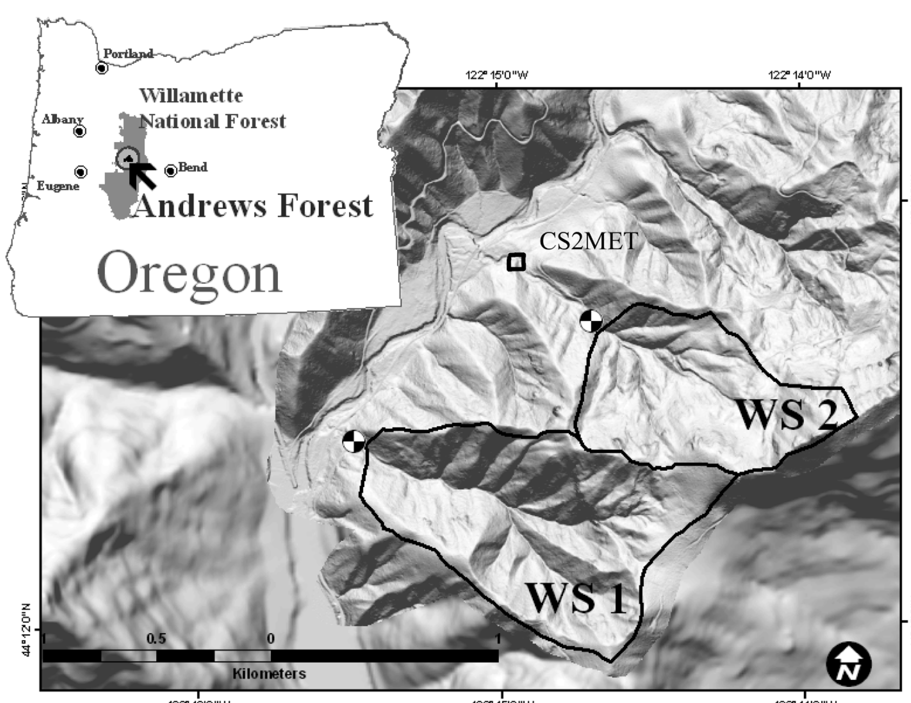

Fig. 3. Study basins Watersheds 1 (WS1, treated) and 2 (WS2, control) in the H.J. Andrews Experimental Forest, Oregon, USA.

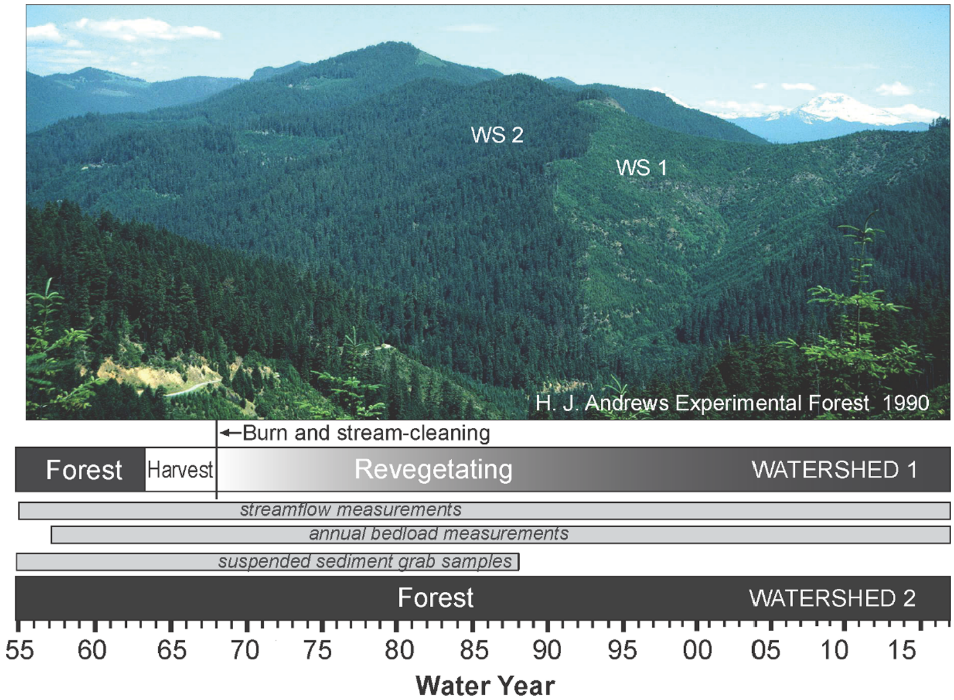

Fig. 4. Timeline of forest treatment and recovery and data availability.

was also removed. WS1 was reseeded in the spring 1967 with Douglas-fir and fill-in planted in 1968. Tree basal area and bole (trunk) biomass in WS1 have increased continuously since harvest, while understory vegetation (< 1.4 m) initially increased in density, peaking above 3,000 stems/ha around 1990, and then decreased as the canopy closed (Halpbern and Lutz, 2013; Lutz and Halpbern, 2006). WS2 is vegetated primarily with 100- to 500-year-old Douglas fir (*Pseudotsuga menziesii*) mixed with western hemlock (*Tsuga heterophylla*) and western red cedar (*Thuja plicata*) (Halpbern and Lutz, 2013; Rothacher et al., 1967).

Logging, storm, and mass movement histories in WS1 have contributed to patterns of sediment production, transport, and export over the entire period of the study (Grant and Wolff, 1991). In particular, large storms in 1964 and 1996, both with recurrence intervals of 50 – 100 years dominate the sediment transport history. Several small debris slides occurred in WS1 on hillslopes following logging during the period 1965–1972 but did not transform into debris flows (Grant and Wolff, 1991).

3.2. Hydrologic data and natural streamflow time series reconstruction

Streamflow monitoring in the H. J. Andrews Experimental Forest began in October 1952 to assess the hydrologic and water quality changes associated with logging and provide baseline data for small, forested catchments at different elevations (Rothacher et al., 1967). Continuous streamflow measurements are recorded at calibrated gage stations in trapezoidal flumes at the mouth of each study basin. For this analysis, we used 15-minute streamflow data for WS1 and WS2 (Johnson and Rothacher, 2018) along with daily precipitation and daily average air temperature data from CS2MET (Daly and McKee, 2019). CS2MET station is located at an elevation of 460 m, downstream from WS2 gauge (Fig. 3).

We used the split sample approach for the least squared linear regression model fitting and validation of the pre-treatment regression model between the control and treated watershed streamflow. Observed streamflow during the pre-treatment period (water years: 1953–1962) was divided between model fitting (water years: 1953–1957) and validation (water years: 1958–1962) periods before fitting a final model using the entire period (water years: 1953–1962) for the purpose of streamflow time series reconstruction during the post treatment period. Streamflow data for the treatment period, water years 1963–1966, were excluded as the values may have been impacted by the logging activities (Fig. 4). Coefficient of determination (R^2), root mean squared error (RMSE), along with Nash-Sutcliffe Efficiency (NSE, Nash and Sutcliffe, 1970) were used to evaluate model performence. Additionally, probability density function plots were produced using observed and reconstructed streamflow time series during pre- and post-treatment periods to ensure the robustness of our reconstruction approach in reproducing the natural streamflow in the treated watershed.

3.3. Sediment sampling and rating curve development

Vertically-integrated suspended sediment grab samples were taken throughout the pre-treatment period starting in 1956 and through 1988 after treatment (Johnson and Fredriksen, 2017) in WS1 and WS2 (Fig. 4). Several additional samples were collected during high flows after 1988 but not included in this analysis. Reduced grab sampling frequency was because of funding and logistical issues. Samples were collected in pint (I US pint = 0.473 L) bottles at the end of the flumes on the rising, peak, and falling limbs of storm hydrographs and during inter-storm periods from October to May. All samples were screened to remove particles > 2 mm, dried, and weighed. When replicate samples were collected, the average sediment concentration was used.

Bedload accumulation was measured annually between 1957 and 2016 by surveying the bottom elevation of sediment basins below the gage stations (Johnson and Rothacher, 2017). Bedload accumulation volumes were then converted to mass using an average bulk density of 1.0 g/cm^3, a value determined from sediment bulk density measurements in the WS1 and WS2 settling ponds (Grant and Wolff, 1991; Fredriksen, 1970b; Swanson et al., 1982). The total volume of material was reduced by 35%, an estimate of the organic content trapped in the sediment basins each year (Swanson et al., 1982). Although the accumulation of organic material probably varied with climate, basin, and land-use, we use a constant fraction because detailed records of organic content were not available for WS1 and WS2. There may be some minor double-counting of suspended sediment, due to settle-out in the

Table 2
Results from the least squared linear regression of log-transformed suspended sediment flux (S_s) as a function of streamflow (Q_{trt}) and time since treatment (T), and annual bedload accumulation (S_B) as a function of maximum annual streamflow $(Q_{trt})_{max}$ and time since treatment (T) for WS1 before and after treatment.

	n[a]	Intercept	[Log (Q_{trt}) [b]] (m^3/s)	[Log (Q_{trt})]^2 (m^3/s)	[Log (Q_{trt})]^3 (m^3/s)	[Log (Q_{trt})]^4 (m^3/s)	sin(k = 1)	cos(k = 1)	T (Years)	R^2	SMF
Suspended Sediment - Log (S_s), [g/s]											
Pre-treatment (1956–1966)	407	1.749	3.080	1.097	0.214	–	−0.109	−0.190	–	0.86	1.869
Post-treatment (1967–1988)	570	3.292	4.228	2.096	0.779	0.112	−0.073	–	−0.055	0.91	2.067
Bedload - Log (S_B), [Mg/km^2]											
Pre-treatment (1957–1966)	9	0.289	2.303	–	–	–	–	–	–	0.62	1.287
Post-treatment (1967–2016)	47	1.825	1.888	–	–	–	–	–	−0.010	0.80	1.114

[a] bedload accumulations below detection limit (1959, 1985) and combined survey years (2010, 2011) were omitted from this analysis. [b]For bedload, we used Log transformed annual maximum 15-minute streamflow.

reservoirs; we believe this amount to be minimal because of the small size of the sediment basins and turbulent flows.

We created rating curves using the pre- (1956–1966) and post-treatment (1967–1988) sediment and streamflow data in order to determine sediment flux and annual yield. Pre-treatment suspended sediment was regressed as a function of log–log polynomial streamflow along with measures of periodicity in the sampling due to snow and other factors (Table 2, Fig. 5). Fourier sine and cosine series of sample day and time along with days since the beginning of the water year were used to capture periodic signals in the suspended sediment samples. Since the treated watershed was replanted soon after harvest, time since harvest was added as an additional independent variable for capturing recovery and predicting its role on post-treatment sediment flux. Annual maximum 15-minute streamflow for the current year, annual maximum 15-minute streamflow for the preceding year, mean annual flow, annual average mean daily temperature, total annual precipitation, annual maximum 1-day total precipitation, and time since harvest (for post-treatment only) were identified as a potential predictors for annual bedload yield during pre- (1957–1966) and post-treatment (1967–2016) periods. However, most of these explanatory variables were statistically insignificant (see section 4.3). All the annual independent and explanatory variables were derived on a water year basis. Both suspended sediment and bedload data during the treatment period (1963–1966) were included in the pre-treatment rating curves. As mentioned previously, clearcutting in WS1 began in 1962 but bedload yields in the treated watershed did not increase until after the burn in 1966 (Supplementary Figure S1). Effect of treatment activities during the 1963–1966 period on the suspended sediment rating curve was statistically non-significant ($p > 0.14$). All rating curves were corrected for bias introduced by the log-transformation using a smearing transformation factor (Uhrich et al., 2014) that does not require the assumption of log-normally distributed residuals (Duan, 1983).

Derived suspended sediment and bedload rating curves for the pre- and post-treatment periods were applied to the observed pre- and post-treatment observed and reconstructed natural streamflow time series to create continuous sediment flux and budget. We calculated three different measures of sediment yield for the post-treatment time period (Fig. 1): 1A) an estimate of actual sediment transported, including changes in both flow regime and sediment supply, 2B) the predicted amount of background sediment yield if the basin had been left in a natural state, and 1B) the predicted amount of background sediment transported along with additional sediment produced due to treatment-induced changes in streamflow. The portion of sediment attributed to changes in streamflow alone can be computed by subtracting the background (2B) from 1B.

4. Results

4.1. Effect of forest treatment on streamflow and water yield

Comparison of streamflow for WS1 (treated) and WS2 (control) using a linear regression model with forest harvest as a binary variable (pre-treatment = 0, post-treatment = 1) shows a statistically significant shift ($p < 0.001$) in the streamflow relationship between the

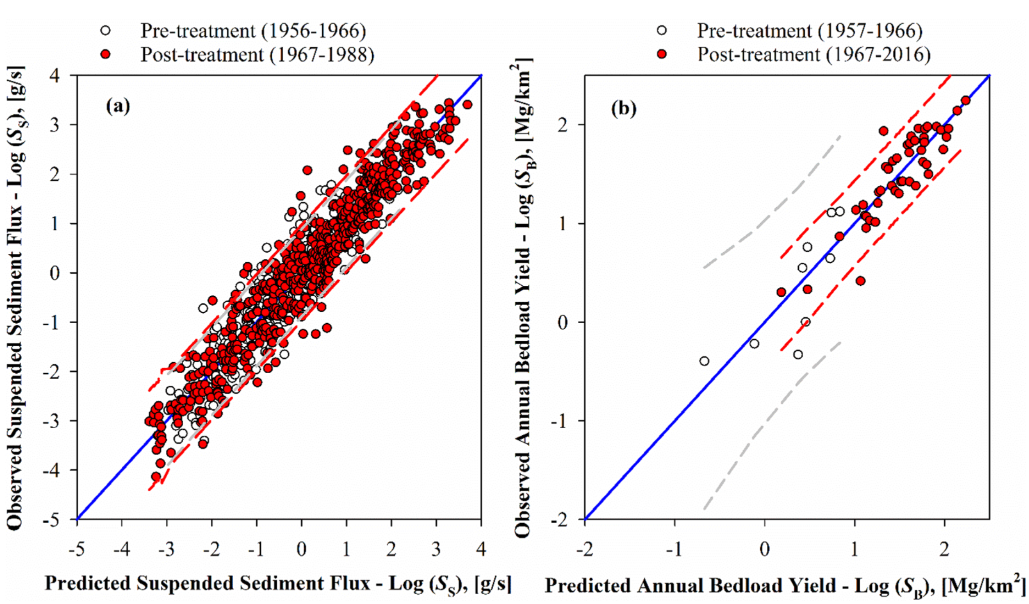

Fig. 5. Observed and predicted event suspended sediment (a) and annual bedload yield (b) for the pre- and post-treatment periods. The solid blue line is the model and dashed lines show 95% prediction interval. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 3
Summary statistics from the linear model predicting treated watershed streamflow, [Log (Q_{trt}), $m^3/s/km^2$], as a function of control streamflow, [Log (Q_{ctrl}), $m^3/s/km^2$], and treatment as a binary variable (pre-treatment = 0, post-treatment = 1).

Model: Log (Q_{trt}) = f(Log (Q_{ctrl}), Treatment)				
Predictor	Estimate	Standard error	t value	Pr(> |t|)
Intercept	-2.732×10^{-02}	1.197×10^{-03}	–22.81	$< 2 \times 10^{-16}$ *
Log (Q_{ctrl})	1.054	6.464×10^{-04}	1629.90	$< 2 \times 10^{-16}$ *
Treatment	1.481×10^{-01}	1.312×10^{-03}	112.90	$< 2 \times 10^{-16}$ *
Log (Q_{ctrl}) × Treatment	9.246×10^{-02}	7.032×10^{-04}	131.49	$< 2 \times 10^{-16}$ *

Residual standard error = 0.241 (N = 2138880), Adjusted R^2 = 0.90, p-value: $< 2.2 \times 10^{-16}$

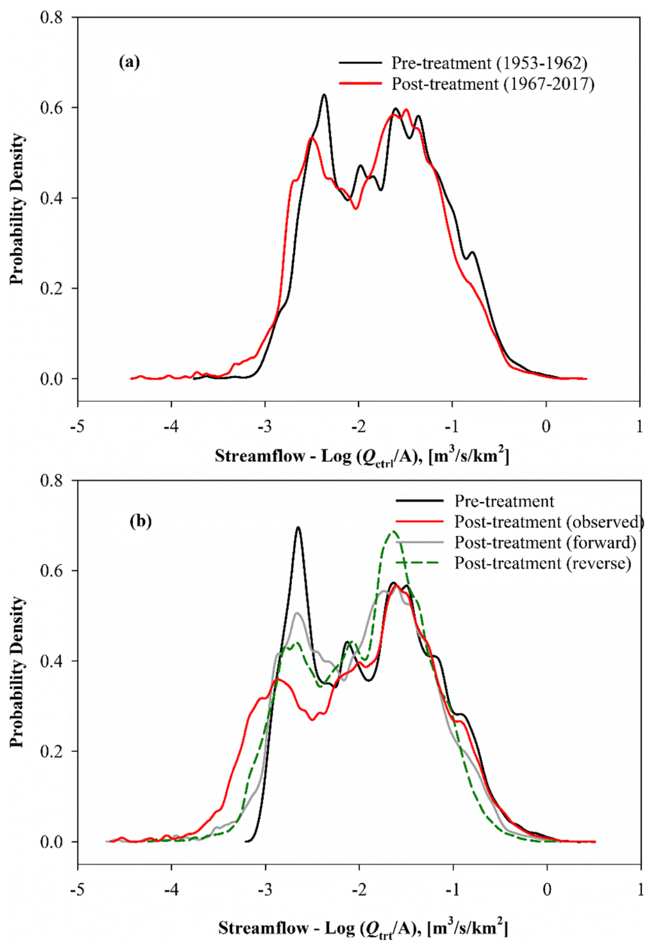

Fig. 6. Probability density function plots for pre- and post-treatment observed streamflow in the control (a) and pre- and post-treatment observed streamflow along with reconstructed natural streamflow, using forward (gray) and reverse (dashed green) regression techniques, for the treated (b) watershed. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

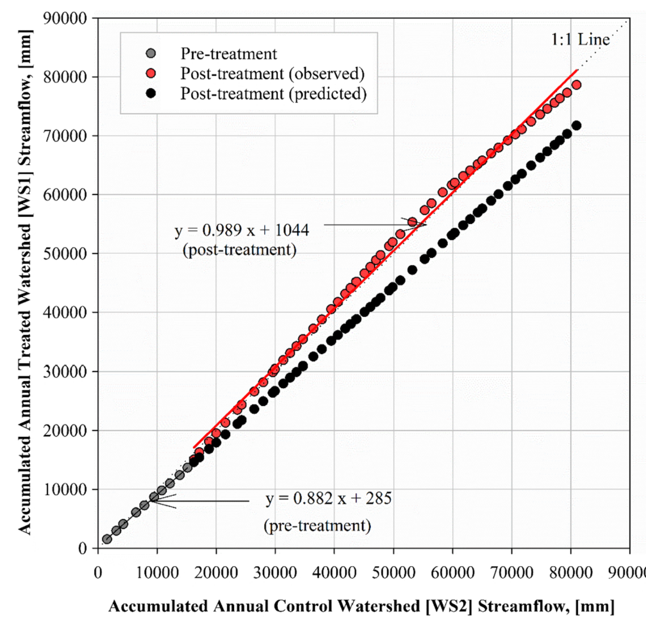

Fig. 7. Double-mass curves of the control (WS2) versus treated (WS1) watersheds (1953–2017).

two watersheds due to forest harvest (Table 3). There is a noticeable shift in the distribution of flows during the post-treatment as compared to pre-treatment period, more so in the treated watershed than those in the control (Fig. 6). Much of the shift in streamflow was in the lower range of flows (i.e., left-side of the probability density plot). We found no statistically significant impact of treatment on annual 15-minute maximum streamflow (p > 0.67 for both treatment and interaction terms). These results are consistent with previous findings showing an increase in streamflow following treatment, particularly for small storms (Beschta et al., 2000; Jones and Grant, 1996; Thomas and Megahan, 1998). In terms of annual water yield, there is an increase in the treated watershed streamflow by 10% or 136 mm/year over the 51 years of post-treatment period (Fig. 7). We attribute the increase in water yield to forest harvest as there was no statistically significant difference (based on one-way ANOVA with post-hoc Tukey HSD) between the pre- and post-treatment annual precipitation (p = 0.8) or temperature (p = 0.8).

We investigated the watershed recovery in terms of streamflow using the aforementioned linear regression model after dividing the post-treatment data into five recovery decades (i.e. 1967–1976, 1977–1986, 1987–1996, 1997–2006, 2007–2016). However, both treatment and interaction term estimates remained statistically significant throughout the 5 recovery decades (Supplementary Table S1). This suggests that even after 51 years of watershed recovery post-treatment streamflow remains different from the pre-treatment period. Stage-streamflow rating curves for WS1 and WS2 have been recently updated using new calibration points between 1996 and 2017 and streamflow beginning water year 1999 was re-calculated (Henshaw et al., 2018). As a result, annual water yield declined between 18 and 24% in WS1 and by 2% in WS2 when compared with the water yield estimated using the original rating curves. Although we believe the updated ratings curves are more accurate, applying multiple rating curves can produce abrupt changes in the streamflow rate and hence complicates long-term analysis.

To further investigate the shifts in streamflow, cumulative monthly random residual time series of streamflow for the treated and control watersheds and precipitation from CS2MET were plotted from October 1952 to September 2017 (Fig. 8). Monthly streamflow and precipitation residual time series, with respect to the mean standardized values, were

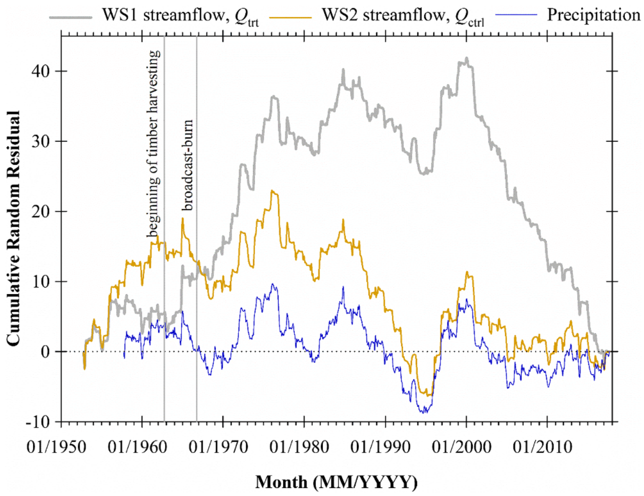

Fig. 8. Cumulative monthly random residuals (with respect to the mean of the measurements) for treated (WS1) and control (WS2) watershed streamflow and precipitation from October 1952 to September 2017. Please note that monitoring of precipitation at CS2MET did not begin until October1957.

developed using the approach described by Mair and Fares (2010). Overall, the trends in WS1 and WS2 streamflow generally follow the observed trends in the precipitation. An upward shift in WS1 and WS2 streamflow is evident half way through the timber harvest, likely driven by the large storms between November 1964 and January 1965. The combined precipitation during this time was over 1900 mm, making it the wettest consecutive three months on record. While WS2 streamflow following this storm continued to decline, driven by a downward trend in precipitation, streamflow in WS1 showed the highest 10 years of sustained increase immediately after the broadcast-burn in the fall of 1966. This beginning of sustained increase in WS1 streamflow was despite below average precipitation from 1966 to 1970. These diverging shifts in WS1 and WS2 streamflow and precipitation strongly suggest timber harvest caused the increase in WS1 streamflow. Another noticeable downward shift in WS1 streamflow occurred between 2001 and 2017, which may have been driven by recent below normal precipitation. A similar recovery pattern is also evident from the double mass analysis (Fig. 7).

4.2. Reconstructed WS1 natural streamflow

As described earlier, we used two approaches based on forward and reverse regression techniques to reconstruct the natural streamflow time series for WS1. Irrespective of the time period, there was a strong relation between the treated and control watershed streamflow with prediction accuracy over 90% (Supplementary Figure S2, Table 4). Performance of the regression model did not diminish when applied against an independent sample for validation (Fig. 9), giving us confidence in the temporal transferability of the model. Time since recovery (variable T in equation (5a)) was a statistically significant ($p < 0.05$) predictor, along with the treated watershed streamflow, of post-treatment streamflow in the control watershed. Overall, the reverse regression technique performed better (NSE = 0.99) in transferring the pre-treatment streamflow characteristics of WS1 and WS2 to the post-treatment period when compared with the forward regression (NSE = 0.91). The forward regression reconstruction method tends to over-estimate streamflow, especially during low flows (Fig. 9), and is unable to reproduce the observed streamflow distribution during the post-treatment period (Fig. 6b). Reconstructed streamflow using forward regression largely mimics the bi-modal post-treatment streamflow distribution in the control watershed as opposed to the tri-modal distribution observed in the treated watershed. In contrast, the reverse regression technique, as described in section 2.1.2, was able to maintain the tri-modal streamflow distribution behavior in the treated watershed.

4.3. Sediment rating curves and recovery

Among the candidates of independent variables, only streamflow, Fourier terms sin(k = 1), and cos(k = 1) were statistically significant in predicting the pre-treatment suspended sediment flux (Table 2). For the post-treatment suspended sediment flux, only streamflow, sin(k = 1), and time since harvest were statistically significant. Using the first power of streamflow alone explained 82 and 83% of the variability in the suspended sediment flux during the pre- and post-treatment periods, respectively (Supplementary Figure S3). Hence, adding these additional predictors improved the model by 4–8% in terms of additional explained variance. Time since treatment alone increased the post-treatment rating curve model R^2 by 5%. Akaike Information Criteria (AIC) during the pre-treatment period was 590 for first power streamflow only model (1) and 509 for the final model. In the post-treatment period AIC declined (the lower the AIC, the better the model) from 1,175 for first power of streamflow only model, 1,017 for first power streamflow and time since harvest model, to 822 for the final model (Table 2). Comparing the observed and predicted suspended sediment flux along with 95% prediction interval showed a similar relatively narrow range, i.e. a high degree of precision of the regression results (Fig. 5a). Approximately 96% of the data points are within the range of values predicted by this regression model.

The bedload yield was related only to annual maximum 15-minute streamflow during the pre-treatment and maximum streamflow and time since harvest during the post-treatment period (Supplementary Figure S3, Table 2). Mean annual streamflow, annual precipitation, average annual mean daily temperature, annual maximum 1-day total precipitation and annual maximum 15-minute streamflow of the preceding year were all statistically insignificant ($p > 0.05$). In terms of prediction interval, only two data points (water years 1991 and 1973) were outside the bound for post-treatment and none for the pre-treatment (Fig. 5b). The linear model over-predicted the annual bedload yield in 1991 (3 Mg/km^2 observed vs. 13 Mg/km^2) and under-predicted in 1973 (86 Mg/km^2 observed vs. 23 Mg/km^2). A relatively wide range of prediction interval for the pre-treatment regression model suggests higher prediction uncertainty. Only 62% of the variation in pre-treatment bedload yield can be explained by this linear model as opposed to 80% during the post-treatment.

Timber harvest shows a statistically significant ($p < 0.001$) impact on sediment-streamflow relationships (Table 5). For suspended

Table 4
Linear regression models used in natural streamflow reconstruction and associated performance statistics during the pre- and post-treatment periods. Model performance statistics (R^2, RMSE, and NSE) were calculated after retransformation of the data to original scale.

Equation	Period	Smearing factor	R^2	RMSE	NSE
Log (Q_{trt}) = 7.9519 × 10^{-3} + 1.071 × Log (Q_{ctrl}) [eqn. (2)]	Fitting (1953–1957)	1.073	0.92	0.027	0.92
	Validation (1958–1962)	–	0.92	0.024	0.91
Log (Q_{trt}) = -2.7316 × 10^{-2} + 1.054 × Log (Q_{ctrl}) [eqn. (2)]	Pre-treatment (1953–1962)	1.065	0.92	0.026	0.91
Log (Q_{ctrl}) = -4.174 × 10^{-1} + 7.912 × 10^{-1} × Log (Q_{trt}) + 5.823 × 10^{-3} × T [eqn. (5a)]	Post-treatment (1967–2017)	1.108	0.88	0.027	0.87

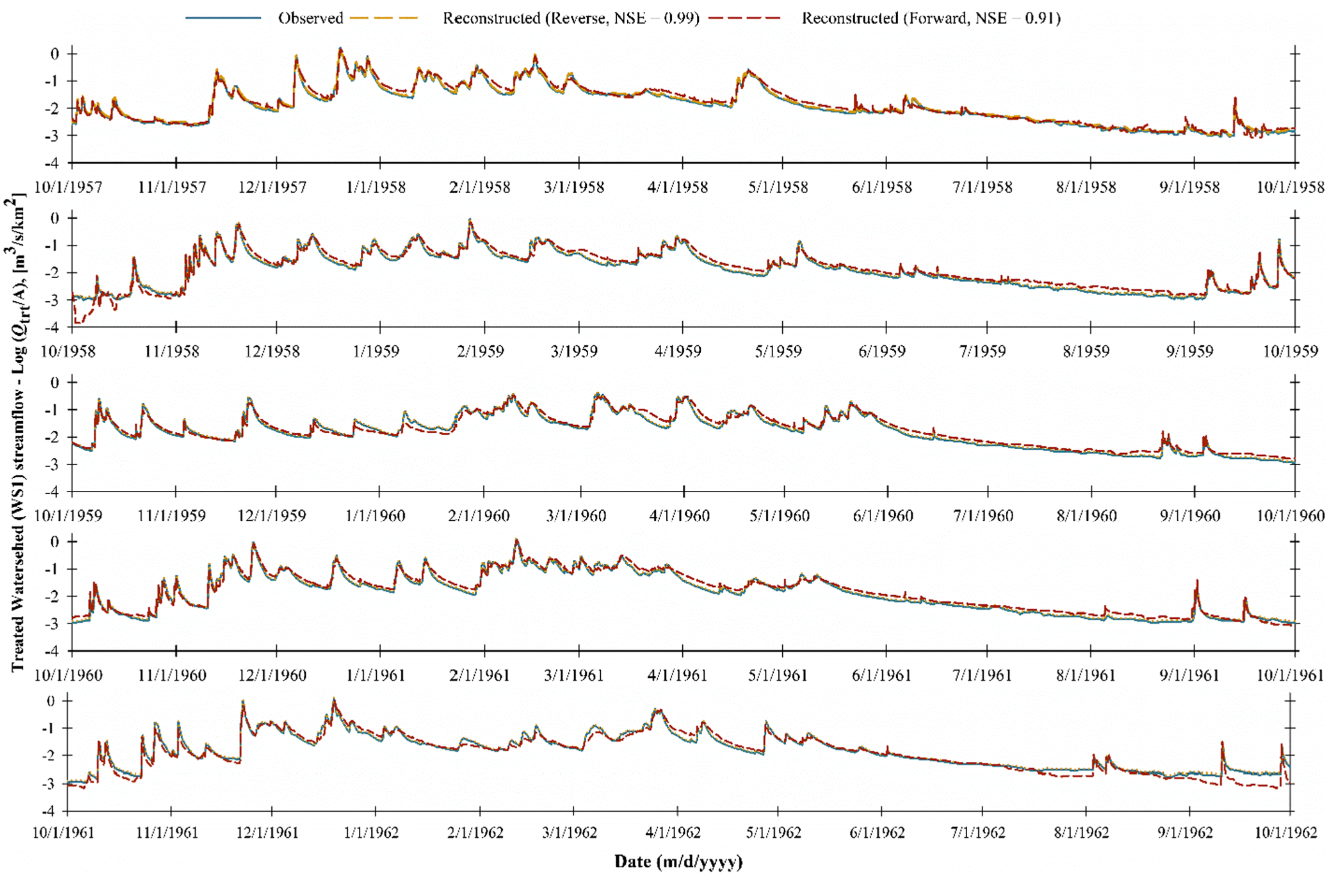

Fig. 9. Validation of streamflow reconstruction technique using forward and reverse regression.

Table 5
Summary statistics from the linear regression model predicting suspended and bedload sediment as a function of streamflow and treatment as a binary variable (pre-treatment = 0, post-treatment = 1).

Suspended Sediment[a] [Model: Log(S_s) = f(Log (Q_{trt}), Treatment)]				
Predictor	Estimate	Standard error	t value	Pr(> |t|)
Intercept	1.062	0.079	13.484	$< 2 \times 10^{-16}$ *
Log (Q_{trt})	0.482	0.056	8.651	$< 2 \times 10^{-16}$ *
Treatment	1.019	0.089	11.477	$< 2 \times 10^{-16}$ *
Log (Q_{trt}) × Treatment	0.456	0.065	7.012	3.51×10^{-12} *
Bedload yield[b] [Model: Log(S_B) = f(Log (Q_{trt})$_{max}$, Treatment)]				
Predictor	Estimate	Standard error	t value	Pr(> |t|)
Intercept	0.342	0.119	2.881	5.918×10^{-3} *
Log (Q_{trt})$_{max}$	1.990	0.512	3.889	3.09×10^{-4} *
Treatment	1.206	0.125	9.676	7.36×10^{-13} *
Log (Q_{trt})$_{max}$ × Treatment	−0.064	0.547	−0.117	9.077×10^{-1}

[a] Residual standard error = 0.624 (N = 784), Adjusted R^2 = 0.58, p-value: $< 2.2 \times 10^{-16}$
[b] Residual standard error = 0.261 (N = 52), Adjusted R^2 = 0.79, p-value: $< 2.2 \times 10^{-16}$

sediment, there is a statistically significant shift in both slope and intercept terms following treatment. In contrast, for bedload only a shift in intercept was statistically significant. This may very well be driven by the differences in the length of record. Suspended sediment data was limited to 1957–1988 and hence limited recovery time when compared with bedload data which extends until 2016. The sediment flux to streamflow relationship shows a gradual recovery (Supplementary Tables S2 & S3). The annual maximum streamflow and bedload sediment relationship by recovery decade shows that the forest harvest has a significant impact on the intercept but not on the slope of the relationship (Supplementary Figure S4).

4.4. Post-treatment sediment yields

We estimate 4,774 Mg/km² of suspended sediment and 2,391 Mg/km² of bedload was transported out of WS1 in the 51 years following treatment, giving average annual yields of 94 and 47 Mg/km² for suspended and bedload respectively (Fig. 10, Table 6). Total average annual yield was 140 Mg/km². Examining the total pre- and post-treatment sediment yields from WS1 through time reveals both the initial increase and roughly exponential decline in sediment yields following treatment (Fig. 10). Annual sediment output from WS1 remained relatively constant (~22 Mg/km²) before and during the treatment period, but increased dramatically after the broadcast burn following timber harvest in 1966. While suspended load declined to pre-treatment levels in the first two decades following treatment, bedload remained elevated, causing the bedload proportion of the total load to increase through time. Bedload accounted for only 17% of the total sediment budget during the pre-treatment period but increased to almost 97% of the total sediment output during the last decade (i.e. 2011–2017). Background bedload and suspended sediment yields estimated using the reconstructed natural streamflow time series based on the reverse regression technique were 32 and 284 Mg/km², respectively (Table 6). This translates into a mean annual total (bedload + suspended) sediment yield of 6 Mg/km². These estimates of background sediment yields are within the range of erosion rates reported for the Pacific Northwest (Swanson et al., 1982).

Changes in sediment supply overwhelmingly dominate streamflow in terms of increased sediment flux. Changes in streamflow alone account for 477 Mg/km² (10%) of the suspended load and 113 Mg/km² (5%) of the bedload over the post-treatment period. Increase in

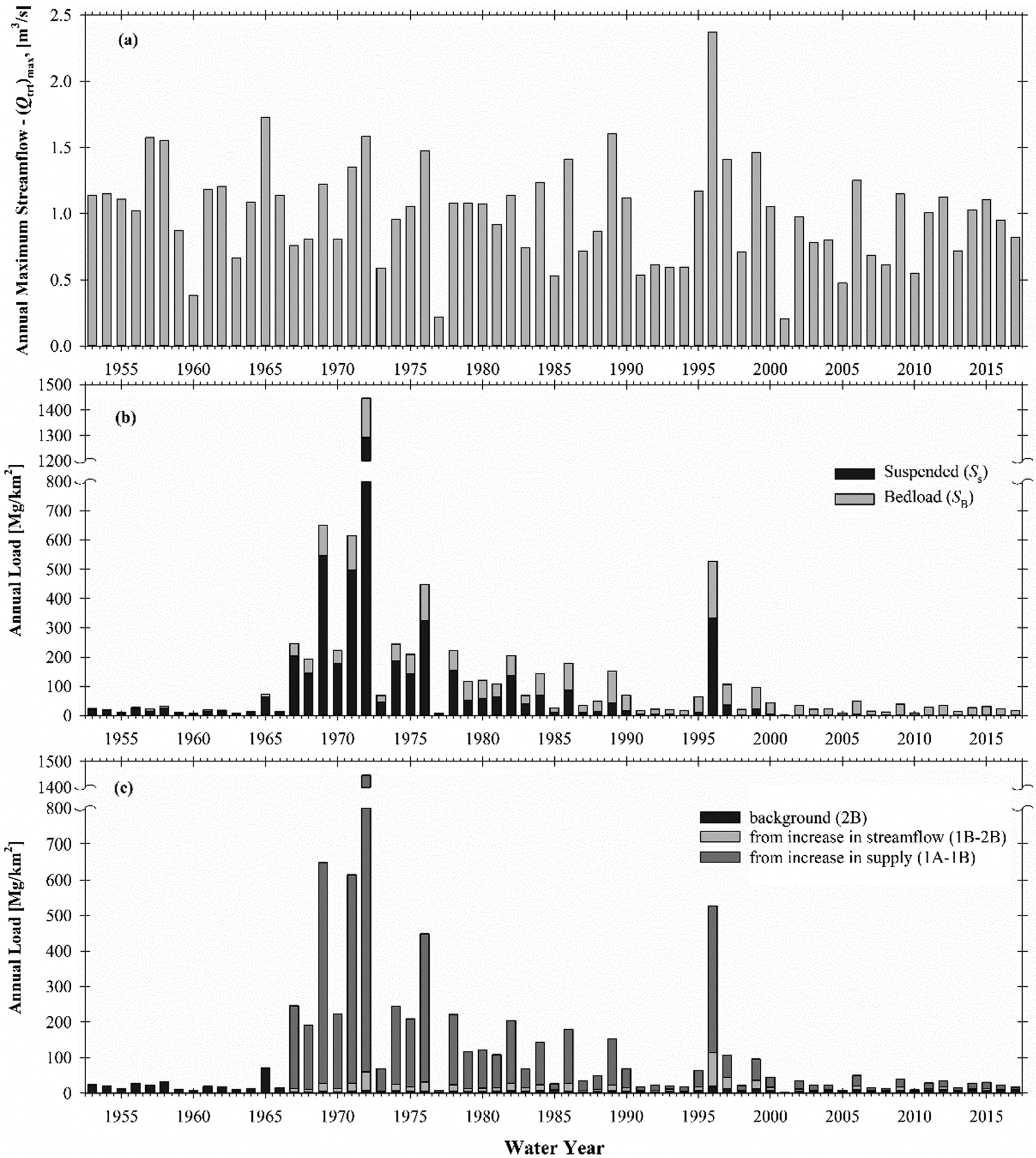

Fig. 10. Watershed 1 (a) Annual peak streamflow and annual total sediment yield as a function of (b) sediment type: suspended sediment (black) and bedload (light gray) and (c) amount transported under three scenarios : 1) reconstructed natural streamflow time series (using reverse regression, Eq. (6)) with pre-treatment rating curve (black), 2) observed post-cut hydrology with pre-treatment sediment rating curve (light gray), showing effect of increased streamflow alone and 3) difference between measured and predicted loads, showing effect of change in sediment supply due to treatment (dark gray).

suspended sediment yield due to increase in sediment supply is 84% of the measured post-treatment total suspended sediment yield. In terms of bedload, 93% of the total measured bedload yield during the post-treatment period can be attributed to an increase in sediment supply.

The first 10 years following treatment yielded 75% of the total post-treatment suspended load but only 33% of the bedload (Fig. 11). Suspended load continued to decrease dramatically in the subsequent 2 recovery decades, with 14% of the total post-treatment yield occurring during decade 2, 9% during decade 3, 2% during decade 4, and $< 1\%$ during decade 5. Bedload yield did not decrease at the same rate (22, 22, 14, and 9% during the recovery decades 2, 3, 4, and 5, respectively). A similar recovery was noted in the cumulative suspended sediment yield attributed to increase in streamflow - both total and increased streamflow attributed suspended sediment stabilized after 1999. We do not see a similar recovery pattern in the bedload yield attributed to change in streamflow. Rather, bedload shows a gradual annual increase with a linear slope of 2.34 Mg/km^2 (R^2 = 0.98). In contrast, cumulative background suspended and bedload yields increased annually by 5.59 Mg/km^2 (R^2 = 0.99) and 0.64 Mg/km^2 (R^2 = 0.97), respectively. The first 10 years following treatment yielded 32% of the total post-treatment suspended load attributed to increase in streamflow which is significantly higher than the 16% of the cumulative background suspended sediment yield and comparable to 33% of the total bedload driven by increased streamflow. The background bedload yield in the

Table 6
Measured and modeled suspended, bedload and total sediment yield for Watershed 1 during the post-treatment period (1967–2017).

Sediment Yield	Scenario	Suspended, Mg/km^2	Bedload, Mg/km^2	Total, Mg/km^2
Total	1A	4,774	2,391	7,165
From increased streamflow + Background	1B	761	145	906
Background	2B	284	32	316
From increased streamflow	1B-2B	477	113	590
From increased sediment supply	1A-1B	4,013	2,214	6,227
Increase from harvest (streamflow + sediment)	1A-2B	4,490	2,359	6,849

first 10 years following treatment was 11% of the total bedload as compared with 26% of the bedload attributed to increase in streamflow (Fig. 11).

4.5. Sediment transport effectiveness

Clearcutting forests in this region results in a sharp increase in suspended sediment transport by the stream network across the entire range of flows. Comparing the density function of post-treatment and background suspended sediment yield versus streamflow shows a dramatic increase in the effectiveness of all flows following treatment (Fig. 12a). Sediment transport effectiveness curves generated by taking the first derivative of the cumulative density function for the 51-year post-treatment period provide information about the distribution of flows responsible for transporting suspended sediment for the two scenarios (Fig. 12b). The peaks of the curves (Fig. 12a) indicate that flows with recurrence intervals between 2 and 5 years are responsible for transporting the majority of the sediment for both scenarios. Not only is the maximum effective streamflow for both scenarios different, but the shape of the two curves suggests that larger flows are more effective following treatment (Fig. 12b). Slopes of the cumulative density functions for the 5 recovery decades relax over time, indicating that this shift in effectiveness towards higher flows exists immediately following treatment, and that larger flows carry a smaller proportion of sediment as the vegetation recovers (Fig. 12b).

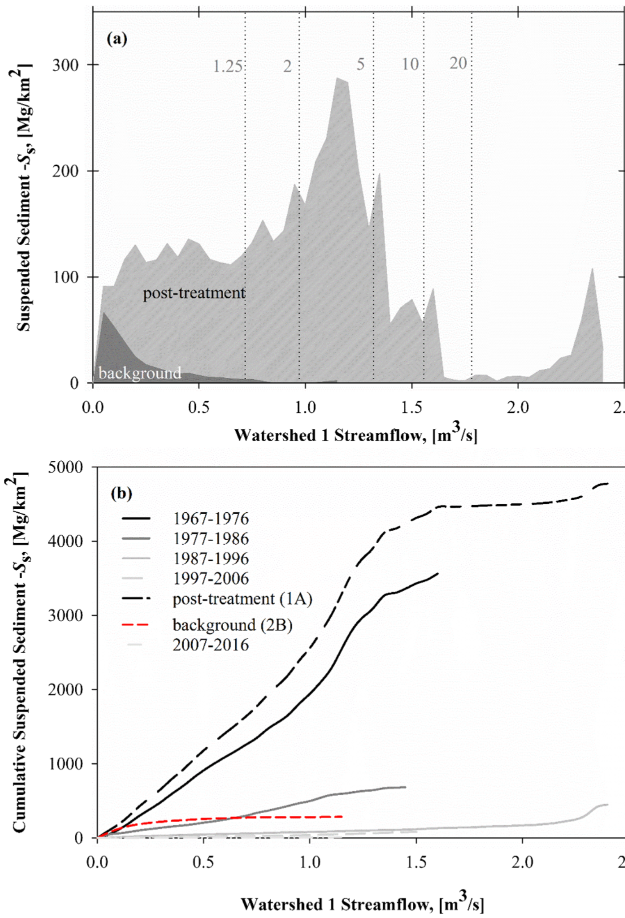

Fig. 12. Geomorphic effectiveness of suspended sediment transport for the range of measured Watershed 1 streamflow. (a) Suspended sediment transport for the measured post-treatment (dark gray) and background yield estimated using reconstructed natural streamflow time series, with return periods noted as vertical lines. (b) Cumulative suspended sediment transport for the measured (black), and reconstructed background sediment yield (dashed black) and the four post-treatment periods (gray).

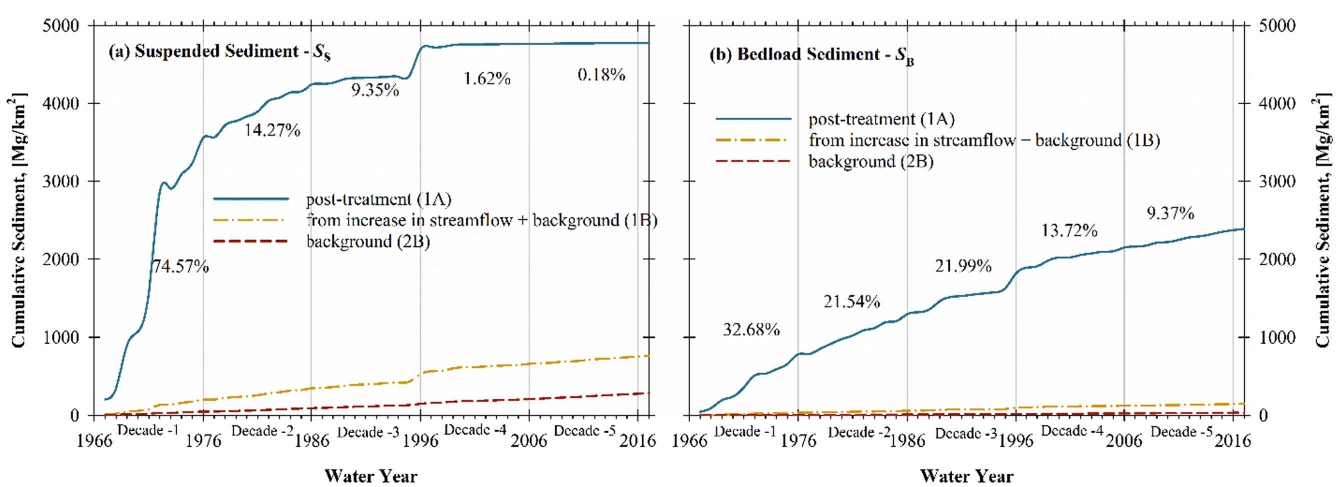

Fig. 11. Cumulative post-treatment WS1 (a) suspended and (b) bedload yields through time for three scenarios: 1) observed post-treatment hydrology and sediment rating curves (blue); 2) observed post-treatment hydrology with pre-treatment sediment rating curve (dashed yellow), showing the background and effect of increased flows; and 3) reconstructed natural streamflow time series using reverse regression (Eq. (6)) with pre-treatment rating curve (dashed red). Percentage shown for amount of total sediment transported in each of the five recovery decades following treatment. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

5. Discussion

5.1. Sediment yield

Our paired catchment results from the H. J. Andrews Experimental Forest demonstrate that the sharp increases in sediment transport following logging can be confidently attributed to the increase in sediment supply and delivery to streams due to the ground disturbances associated with logging rather than increased streamflow. Streamflow increases modestly change the effectiveness of streams to carry sediment but are responsible for < 10% of the increased suspended load and 5% of the increased bedload. These results are the same regardless of whether reverse (Table 6) or forward (Supplementary Table S4) regression is used to establish the post-treatment differences in streamflow, and despite the power law relation between streamflow and sediment transport. Increased sediment supply and delivery to streams contributes overwhelmingly to increased sediment yield in the first few years following harvest. This effect diminishes more or less exponentially over time as both bare areas revegetate and streamflow returns towards pre-treatment levels.

The implication of this finding is that actions to reduce sediment delivery following logging should target reducing ground disturbance above promoting hydrologic recovery of the watershed (say by replanting), although both sets of actions will reduce erosion rates. Reducing ground disturbance can be successfully accomplished by logging methods that minimize stripping vegetation and bare areas and reducing compaction, most notably by suspending logs while transporting them, and avoiding dragging them across the ground. These methods were actually employed in WS1, as discussed below, and the area and extent of soil erosion was undoubtedly less than had the site been tractor yarded (Dyrness, 1965). Maintaining vegetated strips or buffers around streams has also been shown to be an effective way of limiting direct input of sediment to streams (Gomi et al., 2006). Although such practices have become the norm on Federal lands in the Pacific Northwest and elsewhere over the past 20–30 years, they are less commonly adopted in other settings, particularly in the tropics (Putz et al., 2008). Clearcut logging overall has dramatically declined as a forest management practice on federal lands and has largely been replaced by an ecosystem-based management strategy aimed at creating variable stand structures and densities across the landscape (North et al., 2009). Because both canopy removal and ground disturbance are significantly less, both the hydrologic and erosional consequences of these lighter treatments are correspondingly less than those described here (Grant et al., 2008).

In the context of present Pacific Northwest logging practices, therefore, it is tempting to view the WS1 treatment and corresponding consequences as an end-member case of an out-dated logging disturbance. This needs to be qualified on several counts. First, WS1 was deliberately logged without roads or skid trails, and cut logs were lifted by a skyline suspension system rather than being dragged across the ground. A neighboring watershed (WS3) that was also a part of the original paired watershed study had only 25% of its canopy area removed by a skyline suspension system, but 6% of the basin was roaded, resulting in twice the area of deep soil disturbance, and three times the area of compaction as WS1 (Dyrness, 1967).

As a consequence of these differences, most notably as a result of mass failures initiated at roads, WS3 produced four times the erosion as WS1 over the same period, underscoring the importance of mass movement events triggered by logging (Grant and Wolff, 1991). So even within the limited scope of the original paired watershed study, WS1 was less disturbed and produced less sediment than WS3. Previous studies have also highlighted the importance of storm history in relation to the timing of logging; logging was incomplete in WS1 at the time of a major regional flood in 1964, resulting in less erosion than might have otherwise occurred (Grant and Wolff, 1991).

Other forest practices that were common at the time that WS1 was logged may also have influenced results. For example, the practice of burning residual forest material (slash) following logging and removing woody debris from streams may have exacerbated sediment delivery and transport in the years following logging. Post-harvest declines in suspended sediment transport in the decades following treatment, even below pre-harvest rates in some years (Fig. 5b), can be attributed to a combination of dense vegetation regrowth, increased delivery of woody debris to the WS1 channel, and possibly increased bed stability during periods between large storms.

All of these factors limit the applicability of an end-member attribution for WS1. Further context is that erosion rates overall are significantly lower by several orders of magnitude for the western Cascades of Oregon than for many other steep mountainous areas (Walling and Webb, 1996); much of this is due to lithologic and climatic differences. Our results must therefore be placed within these larger contexts in order to interpret the magnitude of land-use effects and absolute degree of disturbance captured by the paired watershed study.

The applicability of our findings to other watersheds beyond the Pacific Northwest and U.S. is inevitably conditioned by differences in topography, geology, forest and soil properties, and logging practices. While the absolute magnitude of changes in hydrology and sediment transport cannot be widely extrapolated beyond the region, we would anticipate that the very strong control of sediment delivery over hydrologic change is likely to be observed elsewhere as long as forest cover is maintained. If, however, land use changes result in dramatic increases in impermeable areas and soil compaction, particularly in tropical areas (e.g., Ziegler et al., 2006), or urbanization, it is likely that hydrologic changes will play an increasingly important role in mediating accelerated sediment transport.

Our results highlight both the strengths and limitations of paired small watershed studies as a means of evaluating the effects of land management treatments on watershed processes. Long-term paired basin studies provide an unparalleled means of empirically and quantitatively assessing the consequences of land-use activities at a scale that is large enough to be relevant for land management, yet small enough to provide insight into specific physical mechanisms driving changes (Andreassian, 2004). They represent real-time, full-scale landscape experiments, and have provided a foundation for characterizing both the magnitudes and variabilities of hydrologic and geomorphic response to disturbances over time scales of years to decades. Such timescales are sufficient to capture a range of climatic conditions, and also provide a basis for measuring recovery or return towards pre-disturbance conditions; it is difficult to imagine arriving at these data by any other means. Data from these experiments can be used to derive sediment transport and transport efficiency relationships that can be generalized to other watersheds and settings. And they can be used to compare responses across biogeoclimatic gradients and help populate and test regional scale sediment transport models (i.e., O'Connor et al., 2014).

Yet the diversity of responses from paired watershed studies underscores the complexities of interpreting responses to land-use disturbances in simple cause-and-effect terms (Andréassian, 2004; McDonnell et al., 2018). Moreover, our comparison of forward and reverse regression techniques shows that different methods produce different results (but similar interpretations) when comparing control-treatment pairs. This inevitably raises the question of what is the "right" approach to compare controls and treatments, a question for which there is no simple answer. Reverse regression better captures the changes in hydrograph due to climate variability but may ignore the fact that forest harvest may also alter the shape of the hydrograph. Forward regression is based on the assumption that pre-treatment relationships between the control and treated watersheds are temporally transferable. On balance, we favor the reverse regression approach if the goal is to more closely capture the actual flow variability that can occur during the post-treatment period due to climate non-stationarity.

5.2. Uncertainty in the sediment budget

Several assumptions underlying our approach deserve comment. The one potentially having the largest impact on the results of this study involves changes in the streambed elevation and flow rating curves over time. Large flood events, such as those in 1965 and 1996, are major drivers of not only the sediment budget, but also channel geometry and stream bed morphology. Bywater-Reyes et al. (2018) show an increase in Lookout Creek, mainstem of the H. J. Andrews Experimental Forest, bed elevation by 0.2 and 0.3 m during 1965 and 1996 floods, respectively. Although, streambed usually recovers to pre-flood levels, applying a static streamflow rating curve to a temporally evolving channel geometry adds a degree of uncertainty in the estimated runoff. Change in streambed elevation is less of an issue though when measuring streamflow using fixed flumes as in the case of WS1 and WS2.

Measurements of suspended sediment stopped in the 1980 s for logistical and budgetary reasons. We assumed that the sediment rating curves are representative of all possible values over the entire streamflow range. While the high density of measured points (n = 406, pre-treatment and n = 570, post-treatment) makes this a reasonable assumption for low to moderate streamflows (sampled flow range: 0.0001–1.52 m^3/s, pre-treatment and 0.00025–1.46 m^3/s, post-treatment), the infrequency of high flows and therefore paucity of sediment samples collected at high streamflows may lead to underestimates of total suspended sediment yield. The range of flows observed in the two watersheds during the post-treatment period varied over 0–2.39 m^3/s.

In spite of these assumptions, the method appears to produce reasonably robust results. We tested it by comparing suspended sediment yields estimated from a rating curve to three-week composite suspended sediment samples collected proportional to streamflow for the control WS2. Results from the regression approach compared favorably with the measured sediment yields (R^2 = 0.83, n = 280). Furthermore, our estimates of average annual background sediment yield for WS1 (6–10 Mg/Km^2 suspended and 0.6–1.6 Mg/Km^2 bedload) are similar to the (9 Mg/km^2 suspended and 3.2 Mg/km^2 bedload) values reported from a nearby watershed (watershed 10) equipped with an automatic pumping water sampler and bedload trap (Swanson et al., 1982).

While the focus of this study has been exclusively on the effects of clearcut logging, a similar paired-watershed approach could be used in other intensively managed or disturbed landscapes. An intriguing possibility posed by a more widespread adoption and utilization of data from long-term paired watershed studies across a range of intensively managed landscapes (i.e., agricultural, those burned by wildfire or shifting agriculture, urbanized, restored by various means) is that we could begin to more rigorously assess the effects of land-use across the full panoply of management types in various geographic settings. One outcome of this type of multiple land-use comparisons is that we could begin to better recognize the full range of land-use effects on erosional processes and rates, and populate conceptual models of disturbance (e.g., Figure 6.7 in Turner and Gardner, 2015) with actual quantitative measurements. This would help compare different land-uses in terms of both their relative and absolute effects on sediment production, transport, and yield, giving a stronger foundation for land-use decisions and global sediment transport models.

6. Conclusions

Long-term paired streamflow and sediment data from small experimental catchments permit disentangling the separate effects of increased sediment supply and increased runoff on sediment transport following timber harvest. Our results suggest that streamflow increases alone produce modest increases in sediment transport rates, resulting in nearly twice as much sediment transported out of WS1 following treatment than would have been transported had the basin not been harvested. Changes in sediment supply following harvest have far more influence —twenty-fold —on the sediment transport regime. Clearcutting WS1 increased the effectiveness of all sediment transporting flows by approximately an order of magnitude and increased the relative effectiveness of larger magnitude, less frequent flows in the first decade following treatment.

Sediment yields following harvest declined approximately exponentially over time, as both sediment source areas and vegetation recovered. Annual suspended sediment yields returned to pre-treatment levels in the first two decades following treatment, yet bedload yields remained high throughout the duration of this study. Results of this study hold promise for better targeting land management and restoration activities to minimize the long-term impacts, consequences, and legacies of intensive land-use disturbances.

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.

Acknowledgements

We thank the many people who gave up decades of Thanksgiving dinners to collect sediment samples at the H. J. Andrews Experimental Forest, the survey crews who measured bedload accumulation, Don Henshaw and Hazel Hammond for providing us with the flow and sediment data, and Stephen Lancaster, Sherri Johnson, and Fred Swanson for valuable input during study development.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jhydrol.2019.124259.

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