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Quantifying effects of forest harvesting on sources of suspended sediment to an Oregon Coast Range headwater stream

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ABSTRACT

Elevated fine sediment transport to streams can negatively affect aquatic ecosystem health, downstream infrastructure, and community water supply. Forest harvesting activities can increase the delivery of fine sediment to streams due to intensified erosion or mass wasting from hillslopes, roads, and stream channels. However, quantifying the effects of forest harvesting on sediment inputs to streams and the effectiveness of current best management practices (BMPs) at mitigating these effects remains a challenge. In this study, we used sediment source fingerprinting techniques to quantify and compare the sources of suspended sediment to a stream draining a recently harvested catchment and a nearby, unharvested reference catchment in the Oregon Coast Range of the U.S. Pacific Northwest. In each stream, we quantified the proportional contributions of suspended sediment from three potential source areas: hillslopes, roads, and streambanks. The primary source of suspended sediment in the harvested catchment was streambank sediment (90.2 ± 3.4%) with lesser amounts of hillslope (7.1 ± 3.1%) and road (3.6 ± 3.6%) sediment. Interestingly, the proportional contributions of suspended sediment in the reference catchment were similar, with the majority from streambanks (93.1 ± 1.8%) followed by hillslopes (6.9 ± 1.8%). There were no contributions from roads in the reference catchment, despite a similar road network as the harvested catchment. We also quantified monthly sediment mass eroded from 36 × 1 m^2 hillslope plots. The sediment mass eroded from the general harvest area (96.5 ± 57.0 (*SE*) g) was ~10.6-times greater than the sediment collected in the riparian buffer (9.1 ± 1.9 g) and ~4.6-times greater than the sediment collected on the unharvested, reference hillslope (21.0 ± 3.3 g). While this study provides evidence of effectiveness of contemporary BMPs (e.g., riparian management areas, limits to cutblock size, reduced impact forest harvesting techniques, road building and maintenance) at mitigating sediment delivery to streams, additional research is needed as existing studies do not adequately reflect the broad range of climate, geology, topography, and vegetation in the Pacific Northwest, which drive highly variable hydrologic and geomorphic processes in the region.

1. Introduction

Suspended sediment often accounts for the majority of particulate matter transported by rivers and streams (Meade et al., 1990; Walling and Fang, 2003; Turowski et al., 2010). Forested headwater streams naturally transport suspended sediment derived from both external (e.g., bank erosion, hillslope erosion, mass movements, and linear features) and in-channel sources (e.g., fine sediment deposited in the streambed or behind large wood) (Gomi et al., 2005). However, excessive transport and deposition of fine sediment in headwater streams can have multiple harmful effects on aquatic habitat, primary producers, macroinvertebrates, and fish (Wood and Armitage, 1997; Greig et al., 2005; Bilottaa and Braziera, 2008; Olson and Hawkins, 2017). High concentrations of suspended sediment also increase the potential for transport of other water quality constituents, including nutrients, heavy metals, organics, and pathogens, which can affect aquatic ecosystems and create challenges for drinking water treatment in downstream communities (Dearmont et al., 1998; Emelko et al., 2011). Downstream transport of elevated sediment loads can also impact estuarine and coastal water quality (Thrush et al., 2004), such as smothering of benthic communities (Norkko et al., 2002; Thrush et al., 2003).

Given the many potential negative effects associated with excessive sediment in water bodies, there has long been concern for increased sediment supply to streams due to forest management activities (Harr and Fredriksen, 1988; Binkley and Brown, 1993). Historically, timber

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harvesting operations in headwater catchments often resulted in increased suspended sediment concentrations and yields in headwater streams (Beschta, 1978; Reid and Dunne, 1984; Grayson et al., 1993). In many cases, these increases in fine sediment inputs to streams were attributed to soil compaction and the creation of impervious surfaces during timber harvesting operations, which represent locations where infiltration-excess overland flow may occur, even during low-intensity precipitation events (Bilby et al., 1989; Megahan et al., 2001; Ziegler et al., 2001; Lane and Sheridan, 2002; Sidle et al., 2004). Forest road networks are often cited as the primary sources of sediment delivery to streams (Luce, 2002; Wemple and Jones, 2003; Brown et al., 2013); however, the use of heavy machinery, such as harvesters, skidders, and yarders, during forest harvesting operations can also compact soils resulting in increased bulk density and decreased air-filled porosity, infiltration capacity, and hydraulic conductivity (Motha et al., 2003; Litschert and MacDonald, 2009). Such changes in soil physical properties can lead to lower infiltration rates and elevated erosion from some harvested hillslopes (Croke et al., 1999). Secondary activities associated with forest harvesting, such as slash burning and disposal, can also expose hillslope mineral soils and increase rates of hillslope erosion (Beschta, 1978; Robichaud and Waldrop, 1994). Moreover, removal of trees and subsequent alteration of the hydrologic regime, including increased runoff and peak flows, can increase streambank erosion and remobilization of stored sediment from in-channel sources (Jones and Grant, 1996; Basher et al., 2011; Birkinshaw et al., 2011).

Due to concerns about water quality, best management practices (BMPs) are now required or encouraged in most regions during forest operations to reduce the potential for erosion and nonpoint source pollution from excessive suspended sediment delivery to water bodies (Broadmeadow and Nisbet, 2004; Ice et al., 2004). Current practices include a broad range of approaches, including retention of forested buffers around water bodies, limited allowable cutblock sizes, restricted harvest operations near water bodies or on steep slopes, use of lighter and longer reach machinery, and road building, use, and maintenance activities (Adams and Storm, 2011; Oregon Forest Resources Institute, 2011; Oregon Department of Forestry, 2018). However, many questions remain about the effectiveness of BMPs at mitigating nonpoint source pollution to protect beneficial uses of water (Cristan et al., 2016). Some of this uncertainty is due to contradictory results from studies, which have included a broad range of forest management practices, management intensities, catchment characteristics (e.g., forest type, soils, geology, climate, physiography, etc.), and implementation of BMPs (Aust and Blinn, 2004; Anderson and Lockaby, 2011). For example, while many recent studies have demonstrated no change or a reduction in erosion and sediment delivery to streams with properly applied BMPs (Keim and Schoenholtz, 1999; Wynn et al., 2000; Hotta et al., 2007), others have observed increased sediment delivery after contemporary forest harvesting practices (Arthur et al., 1998; Wear et al., 2013; Bywater-Reyes et al., 2017). Using 30 years of water quality data from four locations in the Deschutes River watershed in western Washington, Reiter et al. (2009) provided evidence for decreasing trends in turbidity associated with improved BMPs; however, there were still detectable relationships between the annual percent catchment harvested and turbidity levels, associated with sediment delivery to streams.

Uncertainty about the efficacy of BMPs is also partly due to the many challenges associated with identifying the various sources (e.g., general harvest areas, skid trails, roads) of in-stream suspended sediment (Collins and Walling, 2002). Anderson and Lockaby (2011) identified the uncertainty of sediment sources associated with forest management activities as a critical research gap, which remains relevant. Sources of suspended sediment often respond to complex interactions between numerous factors that can produce high temporal and spatial variability in sediment mobilization and delivery to streams (Collins and Walling, 2004). Moreover, it remains unclear how current BMPs might influence sediment connectivity, or the efficiency of transfer of sediment from sources to streams, especially across a heterogeneity of landscapes (Cavalli et al., 2013; Wohl et al., 2019). Additionally, field-based studies, which are necessary to collect representative data to further our understanding of these interactions, have been on the decline because they are increasingly expensive and time consuming (Burt et al., 2015).

Fortunately, sediment fingerprinting techniques have proven to have broad utility for determining temporal or spatially integrated estimates of the likely source or provenance of sediment (Walling, 2005; Collins et al., 2010). The techniques are based on the idea that sediment derived from distinct sources can be differentiated by unique physical or chemical properties (Collins and Walling, 2004). Information on the source of fine sediment is critical for improving understanding of (a) the erosion and sediment delivery processes, (b) sediment-associated nutrient and contaminant fluxes, (c) the differential effects of specific sediment sources on aquatic ecosystem health, and (d) whether best management practices aimed at mitigating sediment transport to water bodies are effective (Ongley et al., 1981; Walling, 2013; Sear et al., 2016).

In the U.S. Pacific Northwest, forests and forest harvesting remain critical for the economy, while clean water is essential for healthy communities, recreational opportunities, and habitat for fish and wildlife. Thus, understanding the effects of current forest management practices on the delivery of various sources of sediment to headwater streams remains an important challenge. Here, we present results from a study of two catchments from the Oregon Coast Range with the goal of determining the effectiveness of contemporary forest harvesting practices at mitigating sediment delivery to streams. Specifically, the objectives of our study were to use sediment source fingerprinting techniques, as well as conventional approaches, to quantify: (a) the primary sources of suspended sediment to headwater streams in a forested and timber harvested catchment, (b) the longitudinal variability, from stream head to outlet, in the primary sources of sediment in a harvested catchment, (c) the variability in the primary sources of suspended sediment throughout the year, and (d) the effectiveness of current BMPs at mitigating sediment movement from harvested hillslopes to streams.

2. Methods

2.1. Site descriptions

The study occurred in two catchments located in the Oregon Coast Range (44.55 °N, 123.52 °W) of the Pacific Northwest. The climate in the region is maritime with average annual precipitation of 1718 mm (30 year normal from 1981 to 2010), with ~72% falling between November to March (PRISM Climate Group, 2004). Approximately 98% of the annual precipitation falls as rain, with snow events occurring infrequently. The study included one catchment (Enos Creek) that was partially clearcut harvested in the summer of 2016 and an unharvested reference catchment (Scheele Creek) located ~3.5 km northwest of Enos Creek (Fig. 1). The two catchments had comparable drainage areas (Enos: 1.2 km^2; Scheele: 1.3 km^2), lithology (Coast Range basalt), and soils (silty clay loam). Topographically, the catchments had similar relief (Enos: 178 m; Scheele: 222 m) and mean slopes (Enos: 10°; Scheele: 12°), with maximum slopes of about 45° at both catchments. The canopy cover along the thalweg of each stream, as quantified with a spherical densiometer, was 71 ± 8 (SD) % at Scheele Creek and 65 ± 11% at Enos Creek.

In the summer of 2016, ~3% (0.16 km^2) of the Enos Creek catchment was harvested following the current Oregon Forest Practices Act policies and BMPs. The total area harvested was relatively small, which was reflective of contemporary BMPs in the region that limit the spatial extent of harvesting. Moreover, a ~15 m (~50 ft) fixed-width riparian buffer was retained at the base of the harvested hillslope, adjacent to the stream. However, the harvest occurred on steep (~40–45°) hillslopes adjacent to the stream for ~530 m of stream length. Harvest

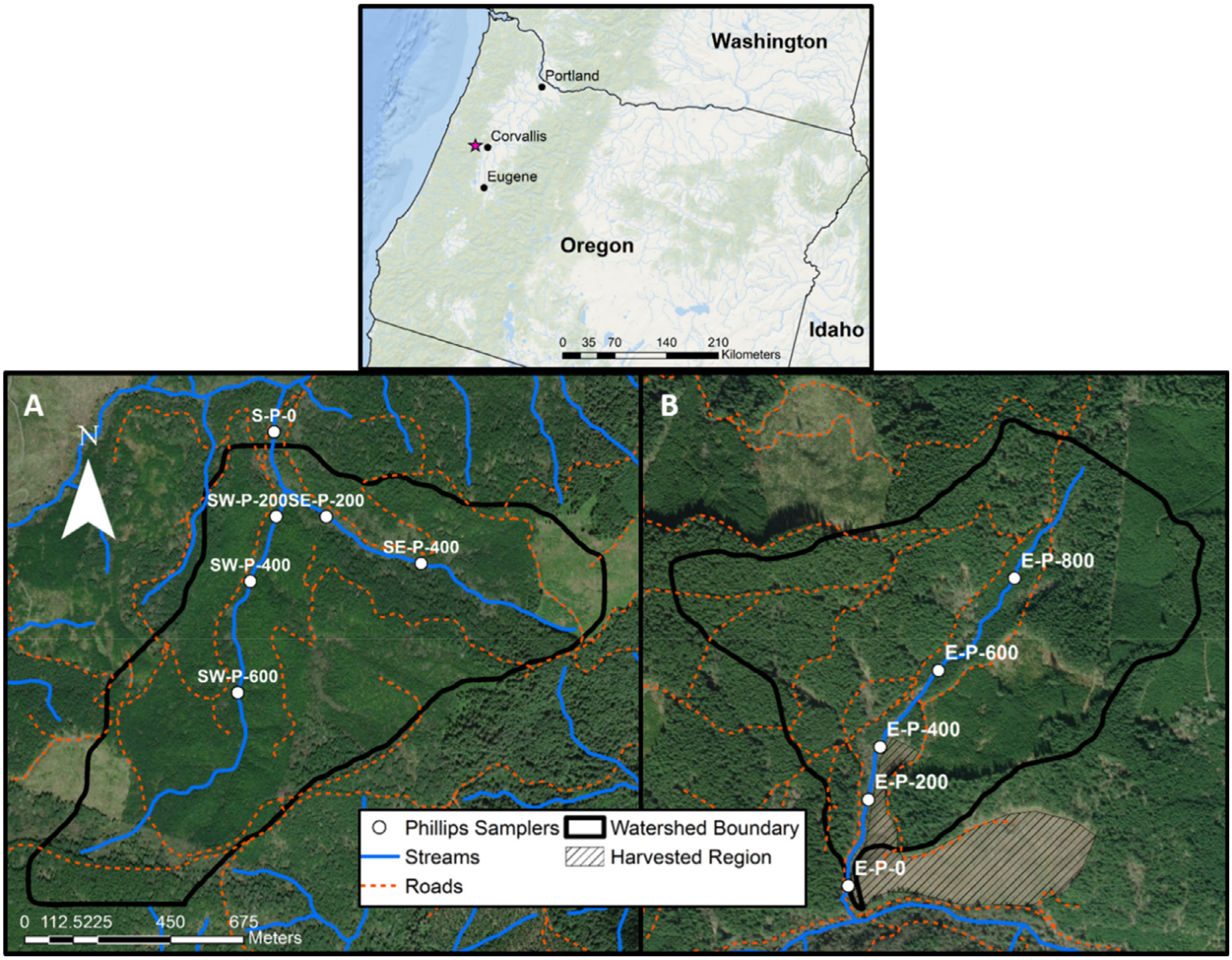

Fig. 1. Maps of the study location in Oregon and (A) the reference catchment (Scheele Creek) and (B) harvested catchment (Enos Creek), indicating the locations of the Phillips samplers, streams, roads, and harvested area in both catchments.

operations were completed using a yarder (skyline logging technique) and log loaders. Felled trees were hauled out of the catchment using a graveled road network. The overall road network was 7.8 km long in the reference catchment and 8.3 km long in the harvested catchment. All roads in both study catchments were legacy roads (> 40–50 years old), which were maintained with current BMPs, including addition of rock during times of hauling, no hauling during extreme wet weather, periodic cleaning of ditches, re-grading after periods of use, and the use of ditch relief culverts to divert water from ditches to the forest floor where it can infiltrate and be filtered, reducing road sediment delivery to streams. The sum distance of roads within ~61 m (200 ft) of the stream, which was an approximation for the sediment delivery length (Megahan and Ketcheson, 1996; Wemple et al., 1996), was 1.4 km (18.4%) in the reference catchment and 2.0 km (24.2%) in the harvested catchment. Roads in the harvested catchment (Enos) were predominantly graveled, whereas roads in the reference catchment (Scheele) were mostly composed of compacted fine sediments. Additionally, the harvested catchment (Enos) had two road crossings upstream of the outlet, with ~36 in. (91 cm) culverts. Comparatively, the forested catchment (Scheele) did not have any road crossings upstream of the outlet.

2.2. Base hydrometric and water quality data

We collected manual stream discharge measurements from each stream to develop rating curves, which enabled a continuous record of discharge from automated measurements of stage. Measurements were collected approximately monthly during baseflow conditions, bimonthly during the rainy season, and during several additional high flow events using the salt dilution gauging procedure (Moore, 2005). In this protocol, electrical conductivity (EC) measurements were collected at one-second intervals using an YSI proDSS Multiparameter Water Quality Meter (YSI Incorporated, Yellow Springs, OH). A salt slug (1 kg of salt, 6 L of water) was prepared and poured ~50 m upstream of the EC sensor. Automated stage measurements were collected at the outlet of each catchment using pressure transducers (Junior Edge Levelogger, ± 0.1% FS, Solinst Canada Ltd, Georgetown, ON) logging at 15 min intervals. Atmospheric pressure was logged at 15 min intervals to compensate water level readings (Barologger Edge, ± 0.05 kPa, Solinst Canada Ltd, Georgetown, ON).

Unfortunately, two high flow events altered channel morphology at the Enos Creek catchment outlet during the first winter—the first of these occurred in late December 2016 and dropped the stream elevation by about 30 cm, while the second occurred in late February 2017 and dropped the stream elevation by about 5 cm. Due to this morphologic change, the Enos Creek rating curve could not be applied to stage measurements for this time period. Since the discharge at Enos Creek and Scheele Creek behaved similarly during the second winter, a linear regression (r^2 = 0.64) based on the second winter was applied to generate a record of discharge at Enos Creek during the first winter.

Meteorological stations were established within each catchment (HOBO U30, Onset Computer Corporation, Bourne, MA). The stations collected data (15-min intervals) on precipitation, air temperature, relative humidity, net radiation, barometric pressure, wind speed, and soil moisture at 10 cm, 30 cm, and 60 cm depths. Additionally, *in situ* water samplers (Teledyne ISCO 6712) were deployed to capture stream water samples during large storm events at the outlets of both catchments. We captured two major events, which occurred on Oct. 14–21, 2016 and Nov. 11–13, 2016. The ISCOs were manually started prior to each storm event and collected ~1 L samples in the centroid of flow and about midway between the streambed and water surface every four hours. Following each event, the samples were collected, filtered, and dried in the lab. Suspended sediment was weighed and the total volume of water was measured in each sample to enable calculation of suspended sediment concentrations (SSCs) during these high flow events.

Finally, soil unsaturated hydraulic conductivity (K) was measured to characterize the potential for water to infiltrate and flow through the soils at each site. We measured K using Mini Disk Infiltrometers (Meter Group, Inc., Pullman, WA) in September 2017 at 24 plots on both the forested and harvested hillslopes. For each of the forested and harvested hillslopes, measurements were taken along eight transects, which were perpendicular to the stream and spaced approximately 75 m apart. Each transect had a measurement at the summit, backslope, and toeslope (Wysocki et al., 2011; Schoeneberger et al., 2012). Given the length of the hillslopes, these measurements were approximately 25 m apart. Prior to measurements, we set the suction on the Mini Disk Infiltrometer to 1.5 cm and added a thin layer of sand over the soil to improve contact between the base of the infiltrometer and the mineral soil. If an organic layer was present, the layer was removed so that the infiltrometer was in contact with the top of the mineral soil layer.

2.3. Silt fence installation, sample collection, processing, and analysis

To address the objective of the effectiveness of riparian buffers at mitigating sediment movement from harvested hillslopes to streams, we installed 36 silt fences in summer 2017. Specifically, 12 silt fences were installed in each of three site types, including: (a) along the harvested hillslope just outside the riparian buffer, (b) within the riparian buffer near the stream edge, and (c) in the riparian area at the base of an unharvested, reference hillslope. At each site type, silt fences were evenly spaced across the hillslope approximately every 50 m.

Each silt fence was approximately one square meter. For each fence, a 48″ x 56″ piece of fabric was cut from Lumite Weed Barrier fabric. At each site, the long end (56″) of the fabric was placed parallel to hillslope contours and eight inches of fabric were folded up from each of the short ends and the downslope side of the fabric to form the silt fence walls; these walls were secured vertically using rebar and wire. Finally, 8 gauge, 8″ x 2″ fabric staples were used to secure the fabric as tightly to the ground as possible. Close attention was given to the front lip (facing upslope) of each fence. In doing this, fabric staples were placed approximately six inches apart along the entire interface and hammered them into the ground until the fabric was flush with the mineral soil directly upslope. When an organic layer was present, the organic layer directly upslope of the fence was gently peeled up while the fence and interface were installed and then replaced, overlapping with the fence interface for about half an inch.

Sediment was collected from the silt fences in Whirl-Pak bags after each of five collection periods, approximately monthly through winter 2017–2018. When conditions were wet and the sediment was muddier, trowels and spoons were used to help collect all of the sediment out of the fence. When conditions were drier, a brush and dustpan were used to help collect all the sediment. After collection, all sediment samples were returned to the laboratory and oven-dried for 24 h at 40 °C. Subsequently, the samples from each fence were sieved to a < 2 mm grain size fraction and weighed. We performed an ANOVA and post hoc Tukey HSD tests to determine if the sediment masses were different between the transects (forested, riparian buffer, harvest).

2.4. Source sediment collection and processing

At the outset of the study, we collected 134 sediment samples from potential source areas across the two study catchments. The potential source areas included representative hillslopes, streambanks, and roads from both catchments. We returned all samples to the laboratory to characterize each of the major sources using chemical fingerprinting analyses. Specifically, we collected six hillslope samples from each of five transects within each catchment for a total of 30 hillslope samples from each of the two study catchments. Transects ran perpendicular to the streams and were spaced ~200 m apart. Sample sites were spaced approximately 15 m apart within each transect. At each sample site, we collected the soil samples from the upper five centimeters of the soil profile after removing any harvest residuals and the organic layer. This depth was considered reflective of the sediment that theoretically could be transported from the hillslope to the stream if surface erosion was a dominant sediment transport mechanism.

Streambank samples were collected from 20 locations in each of the two study catchments. Sample sites were spaced ~50 m (thalweg distance) apart starting at the stream outlet and moving upstream to the channel head. At each sample site, we collected samples vertically across the entire streambank profile by scraping the exposed streambank surface to approximately one centimeter lateral depth with a hand trowel from the top of the profile to the streambed.

Road samples were collected from sections of road that were within 100 m of each of the two study streams. We collected 19 sediment samples from each road network in each of the two study catchments (38 total samples). Sample sites were evenly spaced, approximately 50 m apart moving from the most downstream section of road to the upstream section of road. Samples were again collected with a hand trowel from the top one centimeter at locations where loose sediment particles were observed. Samples from all three source locations were placed in Whirl-Pak bags and refrigerated at 4 °C in the laboratory prior to processing and chemical analyses.

In the lab, all source sediment samples were placed in paper bags and oven-dried for 24 h at 40 °C and sieved to a < 2 mm fraction. Samples were lightly ground using a mortar and pestle to break apart soil aggregates and placed on a SampleTek Model 200 Vial Rotator at low speed for 24 h. After disaggregation of the soils, the < 63 μm fraction was separated and analyzed to more closely match the particle sizes of the suspended sediment samples.

2.5. Suspended sediment collection and processing

Time-integrated suspended sediment samplers (Phillips samplers) were deployed in each stream (5 in the harvested catchment, 6 in the reference catchment; Fig. 1) to capture in-stream suspended sediment samples. The Phillips samplers were constructed following the original specifications with a 4 mm diameter inlet and outlet tube, and a 1 m length of PVC pipe with a 98 mm inside diameter resulting in an internal cross-sectional area of 7543 mm^2 (Phillips et al., 2000).

Samplers were deployed from October 2016 to April 2018 in the centroid of flow and about midway between the streambed and water surface. While Phillips et al. (2000) recommended installation at a height equal to 40% of the stream depth, the midway point was a close approximation due to the shallow depths of these headwater streams. In each stream, the samplers were installed approximately 200 m apart (thalweg distance), equally spaced along the streams to capture potential longitudinal variability (Fig. 1). We collected water and sediment samples from each Phillips sampler on ten occasions, spread evenly across the two winter seasons, 2016–2017 and 2017–2018. During collection, we plugged the rear outlet to ensure no loss of sample then poured all water and suspended sediment in the sampler into 5-gallon buckets and immediately brought back to the lab. In the lab, all samples were centrifuged with a Thermo Sorvall Legend XTR Centrifuge at 3500 rpm for ten minutes to separate the water and suspended sediment. After centrifuging, the majority of the water was slowly suctioned off and disposed. The remaining sediment and small amount of water in each sample was then poured into Falcon tubes and oven-dried for 48 h at 40 °C.

2.6. Chemical analyses

All of the sediment source samples from the hillslope, streambank, and road were analyzed to identify a unique chemical fingerprint for each source area, which was necessary to enable the use of a mixing model to quantify the proportions of each of the suspended sediment samples that were derived from each source. Specifically, we used three main chemical analyses of the suspended sediment to determine the source. First, total carbon (TC) and total nitrogen (TN) concentrations were determined via dry combustion in a Thermo FlashEA 1112 Series. Second, stable isotopes of nitrogen ($\delta^{15}N$) and carbon ($\delta^{13}C$) were measured with a Thermo DeltaPlusXL mass spectrometer. Third, iron (Fe), potassium (K), and calcium (Ca) geochemistry was determined by first performing Mehlich extractions, followed by sample analysis using an ICP-OES: PerkinElmer 2100 DV.

2.7. Statistical analyses

To quantify the relative contributions from each of the potential source areas (hillslopes, streambanks, and roads), an endmember mixing model analysis and Monte Carlo simulation were used. All statistical analyses were run using the Sediment Source Assessment Tool (Sed_SAT), an open-source USGS program based in R and Microsoft Access (Gorman Sanisaca et al., 2017). The following steps (described in detail below) were used to develop the mixing model: (1) detection of outliers, (2) correction of tracers in each source type for differences in organic carbon content, (3) bracket testing of the organic-corrected samples for each tracer to determine if any tracers were not behaving conservatively with transport, (4) forward stepwise linear discriminant function analysis to determine which tracers were best at distinguishing potential source areas, and (5) mixing model and Monte Carlo simulation to determine the contributions from each of my sediment sources and the error about these calculations.

2.7.1. Outlier test

Outliers were removed from each tracer in each individual source group so that a single, potentially erroneous sample measurement (due to a sampling or machine error) would not result in an incorrect characterization of a source area's average chemistry. To do this, each chemical tracer in each sediment source group was tested for normality using the Shapiro-Wilk W test at a 95% confidence level. Any tracer that was not normally distributed was transformed using the Tukey ladder of powers, which transforms the data using six different methods, including the square, square root, cube root, inverse, inverse square root, and logarithm functions. The transformed distributions were again tested for normality using the Shapiro-Wilk W test. We then selected the transformation that yielded the lowest p-value. After all tracer datasets were transformed to normal distributions, data points outside of three standard deviations of the mean were flagged as outliers and discarded for all subsequent analyses.

2.7.2. Organic carbon content corrections

Many chemical tracers have an affinity to organic matter, although the strength of this relationship is often unpredictable and can vary between catchments (Collins et al., 2017). We used a regression analysis to remove the effects of any significant differences in organic content between the source sediment and fluvial sediment data sets:

$$C_n = \{Ti_{i(n)} - [(S_j - CF) \times m]\}\wedge \quad (1)$$

where C_n = tracer after organic correction (untransformed if transformation was applied), $Ti_{i(n)}$ = original value of tracer i in source group n (transformed if applicable), S_j = organic content value of sample j, CF = mean organic content in target samples (transformed if applicable), m = slope of regression line, and ˆ = if transform was applied, the tracer is then untransformed.

The organic matter correction was only applied in instances where the slope of the regression line was found to be significant ($p < .05$). After adjustment, data was corrected to account for the bias resulting from transforming the data. Standard bias correction factors used in this step for each potential distribution transformation were the same as in Gellis et al. (2015) .

2.7.3. Bracket test

The organic-corrected tracer data was analyzed for conservative transport, which was completed by ensuring that the tracer values of the suspended sediment were within the maximum and minimum tracer values of the potential source areas. Any tracer that did not satisfy the following constraint was discarded from all subsequent analyses:

$$min(Y_i) - 0.1 * min(Y_i) < x_i < max(Y_i) - 0.1 * max(Y_i) \quad (2)$$

where x_i = suspended sediment tracer for a specific tracer i and Y_i = vector of all source concentrations for specific tracer i.

2.7.4. Forward stepwise linear discriminant function analysis

A forward stepwise linear discriminant function analysis (DFA) with a significance level of 0.05 was used to determine the linear combination of tracers that best separated the potential sediment sources with unique chemical fingerprints. After running the DFA, weighting factors (W_i) were applied (Eq. (4)) to tracers that correctly classified potential source areas more frequently so they would have a greater influence on the mixing model:

$$W_i = \frac{P_i}{P_{opt}} \quad (3)$$

where P_i = percent of source samples classified correctly using tracer i, P_{opt} = percent of source samples classified correctly using tracer with lowest P_i.

2.7.5. Mixing model

The tracers and data points remaining after each of these steps were used in the mixing model, along with their assigned weighting parameter from the DFA. The following equation was used to calculate the proportion of each potential source area in each suspended sediment sample:

$$\sum_{i=1}^{n} \left\{ \frac{[C_i - (\sum_{i=1}^{m} P_s S_{si})]}{C_i} \right\}^2 W_i \quad (4)$$

with $\sum_{s=1}^{n} P_s = 1$

where C_i = concentration of tracer i in the target samples, P_s = optimized percentage of contribution of source type s, S_{si} = mean concentration of tracer i in source type s (after organic content correction, if applicable), W_i = weighting factor for tracer i, n = number of tracers comprising the optimum composite fingerprint, and m = number of source sediment types.

A Monte-Carlo simulation (n = 1000 iterations) was run, in addition to the mixing model equation, to quantify the potential variability in source area contributions that would still result in a matching suspended sediment chemistry.

3. Results

3.1. Hydrometric and water quality data

Precipitation was ~1.9-times greater during the first data collection

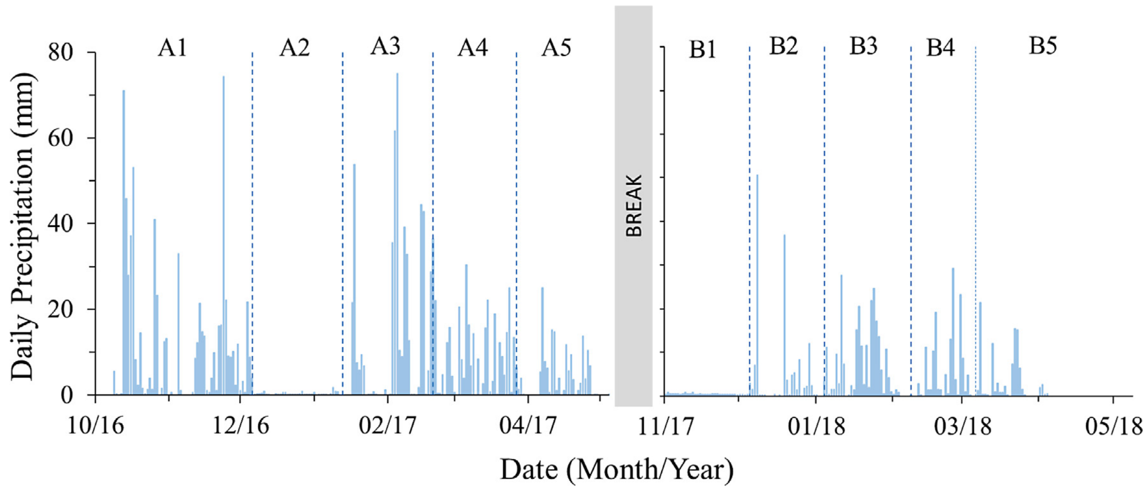

Fig. 2. Precipitation values collected from a tipping bucket rain gauge at the Enos Creek catchment are shown above from both wet seasons. Each wet season is divided into five periods of time based on the collection dates of suspended sediment from the Phillips samplers. The labels assigned to each collection period (A1 = first wet season (A), first collection (1)) will be used throughout subsequent analyses.

Table 1
Descriptive statistic summaries for precipitation and discharge across the ten suspended sediment collection periods at both Enos Creek and Scheele Creek.

Collection period	Total precipitation (mm)	Maximum daily precipitation (mm)	Cumulative wet season precipitation (mm)	Enos (harvested)		Scheele (reference)	
				Average daily discharge ($m^3 s^{-1}$)	Peak daily discharge ($m^3 s^{-1}$)	Average daily discharge ($m^3 s^{-1}$)	Peak daily discharge ($m^3 s^{-1}$)
A1	699.1	74.3	699.1	0.081	0.497	0.090	0.751
A2	9.7	1.7	708.8	0.119	0.196	0.150	0.273
A3	544.6	75.0	1253.4	0.186	0.459	0.260	0.687
A4	319.2	30.4	1572.6	0.149	0.275	0.192	0.394
A5	155.8	25.0	1728.4	0.047	0.157	0.073	0.157
B1	20.4	3.0	20.4	0.060	0.129	0.074	0.148
B2	143.2	50.6	163.6	0.051	0.067	0.051	0.069
B3	232.2	27.6	395.8	0.074	0.144	0.085	0.228
B4	139.2	29.2	535.0	0.069	0.153	0.060	0.131
B5	95.4	21.4	630.4	0.075	0.107	0.051	0.071

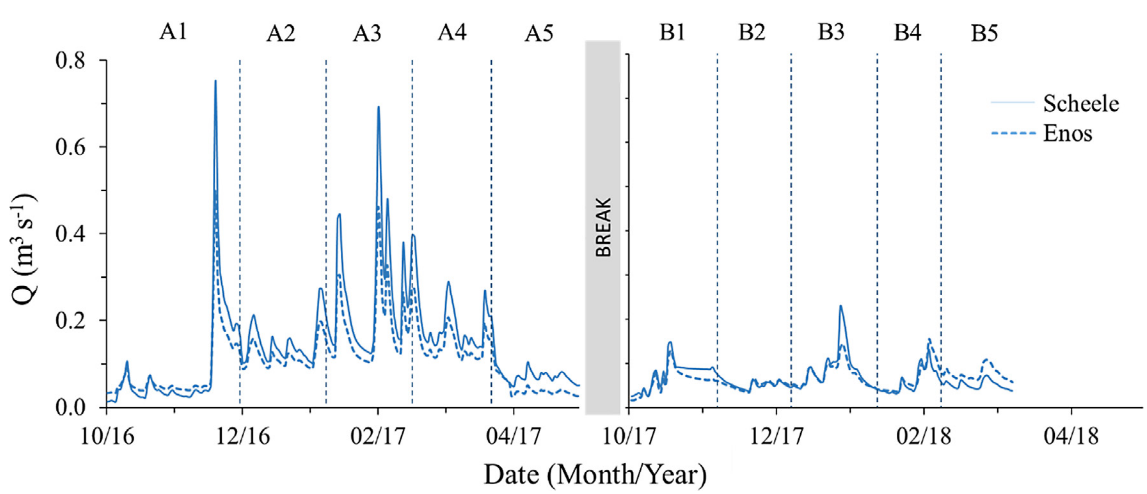

Fig. 3. Discharge values collected from applying rating curves to pressure transducer data from both Enos and Scheele from both wet seasons. Each wet season is divided into five periods of time based on the collection dates of suspended sediment from the Phillips samplers.

period of the study in 2016–17 (October–April: 1728.4 mm) compared to the second data collection period in 2017–18 (October–April: 918.6 mm) (Fig. 2). Moreover, a rare snow event during the second collection period in winter 2016–17 (A2; December 12, 2016–January 13, 2017) likely resulted in an underestimate of precipitation (Table 1), as tipping bucket rain gauges generally have poor reliability in such conditions (Grossi et al., 2017). The precipitation in winter 2016–17 was similar (0.6% greater) to the 30-year mean (1981–2010) for the region (PRISM Climate Group, 2004) while the precipitation received in the winter 2017–18 was 63.3% below normal.

Hydraulic conductivity measurements indicated that precipitation had a high likelihood of infiltrating and percolating to depth in the soils at our sites. Specifically, hydraulic conductivity was 1.1 ± 0.5 (*SE*) cm hr^{-1} in the reference catchment. Comparatively, mean hydraulic conductivity was slightly higher on the forested hillslopes of the harvested catchment (1.5 ± 0.5 cm hr^{-1}), but statistically there was no evidence for a difference between the catchments (p = .51). However, there was suggestive evidence (p = .06) that hydraulic conductivity was greater on the harvested hillslopes (2.3 ± 0.4 cm hr^{-1}) compared to the reference hillslopes.

Average daily discharge in the harvested catchment (Enos) was 0.12 $m^3\ s^{-1}$ during winter 2016–2017 and 0.07 $m^3\ s^{-1}$ during winter 2017–2018 (Table 1). During the winter collection periods of A1 (October 8–December 6, 2016) and A3 (January 13–February 10, 2017) we measured the highest peak daily discharges (0.50 and 0.46 $m^3\ s^{-1}$, respectively)—peak flows were consistently higher in the winter of 2016–17 than in 2017–18 (Fig. 3). Not surprisingly, the late winter collection period A3 had the highest daily average discharge (0.019 $m^3\ s^{-1}$) relative to all other collection periods.

Average daily discharge at the forested reference catchment (Scheele) was 0.13 $m^3\ s^{-1}$ during the first winter and 0.06 $m^3\ s^{-1}$ during the second winter of the study. Likewise, the collection periods A1 and A3 had the highest peak daily discharges (0.75 and 0.69 $m^3\ s^{-1}$) and A3 had the highest average daily discharge (0.26 $m^3\ s^{-1}$). Similar to the harvested catchment, peak daily discharges at the reference catchment were higher in winter 2016–17 than 2017–18 (Fig. 3).

Surprisingly, mean suspended sediment concentrations were ~3.8-times greater in the reference catchment across the two measured high flow events (34.3 ± 8.5 (*SE*) mg L^{-1}) than in the harvested catchment (9.1 ± 1.5 mg L^{-1}). Moreover, while 98% of the samples from the harvested catchment and 87% of samples from the forested catchment had concentrations less than 60 mg L^{-1}, the seven greatest concentrations (up to 295 mg L^{-1}) were observed in the reference catchment (Fig. 4). Thus, during these precipitation events, there was a greater propensity for elevated suspended sediment concentrations at the outlet of the forested, reference catchment.

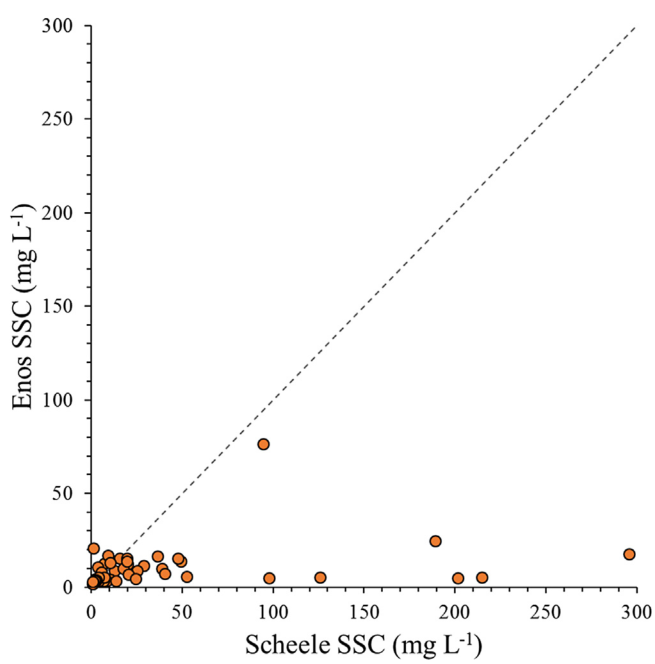

Fig. 4. A comparison of suspended sediment concentrations from the unharvested reference catchment (Scheele) and the harvested catchment (Enos) during two large storm events (October 14–21 and November 11–13, 2016). A 1:1 dashed lined is provided to aid comparison.

3.2. Hillslope sediment mobility

The mean sediment mass collected in the silt fences during winter 2016–17 in the general harvest area at the harvest/buffer edge (96.5 ± 57.0 (*SE*) g) was 10.6-times greater than the sediment collected in the riparian buffer (9.1 ± 1.9 g), and 4.6-times greater than the sediment collected on the unharvested, reference hillslope (21.0 ± 3.3 g; Fig. 5). During each sediment collection period, we collected at least three-times more sediment from the general harvest area hillslopes than from the hillslopes of the unharvested reference and riparian buffer. Statistically, there was strong evidence that the sediment mass mobilized on the general harvest area hillslopes was greater compared to both the riparian buffer (p < .001) and reference hillslope transects (p = .005). Comparatively, there was no evidence the sediment mass was different between the riparian buffer and the reference site (p = .87).

3.3. Mixing model inputs

We did not identify any outliers in the sediment chemistry data for each source area in the harvested catchment (Enos); therefore, we did not remove any samples from the data. Alternatively, we identified one outlier in the $\delta^{15}N$ from the hillslope source area in the reference catchment (Scheele). As such, this sample was discarded for all subsequent analyses.

Six samples were corrected for organic carbon content prior to subsequent analyses because there was strong evidence for a relationship between the sample tracer and percent total carbon; this was assumed to equal total organic carbon because there were no inorganic carbonates in the soils. Thus, three samples from the harvested catchment were corrected, including: (a) hillslope Ca, (b) road K, and (c) streambank TN. Three samples from the reference catchment were also corrected, including: (a) road TN, (b) road Ca, and (c) streambank TN.

At the harvested catchment, the suspended sediment samples were not within the potential source area ranges for organic carbon, calcium, and iron. At the forested catchment, suspended sediment samples were not within source area ranges for iron. These tracers were discarded for further analyses.

Results from stepwise Discriminant Function Analysis (DFA) indicated all remaining tracers could be used to distinguish between sediment source areas in the harvested catchment. However, in the reference catchment, there was no evidence organic carbon and total nitrogen could be used to distinguish sediment source areas. Thus, four tracers were usable for the mixing model at both the harvested catchment (total nitrogen, $\delta^{13}C$, $\delta^{15}N$, K) and the reference catchment ($\delta^{13}C$, $\delta^{15}N$, Ca, K). At the harvested catchment, the DFA successfully classified 85% of the sediment source samples in their correct source while at the forested catchment the DFA successfully classified 91% of the sediment source samples. Prior to the running of the mixing model, discriminatory weighting factors were applied to each of these tracers based upon the percentage of source area samples they could correctly identify in the DFA.

3.4. Catchment suspended sediment sources

Streambanks were the dominant source of suspended sediment in both the harvested and reference catchment. In both catchments, streambanks contributed more than six-times the suspended sediment to the streams relative to the hillslopes and roads. Specifically, streambanks contributed 90.2 ± 3.4 (*SE*) % of the suspended sediment in the harvested catchment (averaged across all collection periods) and 93.1 ± 1.8% of the suspended sediment in the forested reference catchment (Table 2). In the harvested catchment, hillslopes were the second largest contributor of suspended sediment (7.1 ± 3.1%), while roads contributed the least sediment (3.6 ± 3.6%). Comparatively, in the reference catchment, hillslopes were the second largest contributor

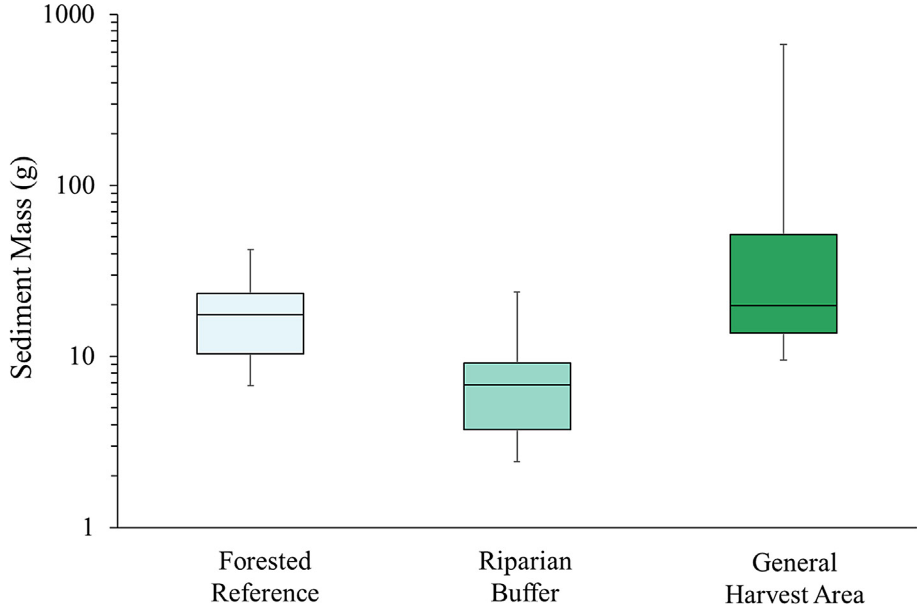

Fig. 5. Sediment mass collect from silt fences across five collection periods from hillslope transects in forested reference, riparian buffer, and general harvest area sites.

of suspended sediment (6.9 ± 1.8%), followed by negligible contributions from roads (0.0 ± 0.0%). Predicted values from the mixing model equation were always within 1% of the means of the Monte Carlo simulation; additionally, the standard deviations of the Monte Carlo simulations were never more than 3%, meaning that the simulation did not produce many outcomes in which the magnitude of contributions from different source areas substantially varied (Table 2).

3.5. Suspended sediment sources upstream and downstream of harvest

Using the suspended sediment samples from the spatially distributed Phillips samplers in the harvested catchment, the principal source of sediment both upstream and downstream from the harvested area was found to be the streambanks. However, the streambanks contributed 95.3 ± 2.2% of the suspended sediment upstream of the harvest but, decreased downstream of the harvest to 90.2 ± 3.4% (Fig. 6). In comparison, the contribution of suspended sediment from roads upstream of the harvest was negligible but, rose to 3.6 ± 3.6% below the harvest. This was expected as the roads in the lower portion of the catchment were closer to the stream, near steeper slopes, and used more frequently. Finally, the hillslopes proportionally contributed more sediment to the stream downstream of the harvest (7.1 ± 3.1%) relative to upstream of it (4.7 ± 2.2%).

3.6. Temporal suspended sediment sources

During winter 2016–17 at the harvested catchment, the beginning of the rising limb (A1; October 8, 2016–December 6, 2017) had a substantially higher proportion of hillslope inputs (27.4%; Fig. 7) than any of the subsequent collection periods; precipitations inputs were also higher (699.1 mm) than in any other collection period. The next highest proportional contribution from hillslopes during any collection period was just 9.3% (A4; February 10–March 27, 2017; Fig. 7). Roads made only a single substantial contribution of sediment to the stream during the A5 (March 27–May 5, 2017) collection period (20.6%). During the winter 2017–18, proportional contributions of sediment to the stream where consistent within endmembers, with streambank contributions ranging from 93.2 to 97.2%, hillslope contributions ranging from 2.8 to 6.8%, and no contributions from roads.

4. Discussion

Analysis of our sediment source fingerprinting data, combined with hillslope sediment masses and in-stream suspended sediment concentrations, suggest that current BMPs (i.e., retention of forested buffers around water bodies, limited allowable cutblock sizes, restricted harvest operations near water bodies or on steep slopes, use of lighter and longer reach machinery, and proactive road building, use, and maintenance activities) were relatively effective at mitigating suspended sediment delivery to the stream after forest harvesting in an Oregon Coast Range catchment. In particular, sediment fingerprinting indicated that forest roads only contributed 4% of the annual suspended sediment in the harvested catchment. Comparatively, there was no evidence from the sediment fingerprinting data of any sediment contribution from roads in the reference stream. Given that the mean suspended sediment concentrations were ~3.8-times greater in the reference catchment than in the harvested catchment, the overall sediment mass from roads in the harvested catchment was likely quite low. This finding was surprising given that unpaved forest roads are often hydrologically connected to the stream network and are nearly impervious surfaces that can lead to increased overland flow and sediment delivery to streams (Bilby et al., 1989; Ziegler et al., 2001; Coe, 2006). For example, using carbon-13 stable isotopes Bravo-Linares et al. (2018) illustrated that 20–98% of sediment in three catchments in south-central Chile originated from unpaved roads. However, the low sediment contributions from roads to streams in our study are consistent with other recent research in the Pacific Northwest. For example, lower suspended sediment concentrations relative to historical studies of forest road construction and use in headwater catchments have been attributed to improvements in road construction and maintenance (Reiter et al., 2009; Arismendi et al., 2017).

Regardless, the slightly higher proportion of suspended sediment from roads in the harvested stream was likely due to repeated use of the roads from heavy vehicles, during log hauling or road maintenance. Comparatively, the roads in the reference catchment were used infrequently and predominantly by recreationists. The bulk of sediment from roads occurred during April 2017 (A5 collection period), which coincided with road maintenance activities (grading) and increased logging truck traffic due to harvesting operations at an adjacent catchment during this collection period. Previous studies have also illustrated a

Table 2
A summary of all the mean and standard deviations for each source area contribution, at each location, at each collection period (A1 through B5) of the study. H represents the hillslope, R the road, and S the streambank. All entries with a “-” instead of a numerical value are due to an insufficient mass of sediment being collected from the Phillips sampler for chemical analysis.

Catchment	Sampler location	Sediment source	Sample collection period									
			A1	A2	A3	A4	A5	B1	B2	B3	B4	B5
Enos (harvested)	P0	H	27.4 ± 1.3	–	0.2 ± 0.8	9.3 ± 1.3	0.0 ± 0.0	5.9 ± 1.4	6.8 ± 1.4	4.2 ± 1.5	–	2.8 ± 1.5
		R	0.0 ± 0.0	–	0.0 ± 0.0	0.0 ± 0.0	28.6 ± 1.2	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	–	0.0 ± 0.0
		S	72.6 ± 1.3	–	99.8 ± 0.8	90.7 ± 1.3	78.4 ± 1.2	94.1 ± 1.4	93.2 ± 1.4	95.8 ± 1.5	–	97.2 ± 1.5
	P200	H	9.3 ± 1.3	3.8 ± 1.5	12.6 ± 1.3	23.0 ± 1.3	9.8 ± 1.2	–	–	–	–	5.8 ± 1.4
		R	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	–	–	–	–	0.0 ± 0.0
		S	90.7 ± 1.3	96.2 ± 1.5	87.4 ± 1.3	77.0 ± 1.3	90.2 ± 1.2	–	–	–	–	94.2 ± 1.4
	P400	H	–	7.0 ± 1.4	2.5 ± 1.4	–	16.7 ± 1.3	0.2 ± 0.8	0.4 ± 1.0	8.6 ± 1.4	5.6 ± 1.4	–
		R	–	0.0 ± 0.0	0.0 ± 0.0	–	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	–
		S	–	93.0 ± 1.4	97.5 ± 1.4	–	83.3 ± 1.3	99.8 ± 0.8	99.6 ± 1.0	91.4 ± 1.4	94.4 ± 1.4	–
	P600	H	16.4 ± 1.3	–	8.8 ± 1.3	1.1 ± 1.4	–	1.7 ± 1.4	2.6 ± 1.6	0.0 ± 0.1	2.5 ± 1.4	–
		R	0.0 ± 0.0	–	0.0 ± 0.0	0.0 ± 0.0	–	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	–
		S	83.6 ± 1.3	–	91.2 ± 1.3	98.9 ± 1.4	–	98.3 ± 1.4	97.4 ± 1.6	100.0 ± 0.1	97.5 ± 1.4	–
	P800	H	4.6 ± 1.4	–	–	–	–	–	–	–	8.3 ± 1.4	–
		R	0.0 ± 0.0	–	–	–	–	–	–	–	0.0 ± 0.0	–
		S	95.6 ± 1.4	–	–	–	–	–	–	–	91.7 ± 1.4	–
Scheele (reference)	P-400	H	–	9.6 ± 1.1	0.0 ± 0.0	–	–	–	–	6.5 ± 2.2	0.0 ± 0.0	–
		R	–	0.0 ± 0.1	0.0 ± 0.0	–	–	–	–	0.7 ± 2.1	0.0 ± 0.0	–
		S	–	90.4 ± 1.1	100.0 ± 0.0	–	–	–	–	92.8 ± 1.1	100.0 ± 0.0	–
	P-200	H	–	10.2 ± 1.1	–	–	13.2 ± 1.1	5.0 ± 1.1	–	0.0 ± 0.1	0.1 ± 0.6	0.0 ± 0.1
		R	–	0.0 ± 0.0	–	–	0.0 ± 0.0	0.0 ± 0.0	–	0.0 ± 0.1	0.0 ± 0.0	0.0 ± 0.1
		S	–	89.8 ± 1.1	–	–	86.8 ± 1.1	95.0 ± 1.1	–	100.0 ± 0.1	99.9 ± 0.6	100.0 ± 0.1
	0	H	1.8 ± 1.4	4.3 ± 1.2	12.7 ± 1.0	7.3 ± 1.1	14.7 ± 1.0	2.6 ± 1.3	6.2 ± 1.2	12.6 ± 1.1	0.1 ± 0.4)	–
		R	0.1 ± 0.2	0.1 ± 0.4	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.2	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.1	–
		S	98.2 ± 1.3	95.6 ± 1.1	87.3 ± 1.0	92.7 ± 1.1	85.3 ± 1.0	97.4 ± 1.2	93.8 ± 1.2	87.4 ± 1.1	99.9 ± 0.4	–
	P200	H	10.7 ± 1.1	–	22.2 ± 1.0	16.1 ± 1.0	–	0.5 ± 1.0	6.9 ± 1.3	9.6 ± 1.4	13.6 ± 1.1	–
		R	0.0 ± 0.0	–	0.0 ± 0.0	0.0 ± 0.0	–	0.1 ± 0.2	0.1 ± 0.8	0.1 ± 0.8	0.0 ± 0.0	–
		S	89.3 ± 1.1	–	77.8 ± 1.0	83.9 ± 1.0	–	99.4 ± 1.0	93.0 ± 1.1	90.3 ± 1.1	86.4 ± 1.1	–
	P400	H	–	11.5 ± 1.2	–	–	21.3 ± 0.9	–	0.0 ± 0.0	23.8 ± 1.0	11.0 ± 1.1	4.3 ± 1.3
		R	–	0.0 ± 0.0	–	–	0.0 ± 0.0	–	0.0 ± 0.0	0.0 ± 0.3	0.0 ± 0.0	0.1 ± 0.7
		S	–	88.5 ± 1.2	–	–	78.7 ± 0.9	–	100.0 ± 0.0	76.2 ± 1.0	89.0 ± 1.1	95.6 ± 1.1
	P600	H	–	18.6 ± 1.0	–	0.0 ± 0.0	–	–	–	16.1 ± 1.0	11.8 ± 1.1	–
		R	–	0.0 ± 0.0	–	0.0 ± 0.0	–	–	–	0.0 ± 0.0	0.0 ± 0.0	–
		S	–	81.4 ± 1.0	–	100.0 ± 0.0	–	–	–	83.9 ± 1.0	88.2 ± 1.1	–

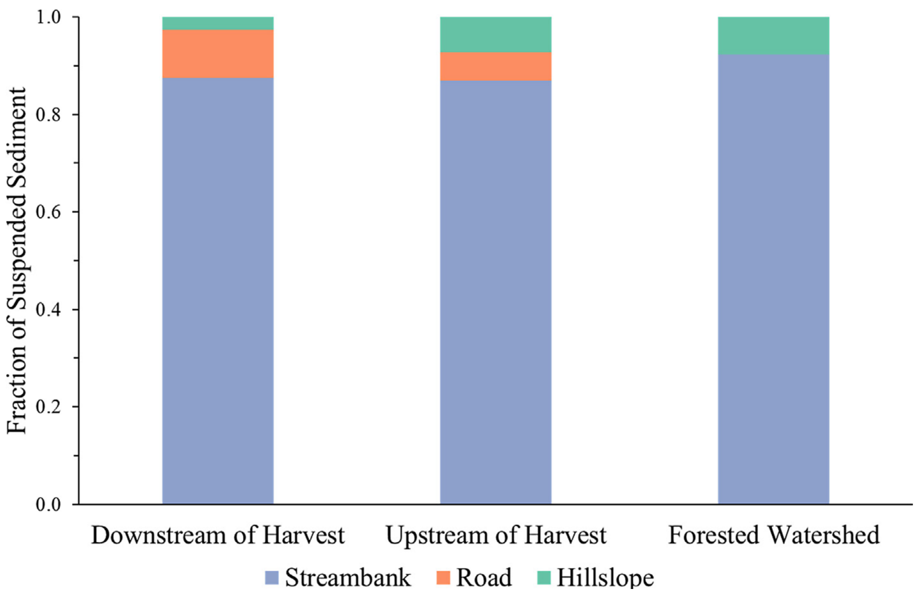

Fig. 6. Mixing model results for source area contributions downstream of the harvest (Enos Sampler P0), upstream of the harvest (Enos Sampler P600), and at the forested catchment (Scheele Sampler P0). Source contributions are averaged across all collection periods in which a sufficient mass of sediment was collected for chemical analysis.

greater supply of sediment (~2- to 100-times) on heavily used gravel roads relative to lightly used roads (Reid and Dunne, 1984; Megahan et al., 2001; Sheridan et al., 2006; van Meerveld et al., 2014; Sosa-Pérez and MacDonald, 2017).

Interestingly, while the overall contribution of sediment from roads was relatively low, our longitudinal analysis indicated an increasing proportion of sediment delivery to the stream from roads and hillslopes in the lower portions of the harvested catchment. This increase in sediment at the downstream portion of the harvested catchment may also be partially attributable to more vehicle traffic; however, the lower gauging site at the harvested catchment outlet was just downstream from a culvert and the only stream crossing in the catchment. Road-stream crossings and culverts have the potential to increase sediment delivery rates to streams by creating direct flow pathways from the road network to the stream, expanding drainage networks, and increasing areas susceptible to erosion (Wemple et al., 2001; Brown et al., 2013; Lang et al., 2018). However, the effects are typically dynamic and dependent on many site specific factors, including road slope, road surfacing, road maintenance, proximity and connectivity to the stream, traffic type and amount, and type of stream crossing (Luce and Black, 1999; Sheridan and Noske, 2007; Lang et al., 2018).

Our sediment source tracing data also indicated that the hillslopes contributed similar proportions of in-stream sediment in the harvested catchment (~7.1%) and reference catchment (~6.9%). This was an important finding, which illustrated the effectiveness of current BMPs, including limits to cutblock sizes (maximum 48.5 ha in Oregon) and the retention of streamside vegetation buffers, at mitigating sediment transport from the general harvest area to streams. This finding was also supported by the sediment mass data collected from silt fences in the general harvest area, riparian buffer, and an unharvested, reference hillslope, which suggested that sediment erodibility was elevated on the harvested hillslope due to the harvesting activity. However, there were no differences in sediment masses from the riparian buffer and the reference hillslopes. This was an important finding, as there have been few studies explicitly quantifying the proportional amount of sediment delivered to streams from the general harvest area, despite generally representing the largest area of disturbance associated with forest harvesting activity (Miller et al., 1996; Ampoorter et al., 2012). As a result, there remains uncertainties about the degree to which the general harvest areas and adjacent riparian areas act as sources or sinks for runoff and sediment transport (Croke et al., 1999; Wallbrink and Croke, 2002). Our findings are consistent with recent research showing that riparian areas may be effective at mitigating suspended sediment transport from harvested hillslopes to streams (Bywater-Reyes et al., 2018; Hatten et al., 2018; Puntenney-Desmond et al., 2020). However, in catchments with more friable lithologies (e.g., sandstone) there are

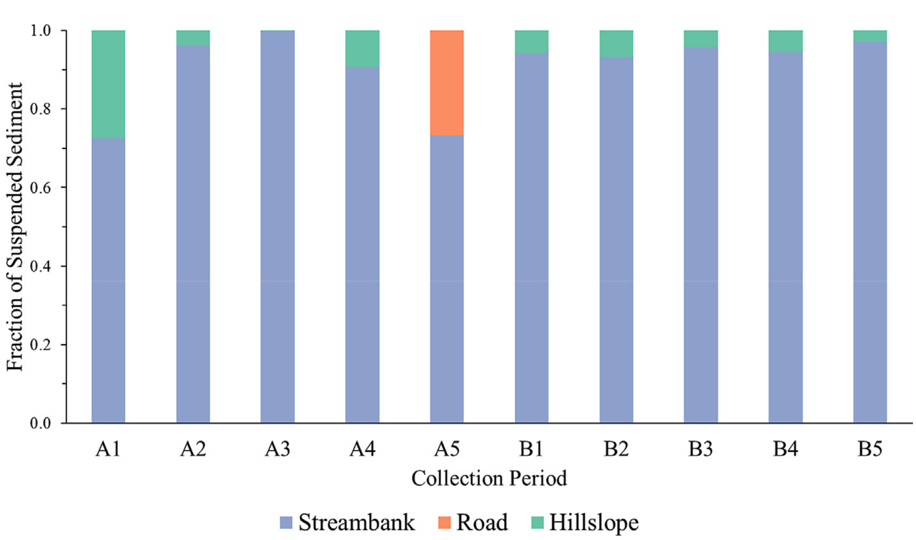

Fig. 7. Mixing model results for source area contributions at the outlet of Enos Creek catchment across the five collection periods in the first winter. For collection period A2, data from Sampler P200 was used in place of sampler P0 because an insufficient mass of sediment was collected for chemical analyses. For collection period B4, sampler P600 is used in place of sampler P0 for the same reason.

still instances where harvest units provide a source of sediment following contemporary forest harvesting (Macdonald et al., 2003; Motha et al., 2003; Bywater-Reyes et al., 2017), indicating the need to continue to improve our understanding of the processes and drivers of runoff and sediment production from general harvest areas into or through riparian buffers as a function of physiography.

Temporally, hillslope sediment supply was the highest during the first collection period (A1, Oct 8–Dec 6, 2016; 27.4%), which may be due to several factors. First, precipitation (695.2 mm) during this two month period was ~3.8-times greater than the mean precipitation of all of the other collection periods. In fact, five of the ten largest single day precipitation amounts (40.9–70.3 mm day^{-1}) during the study fell during this period. High precipitation inputs increase the potential for runoff and sediment erosion from localized hillslope areas (Mohr et al., 2013). Second, the first collection period occurred at the beginning of the rainy season in the region with only 17.2 mm of precipitation falling during the preceding four months. As, such there was likely an abundant supply of the most mobile and erodible hillslope sediment during this initial measurement period. During the remainder of the first winter and the second measurement period, hillslope inputs were consistently low (2–7%).

Interestingly, the primary source of sediment in both study catchments was the streambank. Specifically, the streambank contributed nine times more fine sediment to the streams in both study catchments relative to the roads and hillslopes combined. Qualitatively, the importance of inputs of sediment from streambank erosion was further evidenced by changes in channel morphology after two large rainfall events in December 2016 and February 2017, which produced streambank failure and widening of the channels. Such transitions from hillslope to bank sourced sediment throughout the course of a wet season have been observed elsewhere as available hillslope sediment supply is exhausted (Terajima et al., 1997; Whiting et al., 2005). Our findings are also consistent with others, who have previously identified streambank erosion, along with rapid mass wasting, as the primary sources of sediment supply in small forested streams of the Pacific Northwest (Hassan et al., 2005). Beschta (1979) noted that streambank scour was the most likely source of increased suspended sediment in an Oregon Coast Range stream after forest harvesting and debris removal from the stream channel, due to increased streamflow velocities.

Our observations are also in partial agreement with previous studies from Idaho, USA (Karwan et al., 2007), Georgia, USA (Fraser et al., 2012), North Carolina, USA (Voli et al., 2013), Virginia, USA (Gellis and Sanisaca, 2018), New Zealand (Basher et al., 2011), and Japan (Hotta et al., 2007), which all hypothesized that elevated sediment concentrations after forest harvesting were associated with scouring of channel banks or mobilization of channel-stored sediment during high flow events. However, a study by Schuller et al. (2013) in south-central Chile, which has a similar climatic regime as our study sties, illustrated that the relative contribution from the stream banks decreased by ~17% and 30% after forest harvesting in their two study catchments due to demonstrable increases in road and hillslope contributions.

While it was not surprising that streambanks were the dominant source of sediment in both our study catchments, the proportion of suspended sediment attributable to streambank sources (~90–93%) was much greater than has been observed elsewhere. For example, streambanks only accounted for 60–62% of stream sediment during high streamflow events in the Piedmont region of the southeastern U.S., where streams often transport large amounts of suspended sediment due to highly erodible streambanks combined with high precipitation intensities (Mukundan et al., 2010; McCarney-Castle et al., 2017). Additionally, streambanks accounted for just 32–51% of sediment inputs in a catchment in the central Canadian prairies used for agricultural purposes and with minimal riparian management area (Koiter et al., 2013). Similar to our study, the stream channel in one forested catchment in a study in south-central Chile was the source of ~85% of the sediment output—this was attributed to lesser contributions from the other potential sources due to dense vegetation on the hillslopes and limited road use (Schuller et al., 2013). The high variability in streambank sources across studies highlights the importance of local physiographic attributes, geomorphic processes, watershed use, and forest management activities in driving catchment sediment dynamics. We posit that the high proportion of streambank sediment in our study was likely due to a combination of factors, including (a) a proportionally small area of the catchment harvested, which is consistent with current forest management practices, (b) effectiveness of riparian management areas at mitigating hillslope sediment transport to streams, (c) road maintenance and low vehicle traffic, resulting in comparatively low road sourced sediment, and (d) relatively high stream transport capacity due to high channel slope and high annual precipitation in the region.

5. Conclusions

Suspended sediment remains a key water quality parameter of concern during forest harvesting operations (Anderson and Lockaby, 2011). As such, best management practices (BMPs) have been developed and implemented to minimize effects on water quality (Ice, 2004; Ice et al., 2010). Although there have been advances in BMPs, their efficacy at reducing water quality impacts remains uncertain. In our study, sediment source fingerprinting techniques indicated that BMPs were relatively effective at minimizing sediment delivery from roads and hillslopes following forest harvesting of a catchment in the Oregon Coast Range. While harvesting activity resulted in mobilization of hillslope sediments, the riparian buffer was effective at reducing sediment transport to the stream. Despite minimal effects from harvesting, our study was able to document road contributions coinciding with a period of road maintenance and increased logging traffic. In both the harvested and reference catchments, streambanks were the primary contributor of sediment, with the greatest sediment concentrations observed in the reference catchment. This highlighted the importance of sediment stored in-channel and the role of catchment lithology in driving the sediment regime. Given the growing demands on forest products and hydrologic ecosystem services, it remains important to continue to improve our understanding of the impacts of our forest management decisions to avoid unintended degradation of water quality and aquatic ecosystems. Our study has illustrated that sediment source tracing, combined with traditional procedures for investigating erosion and sediment transport to streams, can provide reliable information to inform forest watershed management.

CRediT authorship contribution statement

Aaron A. Rachels: Conceptualization, Methodology, Formal analysis, Investigation, Writing - original draft, Writing - review & editing, Visualization. **Kevin D. Bladon:** Conceptualization, Methodology, Validation, Investigation, Resources, Writing - original draft, Writing - review & editing, Visualization, Supervision, Project administration, Funding acquisition. **Sharon Bywater-Reyes:** Conceptualization, Methodology, Investigation, Writing - review & editing, Supervision, Project administration. **Jeff A. Hatten:** Conceptualization, Methodology, Resources, 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.

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