

Long-term response in nutrient load from commercial forest management operations in a mountainous watershed

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ABSTRACT

An increased emphasis on fuel management strategies to mitigate wildfire risks is raising the awareness and need for comprehensive forest management strategies that satisfy long-term water quantity and water quality needs. Forest management activities can alter the soil nutrient pools and affect the timing and magnitude of stream water quantity and quality. We investigated the effect of contemporary forest management activities, including clear-cutting and thinning, on water yield and stream nitrogen and phosphorus dynamics in a quarter-century-long (1992–2016) paired and nested watershed study in the interior Pacific Northwest, US. Monthly water samples were collected and analyzed for total Kjeldhal nitrogen (TKN), total available nitrogen (TAN), nitrate + nitrite ($\text{NO}_3 + \text{NO}_2$), total phosphorus (TP), and orthophosphate (OP) concentrations throughout the study period. Five years of pre-disturbance data, 4 years of post-road construction, 6 years of Phase I (PH-I) experimental post-harvest, and 9 years of Phase II (PH-II) operational harvest data were analyzed using a before-after, control-impact paired series (BACIPS) study design. We found statistically significant increases in stream $\text{NO}_3 + \text{NO}_2$ loading from the paired and nested watersheds following timber harvest treatments. In the case of OP, any increase in nutrient load was attributed to increases in streamflow, as OP concentrations remained near minimum detectable concentrations. Streamflow increased the greatest following clear-cut practices (33.4% at W1 during PH-I) with the largest response in stream $\text{NO}_3 + \text{NO}_2$ concentration (up to 0.33 mg-N L^{-1} at W1). In-stream $\text{NO}_3 + \text{NO}_2$ and OP concentrations were lower in the downstream cumulative watersheds, which was likely due to dilution and nutrient assimilation effects. Interestingly, the $\text{NO}_3 + \text{NO}_2$ concentration, streamflow, and loads of $\text{NO}_3 + \text{NO}_2$ and OP from the undisturbed control watershed also increased. This increase was, however, smaller than the harvested watersheds and likely driven by climate variability or subtle forest succession changes. In summary, we found that contemporary forest management activities increased stream $\text{NO}_3 + \text{NO}_2$ concentrations and loads following timber harvest activities, but these effects were also attenuated due to downstream uptake processes. Furthermore, relative to post-wildfire impacts, these nutrient increases are small and short-lived.

1. Introduction

Upland forested regions provide a wide range of ecosystem services ranging from a valuable source of timber for the forest products industry to drinking and irrigated water supply, to fish and wildlife habitat, and human recreation. Forestland managers have advanced management practices over the last 30 or more years in an effort to minimize the effects of harvest activities on downstream aquatic ecosystems. In older unmanaged forests the increasing threat of wildfire due to long-term fire

suppression strategies and forest fuel accumulation (Agee and Skinner, 2005; Schoennagel et al., 2017), climate change (Schoennagel et al., 2017; Westerling et al., 2006), and insect/disease (Bentz et al., 2010; Collins et al., 2012) has elevated the need and consideration for more active forest management involving clear-cutting, thinning, and prescribed burns. In forested landscapes, any forest management strategy, including the ‘do-nothing’ option, will likely have long-term consequences on water quantity and quality (Johnson et al., 2011; Miller et al., 2005). It is becoming more widely recognized that wildfire is a

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part of natural forest ecosystem development and that the fire suppression strategy coupled with climate change has led to higher severity fires and increased post-fire environmental impacts (Schoennagel et al., 2017; Westerling et al., 2006). Recent shifts in the wildfire regime have led to increased research, illustrating the effects on the soil properties (Certini, 2005; Huffman et al., 2001; Martin and Moody, 2001; Robichaud, 2000) and the hydrologic response (Hallema et al., 2018; Niemeyer et al., 2020) that can result in potentially large sediment and nutrient loading (Emelko et al., 2016; Kunze and Stednick, 2006; Robinne et al., 2020; Rust et al., 2018) to the receiving waterbodies. To the nearly 180 million people and approximately 2/3 of municipalities across the US which rely on forested lands for drinking water (NRC, 2008), the impact of excessive nutrient and sediment loading following a wildfire is a great concern (Bladon et al., 2014; Jones et al., 2008; NRC, 2008). Many communities that rely solely on a single reservoir in forested landscapes that are highly vulnerable to wildfires are considering thinning operations to minimize the risk and severity of future wildfires despite the potential for short-term water quality impacts (Gannon et al., 2019). In recognition of the need to thin forests to reduce the severity and extent of wildfires and to maintain the full range of ecosystem services in commercial forestlands, there is a great need to identify and understand the short- and long-term effects of these forest management strategies on water quantity and quality.

Forested watersheds are often perceived as a sink rather than a source of nutrients, and although they continue to attract less attention in terms of nutrient management as compared to the watersheds dominated by agricultural and urban landscapes (Miller et al., 2005; Piatak and Allen, 2001), significant progress has been made in recent decades to understand nutrient effects in forested streams (Binkley et al., 2004; Argerich et al., 2013). While there are studies that have documented increases in nutrient concentrations from streams draining forested watersheds (Dubrovsky et al., 2010; Eddy-Miller et al., 2016), this is also contrasted with other studies that demonstrate decreasing trends in some forestland streams (Argerich et al. 2013). It is important to note that elevated nitrogen and phosphorus concentrations observed in forestland streams are generally orders of magnitudes lower than nutrient criteria set by the US Environmental Protection Agency (EPA), and any increases from contemporary forest management practices are very small compared to other land uses (Binkley, 2001; Ice and Binkley, 2003). Stream chemistry in forested landscapes is complex and is subject to considerable geographic and temporal variability (Feller, 2005). Natural variations in stream nutrient concentrations and loads can occur because of geology (Nagorski et al., 2003), wildfires (Miller et al., 2006, 2005; Murphy et al., 2006), precipitation dynamics, and consequent changes in discharge (Bhangu and Whitfield, 1997) and biological activity (Minshall et al., 2001; Peterson et al., 2001). Numerous studies have demonstrated that anthropogenic disturbances and forest management practices such as road construction, timber harvest, site preparation, and maintenance, increase nutrient concentrations (Anderson and Lockaby, 2011; Gravelle et al., 2009; Kreutzweiser et al., 2008) and loads (Dahlgren, 1998; Palviainen et al., 2015, 2014) in streams draining forested watersheds, but these increases are typically small in magnitude and limited in duration (Loehle et al., 2014).

This magnitude and duration of such changes in nutrient loads during and after forest management activities is dependent on several factors including topography, soil properties, watershed hydrology, the timing and type of management practices (e.g. broadcast burning of slash, competition release herbicide spraying), harvest intensity, vegetation recovery, climate, atmospheric deposition, mineralization rates, and relative distance of disturbance from streams (Kreutzweiser et al., 2008; Lintern et al., 2017). For instance, many studies have reported statistically significant changes in the stream water nutrient concentrations (Baldigo et al., 2005; Boggs et al., 2016; Martin et al., 2000; Tremblay et al., 2009) and loads (Boggs et al., 2016) following both partial and clear-cut harvesting treatments. Palviainen et al. (2014) reported minimal impact on stream nutrient loads from boreal forest

watersheds when clear cuts occurred on small scales (<10% of the total watershed area) and wider stream buffer zones (between 10 and 454 m) were maintained. But when the clear-cut exceeded 30% of the watershed area, annual loads increased 72% for total nitrogen, 76% for total organic nitrogen, 1056% for nitrate, 35% for phosphate, and 715% for suspended sediments, and this nutrient increase, in general, was sustained for more than 10 years. Siemion et al. (2011) reported that the initial effects and the long-term water quality response were directly related to harvest intensity, defined as the percent of total basal area removed, and the stream water nutrient concentrations declined faster in forests with lower intensity harvests. Regardless of harvest intensity, research compiled from recent decades has also demonstrated that contemporary forestry Best Management Practices (BMPs) appear to reduce water quality impacts by 80–90% compared to impacts from historic legacy practices (Loehle et al., 2014). While Swank et al. (2001) reported that the nitrate-nitrogen response peaked three years following the clear-cut harvest, Wang et al. (2006) have reported a five-fold increase in the nitrate-nitrogen loads from a catchment draining a hardwood forest one year after the forest was partial-cut. Other studies have also reported elevated nutrient concentrations and loads in streams draining clear-cut forests, especially during large storm events (Dahlgren, 1998; Oda et al., 2011). Many studies have found that atmospheric deposition and nutrient mineralization rates following management practices coupled with the high stream discharge associated with the storm events govern the variability in stream nutrient concentrations (Akselsson et al., 2004; Dahlgren, 1998; Kortelainen et al., 2006; Palviainen et al., 2014). All these studies reveal that the response in water quantity and quality to natural and anthropogenic disturbances in forested ecosystems can vary widely both in space and time. It is therefore imperative to further improve our understanding of the nutrient loads dynamics occurring in both relatively undisturbed as well as commercially managed forested watersheds.

Previous work in the Mica Creek Experimental Watershed (MCEW) focused on experimental treatment impacts of timber harvest on water yield (Hubbart et al., 2007), peak stream temperature (Gravelle and Link, 2007), suspended sediment loads (Karwan et al., 2007), and stream nutrient concentrations (Gravelle et al., 2009). In this study, we explore in greater depth the long-term response in nutrient loads following early vegetation regeneration from the actively managed timber harvest units in the MCEW. The primary objective of this study was to quantify and compare the differences in nitrogen and phosphorus concentrations and loads between the Pre-disturbance period, road construction (Post-road), experimental harvest (PH-I), and operational harvest (PH-II) with simultaneous regeneration. We analyze the extent to which the site management operations and harvesting practices occurring during two distinct time periods, Phase I (PH-I) and Phase II (PH-II), and how these activities affected nutrient yield from both a hydrological yield and nutrient concentration perspective.

2. Methods

2.1. Study area

This study was carried out in the Mica Creek Experimental Watershed (MCEW) located in Shoshone County in northern Idaho, US (Fig. 1). MCEW occupies roughly 2700 ha and is located approximately between latitudes 47° 11' 36" N and 47° 08' 13" N and longitudes 116° 18' 42" W and 116° 13' 5" W. It is a paired and nested watershed draining into the St. Joe River and is privately owned by PotlatchDeltic Corporation, a timberland real estate investment trust.

Elevation in the MCEW ranges from 1000 m and 1600 m a.m.s.l. The mountainous watershed has v-shaped valleys with slopes of 15–30% and stream gradients between 5 and 20%. The geology of the region is largely dominated by Wallace gneiss/Prichard quartzite parent material, which is overlain by silty soils. Annual mean precipitation is approximately 1450 mm, more than half of which falls primarily as snow from

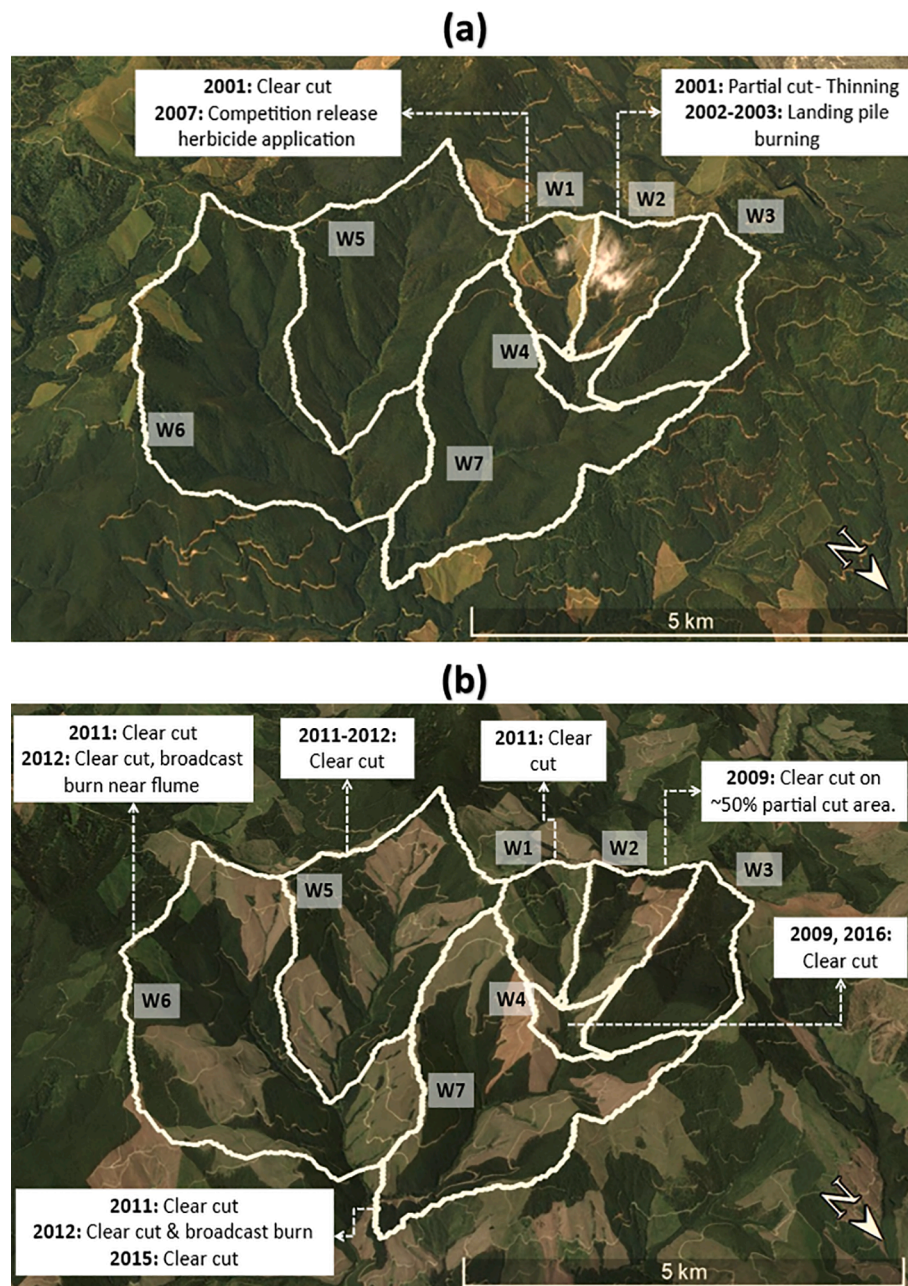


Fig. 1. (a) MCEW land cover (2005) before PH-II; (b) MCEW land cover (2016) post-PH-II harvest treatment.

October to March. The average annual temperature in the region is 5 °C with warm and dry summers and cold and wet winters. MCEW was harvested and burned between 1920 and 1930. The current vegetation cover in the watershed consists primarily of 90–100-year-old naturally regenerated second-growth mixed conifer and < 15-year-old replanted mixed conifer stands with majority comprised of Grand fir (*Abies grandis*), Douglas-fir (*Pseudotsuga menziesii*), Western red cedar (*Thuja plicata*) and Western larch (*Larix occidentalis*), and the under-story cover dominated by grasses, forbs, and shrubs.

2.1.1. Treatment sites

For this analysis time periods have been broken into four distinct phases: Pre-disturbance (1992–1997), Post-road (1997–2001), experimental-harvest Phase I (PH-I) (2001–2007), and operational sequential harvest Phase II (PH-II) when the extent and frequency of harvests increased (2007–2016). PH-I represents an experimental

treatment phase during which harvest activities were experimentally controlled (only upstream headwater watersheds were harvested and mature vegetation removal ranged between 24% and 47%) followed by site management operations including broadcast burning and replanting. PH-II represents the post-experimental phase where the study area transitioned to operational treatments that consisted of additional road construction and timber harvest, with site management operations including pile burning and competition release herbicide application. During this operational phase, the mature vegetation removal in the upstream and cumulative downstream watersheds ranged between 36% and 50% and 17–28%, respectively (Table 1). No activity occurred for the first 5 years of the study (Pre-disturbance phase) to provide baseline data. Roads for harvesting timber were constructed in the fall of 1997. This included restoration and slight widening of the existing primary road, which in the early 1930s was a railroad bed used for harvesting, to suit heavy truck traffic. Mid-slope access roads designed for log hauling

Table 1

Summary of MCEW characteristics and harvesting activities. CC refers to clear-cut, PC refers to partial-cut and C refers to control sites.

Watershed	Area (ha)	Harvest type	% Area harvested (PH-I)	%Area harvested (PH-II)	Mean degree slope	Elevation range (m)
W1	139	CC	47.1%	50.5%	19	1205–1528
W2	177	PC/CC	23.8%	35.8%	18	1201–1612
W3	227	C	0.0%	0.0%	17	1193–1612
W4	597	PC/CC	18.0%	25.7%	18	1169–1612
W5	667	C/CC	0.0%	17.5%	18	1055–1528
W6	1456	C/CC	0.0%	28.2%	18	1017–1594
W7	1226	CC/PC	9.0%	26.1%	18	1008–1612

were constructed in W1 and W2. Harvesting in PH-I occurred in the late summer and fall of 2001 and was completed with a combination of line and tractor skidding. This four-year period between road construction and harvesting provided for the isolation of road construction effects from harvest effects (post-road phase). About 23 ha of north-facing slopes and 43 ha of southeast-facing slopes in a 139 ha Watershed 1 (W1) were clear-cut harvested in 2001 (Fig. 1a). The lower slopes in this unit were broadcast burned in spring 2003 and replanted with mixed conifer species. The northeast and southeast slopes in a 176 ha Watershed 2 (W2) consisting of 34 and 49 ha, respectively were partial-cut harvested (i.e. 50% of the canopy was removed over 50% of the area) in 2001. Watersheds 4 (W4) and 7 (W7) were considered as the downstream cumulative effect assessment sites and Watersheds 5 (W5) and 6 (W6) were regarded as controls for these cumulative sites, respectively.

The PH-II harvesting treatment period began around the 2008 water year (Fig. 1b). New roads were initially constructed (with subsequent new construction throughout PH-II) for harvesting the downstream watersheds (W5–W7). In the summer of 2009, a portion of W2 that was partially cut as part of PH-I experimental treatment in 2001 was clear cut. Portions of the unharvested downstream control watersheds W5 and W6 were clear-cut harvested and broadcast burned in 2011 and 2012. A small portion of W1 adjacent to W4 was also clear-cut in 2011 and vegetation near the south side of the W1 flume was removed. Portions of watershed W7 were clear cut during 2011 (southwest of the flume), 2012 (near flume), and 2015 (northeast of the flume), respectively. Competition release herbicide, which generally consisted of some mixture of oust (sulfometuron methyl); Accord Concentrate (glyphosate); Atrazine 4L; or Arsenal AC (imazapyr), were applied to several post-harvest areas during the PH-II period. Although a herbicide treatment was first applied in W1 in August 2007 at the end of the PH-I period, this treatment was considered part of PH-II since the response from this activity occurred during PH-II. All forest management practices followed the Idaho Forest Practices Act and Stream Protection Zone (SPZ) requirements (Idaho Administrative Procedures Act (IDAPA), 2020; Idaho Department of Lands (IDL), 2000). This includes maintaining 75 ft (22.86 m) stream protection zones on each side of fish-bearing (Class I) streams. All existing shade in the Class I stream buffer was retained although the regulations before 2014 permitted the removal of 25% of the existing shade. Buffer requirements for non-fish-bearing (Class II) streams include a 30 ft (9.14 m) equipment exclusion zones on each side of the streams, but merchantable timber is allowed to be removed.

2.2. Water sampling and chemical analyses

Monthly manual grab samples of stream water were collected at each of the seven flumes in the MCEW using 250 ml high-density polyethylene (HDPE) bottles, starting in June 1992 until September 2016. The sample collection was continuous except in some winter months when harsh weather conditions limited access to the flumes. Although event-based storm sampling is preferable to fixed monthly grab sampling for capturing event loads, this was not feasible considering minimum holding times and the remote location of the site. The long

consistent monthly sampling record (i.e. over 25 years of data) in this data set however minimizes the risk of short term biases in the data. The collected samples were sent to water quality laboratories at Oregon State University in Corvallis, OR (1992–1997), the National Council for Air and Stream Improvement, Inc. (NCASI) in Anacortes, WA (1998–1999), and NCASI in Corvallis, OR (2000–2016). The samples were analyzed for six parameters, specifically nitrate + nitrite ($\text{NO}_3 + \text{NO}_2$), total Kjeldhal nitrogen (TKN), total ammonia nitrogen (TAN) containing un-ionized (NH_3) and ionized (NH_4^+) ammonia, total nitrogen (TN), total phosphorus (TP), and orthophosphate (OP). The TAN measurements started from August 1999 and the TP measurements were discontinued after July 2014. A detailed description of the sample collection and the chemical analyses (methods, instrument calibration procedures, and instrument detection limits) is included in Gravelle et al. (2009).

2.3. Load estimation

The monthly nutrient concentration data along with the corresponding daily discharge data were used to estimate the daily N and P loads for each of the seven flume sites using the Load Estimator (LOADEST) software (Runkel et al., 2004). LOADEST has nine pre-defined regression models to estimate nutrient loads given the measured nutrient concentrations and flows. In this study, the LOADEST model was allowed to automatically select the best regression model based on the lowest Akaike Information Criterion (AIC) value. The specific regression models selected by LOADEST to estimate N and P daily loads for each watershed as well as model performance metrics are presented in Table A.1 in the supporting information.

2.4. Study design and data analyses

A before-after, control-impact paired series (BACIPS) design (Stewart-Oaten et al., 2001, 1986) was used to evaluate both direct and cumulative effects of commercial forest management practices on stream nutrients in paired and nested watersheds that comprise the MCEW. In this study, the paired watershed approach was applied to the upper W1, W2, and W3 watersheds each with areas of approximately 139 ha, 177 ha, and 227 ha, respectively (Fig. 1). The cumulative effects of management in the upper watersheds during Phase I were assessed using the downstream watersheds W4 and W7, and their paired control watersheds W5 and W6 that occupy 597 ha and 667 ha, and 1456 ha and 1226 ha, respectively.

The nutrient data sample size (n) of 54, 34, 50, and 120 instantaneous measurements were available during the Pre-disturbance, Post-road, PH-I, and PH-II treatment periods, respectively. The nutrient data from the lab analyses were used to establish a linear relationship in stream nutrient concentrations between control and treatment watersheds during the Pre-disturbance phase. Similarly, BACIPS relationships between the control and treatment watersheds during the Pre-disturbance phase were also established for daily streamflow observations as well as nutrient loads. Since portions of the downstream control watersheds (W5 and W6) were harvested during PH-II, all the downstream sites during this treatment period were compared to the

remaining control site (W3). To do so, linear relationships between monthly nutrient concentrations, daily streamflow, and daily nutrient load observations at W3 and the downstream control sites (W5 & W6) and the downstream cumulative sites (W4 & W7) were established. The intercept ($X_{control}$) and slope (a) from these relationships were then used to predict the nutrient concentrations, streamflow, and loads that would have occurred in the treatment watersheds ($C_{treatment, predicted}$) in the absence of management practices (Equation (1)). $C_{control, observed}$ in Equation (1) is the observed nutrient concentration/streamflow/load at the control watershed. Since this method provides control over climate variability (Bari et al., 1996) and can separate the climatic effects from vegetation cover effects (Hewlett et al., 1969), the estimated difference (ΔC) between the observed ($C_{treatment, observed}$) and predicted ($C_{treatment, predicted}$) concentrations (Equation (2)) are assumed to have occurred because of changes in vegetation cover and/or soil disturbance. Such research design using BACIPS approaches has been implemented in several recent studies (Artigas et al., 2013; Boggs et al., 2016; Menberu et al., 2017; Swank et al., 2001) to investigate the relationship between harvesting and water quality across a diverse range of ecosystems.

$$C_{treatment, predicted} = X_{control} + a(C_{control, observed}) \quad (1)$$

$$\Delta C = C_{treatment, observed} - C_{treatment, predicted} \quad (2)$$

The difference between the observed and predicted nutrient concentrations, streamflow, and nutrient loads were tested for statistical significance using the Student's *t*-test at a 0.05 significance level (α). The statistical software R was used for all statistical computations and analyses. All the *p*-values are based on a comparison of Pre-disturbance (1992–1997) to Post-road (1997–2001), to PH-I (2001–2007), and PH-II (2007–2016) treatment periods except for the TAN. Since TAN ($NH_3 + NH_4$) data collection started in 1999, Post-road concentrations were compared to post-harvest and PH-II concentrations to determine if significant changes in TAN concentrations occurred following timber harvest.

One challenge with analyzing nutrient data from forested watersheds is that a large portion of the data can vary around the minimum detection limit. A small fraction of the total nitrogen and phosphorus observations were below-detection limits, ranging from a maximum of 16% to no observations below detection limits within a particular phase. The fraction of samples below these limits decreased in the later treatment phases. In this study, values smaller than the minimum detection limit were assumed to be half the value of the detection limit. Such a substitution of the values below detection limits is a common approach to handle concentrations below detection limits (Huston and Juarez-Colunga, 2009).

The Modified Mann-Kendall trend test (Hamed and Rao, 1998) was used to identify the monotonic trend in the daily as well as seasonal streamflow. The modified Mann-Kendall test corrects for the autocorrelation effects on the Mann-Kendall test statistic and therefore is suitable for detecting statistically significant trends in autocorrelated data (Hamed and Rao, 1998). The Kendall-Tau statistic (ranges between -1 and 1) provides the strength of ranked correlation between the ordered daily/seasonal streamflow at a certain significance level (α).

Both the length of the water quality record and the multiple paired and nested sub-watershed design provided a very rare opportunity to examine the impacts of two distinct intensities of commercial timber harvest in forested watershed.

3. Results

3.1. Streamflow

3.1.1. Long-term trends in streamflow

Table 2 details seasonal trends in the daily mean streamflow across all management phases at the MCEW watersheds. Statistically significant increasing trends in streamflow were identified in all the watersheds during the fall and winter seasons. The relative magnitude of this increasing trend was highest during the fall with Kendall- τ values ranging between 0.30 and 0.48 followed by the magnitudes during winter which ranged between 0.22 and 0.32. Summer flows increases were significant only at W5 and W6. Spring and summer streamflow showed no significant change at W3.

3.1.2. Effects of site treatment on streamflow

Trends in mean annual streamflow in all the MCEW watersheds, including the control (W3) watershed, are displayed in Fig. 2. Table 3 summarizes the observed streamflow and the estimated change in streamflow for each treatment phase along with the Student's *t*-test *p*-values between the treatment–control watershed pairs at MCEW. During the Post-road phase, a statistically significant but relatively small increase in streamflow occurred in the paired watersheds W1 (58 mm yr^{-1} , *p*-value = 0.039) and W2 (66 mm yr^{-1} , *p*-value = 0.012). In all post-harvest watersheds (PH-I or PH-II treated), statistically significant increases in stream flows were observed. For the PH-I period, the largest increase in streamflow was observed in the W1 (248 mm yr^{-1} , *p*-value < 0.001) watershed while the increase at the downstream cumulative sites W4 (*p*-value < 0.001) and W7 (*p*-value = 0.002) amounted to 131 mm yr^{-1} and 51 mm yr^{-1} , respectively. During the PH-II treatment, the highest increase in streamflow occurred at the W2 (252 mm yr^{-1} , *p*-value < 0.001) followed by W5 (248 mm yr^{-1} , *p*-value < 0.001) watershed. The increase in streamflow at the downstream cumulative sites (*p*-value < 0.001) W4 and W7 during this treatment phase amounted to 186 mm yr^{-1} and 91 mm yr^{-1} , respectively.

3.2. Nitrogen and phosphorus concentrations and loads

3.2.1. Total kjeldahl nitrogen (TKN)

There were minimal changes in TKN concentration (Figure A.1), with a slight observed reduction in long-term TKN loads. Based on Student's *t*-test results (Table 5), significant changes in TKN concentrations only occurred in one of the two paired headwater catchments (W2) during PH-I and PH-II treatment periods, and at the W5, W6, and W7 watersheds during the PH-II treatment period only. The TKN concentration at W2 decreased by 0.018 mg-N L^{-1} and 0.028 mg-N L^{-1} during PH-I (*p*-value = 0.017) and PH-II (*p*-value = 0.011) treatment periods, respectively. Despite the reduced concentrations, the TKN load from W2

Table 2

Kendall rank correlation coefficient statistics for the daily mean streamflow series for each season (months' abbreviations in parentheses). All Kendall- τ values at a *p*-value < 0.05 indicate a statistically significant trend.

	Winter (DJF)		Spring (MAM)		Summer (JJA)		Fall (SON)	
	Kendall- τ	<i>p</i> -value	Kendall- τ	<i>p</i> -value	Kendall- τ	<i>p</i> -value	Kendall- τ	<i>p</i> -value
W1	0.265	<0.001	0.125	<0.001	0.070	0.235	0.319	<0.001
W2	0.318	<0.001	0.179	<0.001	0.137	0.050	0.409	<0.001
W3	0.253	<0.001	0.049	0.211	0.106	0.069	0.427	<0.001
W4	0.309	<0.001	0.133	<0.001	0.140	0.054	0.483	<0.001
W5	0.276	0.005	0.167	0.015	0.181	<0.001	0.402	0.001
W6	0.263	0.006	0.155	0.022	0.112	0.049	0.300	<0.001
W7	0.217	0.001	0.120	0.005	0.066	0.246	0.300	<0.001

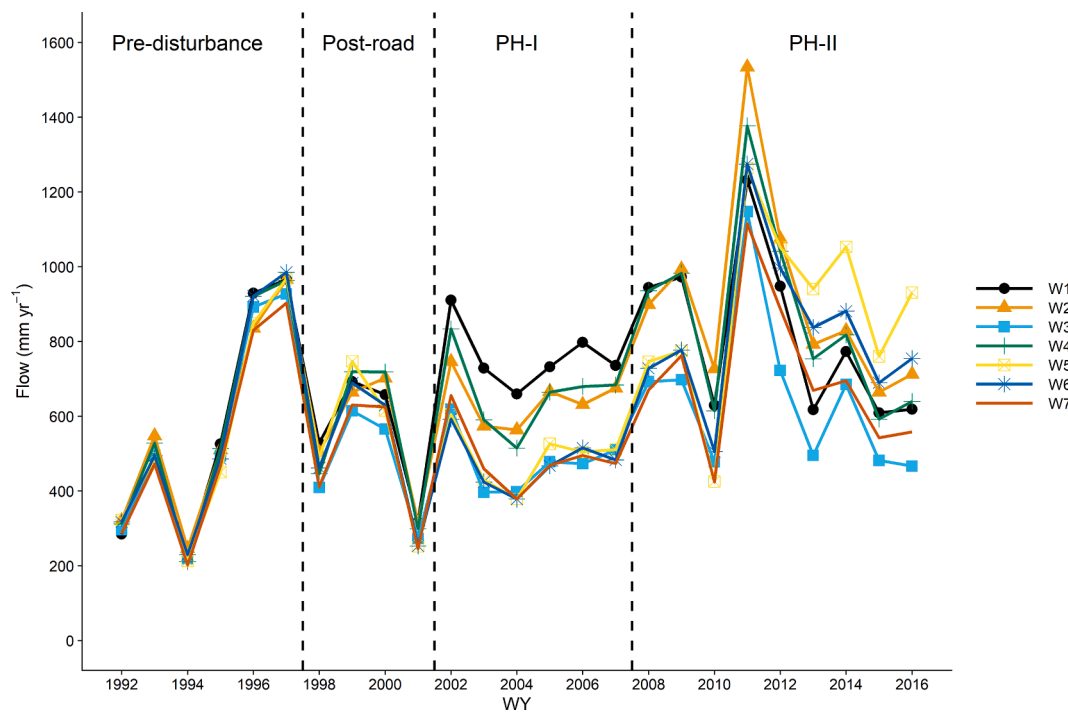


Fig. 2. Mean annual streamflow from all watersheds in the MCEW.

Table 3

Observed streamflow, estimated change in streamflow due to treatment calculated from relationships with paired control watershed response (equation (2)), and Student's *t*-test *p*-value results. Estimated change at $\alpha < 0.05$ indicates a statistically significant change. Insignificant changes are indicated by an asterisk next to the watershed name.

Treatment	Watershed	Mean Observed (mm yr ⁻¹)	Estimated Change (mm yr ⁻¹)	<i>p</i> -value
Post-road	W1	558.45	58.40	0.039
	W2	558.45	65.70	0.012
	W4*	551.15	-21.90	0.472
	W7*	489.10	10.95	0.695
PH-I	W1	740.95	248.20	<
	W2	620.50	131.40	<
	W4	646.05	131.40	<
	W7	474.50	51.10	0.002
PH-II	W1	817.60	146.00	<
	W2	908.85	251.85	<
	W4	865.05	186.15	<
	W7	704.45	91.25	<
	W5	886.95	248.20	<
	W6	828.55	167.90	<

increased by 0.14 kg ha⁻¹ yr⁻¹ and 0.09 kg ha⁻¹ yr⁻¹ during PH-I (*p*-value < 0.001) and PH-II (*p* = 0.006) treatment periods, respectively (Table 6) due to increasing streamflow. TKN concentration at W7 during the PH-II treatment period decreased by 0.020 mg-N L⁻¹ (*p*-value = 0.034) and TKN load during this period was 0.37 kg ha⁻¹ yr⁻¹. The cumulative annual load of TKN from MCEW watersheds is depicted in Figure A.2. Change in the slope of the annual load curve in the cumulative plot indicates the change (increase/decrease) in load whereas the

Table 4

Mean annual load of nitrogen and phosphorus from each watershed per treatment period in kg ha⁻¹ yr⁻¹. NA: data not measured/discontinued.

	W1	W2	W3	W4	W5	W6	W7
TKN	Pre-disturbance	0.67	0.82	0.65	0.81	0.80	0.88
	Post-road	0.86	0.73	0.59	0.75	1.05	0.85
	PH-I	0.32	0.23	0.10	0.18	0.13	0.11
	PH-II	NA	0.44	0.29	0.52	0.49	0.42
TAN	Pre-disturbance	NA	NA	NA	NA	NA	NA
	Post-road	0.07	0.11	0.05	0.26	0.07	0.09
	PH-I	0.08	0.07	0.05	0.08	0.05	0.05
	PH-II	0.09	0.11	0.07	0.10	0.12	0.10
NO ₃ + NO ₂	Pre-disturbance	0.40	0.07	0.07	0.13	0.05	0.08
	Post-road	0.57	0.42	0.16	0.32	0.14	0.24
	PH-I	2.99	0.35	0.10	0.69	0.07	0.06
	PH-II	3.95	1.35	0.33	1.73	0.66	0.77
TP	Pre-disturbance	0.15	0.16	0.12	0.16	0.16	0.18
	Post-road	0.25	0.21	0.15	0.22	0.26	0.23
	PH-I	0.15	0.12	0.11	0.12	0.12	0.09
	PH-II	0.16	0.19	0.08	0.15	0.19	0.16
OP	Pre-disturbance	0.07	0.06	0.04	0.05	0.05	0.05
	Post-road	0.08	0.08	0.05	0.07	0.08	0.08
	PH-I	0.12	0.10	0.06	0.10	0.08	0.07
	PH-II	0.11	0.12	0.06	0.11	0.12	0.10

flattening of this slope indicates recovery of load to the background values. Although a statistically significant decline in concentrations occurred only during the PH-I and PH-II treatment periods, the overall steady TKN load from all watersheds is apparent.

3.2.2. Total ammonia nitrogen (TAN)

Analysis of TAN concentrations (Figure A.3, Table 5) showed a statistically significant decrease at W1 (0.002 mg-NH₃ L⁻¹, *p* = 0.044) and W2 (0.003 mg-NH₃ L⁻¹, *p*-value = 0.042) watersheds during the PH-I

Table 5
Observed nutrient concentrations, estimated change in concentration due to treatment calculated from relationships with paired control watershed response (Equation (2)), and Student's *t*-test *p*-value-value results. The estimated change at *p*-value < 0.05 indicates a statistically significant change.

		Post-Road				Post-Harvest				Phase-II					
		W1	W2	W4	W7	W1	W2	W4	W7	W1	W2	W4	W5	W6	W7
TKN	Mean Observed (mg-N L ⁻¹)	0.166	0.145	0.137	0.178	0.048	0.041	0.037	0.041	NA	0.046	0.045	0.045	0.041	0.048
	Estimated Change (mg-N L ⁻¹)	0.037	0.002	-0.028	0.039	-0.012	-0.018	0.012	-0.012	NA	-0.028	0.008	-0.025	-0.019	-0.020
	<i>p</i> -value	0.218	0.942	0.419	0.287	0.183	0.017	0.176	0.168	NA	0.011	0.454	0.001	0.019	0.034
TAN	Mean Observed (mg-NH ₃ L ⁻¹)	NA	NA	NA	NA	0.011	0.012	0.015	0.013	0.011	0.013	0.011	0.013	0.012	0.013
	Estimated Change (mg-NH ₃ L ⁻¹)	NA	NA	NA	NA	-0.002	-0.003	0.002	-0.001	-0.002	-0.001	-0.001	0.000	-0.008	-0.004
	<i>p</i> -value	NA	NA	NA	NA	0.044	0.042	0.443	0.627	0.015	0.470	0.287	0.580	<0.001	<0.001
NO ₃ + NO ₂	Mean Observed (mg-N L ⁻¹)	0.060	0.039	0.050	0.037	0.338	0.053	0.101	0.049	0.401	0.157	0.192	0.077	0.086	0.144
	Estimated Change (mg-N L ⁻¹)	-0.003	0.011	0.003	0.025	0.276	0.034	0.072	0.038	0.334	0.113	0.128	0.046	0.022	0.117
	<i>p</i> -value	0.793	0.414	0.893	0.082	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	0.027	<0.001
TP	Mean Observed (mg-P L ⁻¹)	0.048	0.040	0.040	0.048	0.020	0.019	0.023	0.022	0.018	0.018	0.016	0.020	0.019	0.017
	Estimated Change (mg-P L ⁻¹)	0.003	-0.009	-0.005	0.009	-0.006	-0.008	-0.003	0.000	0.002	0.003	0.001	0.006	0.004	0.003
	<i>p</i> -value	0.760	0.299	0.560	0.207	0.204	0.118	0.587	0.925	0.146	0.045	0.616	<0.001	0.030	0.048
OP	Mean Observed (mg-P L ⁻¹)	0.014	0.014	0.011	0.015	0.017	0.016	0.016	0.017	0.014	0.013	0.013	0.014	0.013	0.013
	Estimated Change (mg-P L ⁻¹)	0.002	0.003	0.000	0.006	0.003	0.003	0.003	0.007	0.003	0.003	0.002	0.004	-0.003	0.003
	<i>p</i> -value	0.152	0.109	0.821	0.021	0.144	0.136	0.120	<0.001	0.002	0.002	0.009	<0.001	0.068	0.018

Table 6
Nutrient load ($\text{kg ha}^{-1} \text{yr}^{-1}$), the estimated change in load due to treatment calculated from relationships with paired control watershed response (Equation (2)), and Student's *t*-test *p*-value-value results. The estimated change at *p*-value < 0.05 indicates a statistically significant change.

		Post-Road				PH-I		PH-II		W4		W5		W6		W7	
		W1	W2	W4	W7	W1	W2	W1	W2	W4	W7	W4	W5	W6	W7	W4	W7
TKN	Mean Observed	0.859	0.732	0.750	0.365	0.315	0.231	0.136	0.438	0.524	0.136	0.177	0.494	0.424	0.366	0.524	0.366
	Estimated Change	0.243	-0.016	-0.290	-0.141	0.217	0.139	0.092	0.088	0.279	0.092	-0.026	0.184	0.079	0.090	0.279	0.090
TAN	<i>p</i> -value	<0.001	0.665	<0.001	0.004	<0.001	<0.001	<0.001	0.006	<0.001	<0.001	<0.001	<0.001	0.062	0.010	<0.001	0.010
	Mean Observed	NA	NA	NA	NA	0.075	0.068	0.053	0.110	0.096	0.053	0.080	0.117	0.102	0.098	0.096	0.098
NO ₃ + NO ₂	Estimated Change	NA	NA	NA	NA	0.004	-0.046	0.010	-0.032	-0.134	0.010	-0.134	0.029	0.002	0.015	-0.213	0.015
	<i>p</i> -value	NA	NA	NA	NA	0.193	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	0.633	<0.001	<0.001	<0.001
TP	Mean Observed	0.567	0.422	0.321	0.156	2.986	0.352	0.243	1.346	0.685	0.243	0.685	0.664	0.771	1.240	1.727	1.240
	Estimated Change	-0.302	0.272	-0.059	-0.029	2.396	0.251	0.196	1.040	0.496	0.196	0.496	0.471	0.481	0.882	0.931	0.882
OP	<i>p</i> -value	<0.001	<0.001	0.012	0.094	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
	Mean Observed	0.246	0.213	0.223	0.109	0.150	0.122	0.107	0.185	0.123	0.107	0.123	0.194	0.159	0.128	0.145	0.128
OP	Estimated Change	0.068	0.031	-0.041	-0.020	0.015	-0.016	0.033	0.057	-0.005	0.033	-0.005	0.088	0.032	0.027	0.041	0.027
	<i>p</i> -value	<0.001	<0.001	<0.001	<0.001	0.008	0.002	<0.001	<0.001	0.152	<0.001	0.152	<0.001	<0.001	<0.001	<0.001	<0.001
OP	Mean Observed	0.075	0.085	0.065	0.032	0.117	0.098	0.073	0.111	0.102	0.073	0.102	0.122	0.103	0.088	0.106	0.088
	Estimated Change	-0.033	-0.003	-0.011	-0.006	-0.007	-0.002	-0.002	-0.006	0.027	-0.002	0.027	0.031	0.015	-0.001	0.018	-0.001
OP	<i>p</i> -value	<0.001	0.607	0.003	0.665	0.173	0.654	0.454	<0.001	<0.001	0.454	<0.001	<0.001	<0.001	0.769	<0.001	0.769

treatment period. During PH-II, a similar small decline in the TAN concentration was observed at W1, W6, and W7, which amounted to $0.002 \text{ mg-NH}_3 \text{ L}^{-1}$ (*p*-value = 0.015), $0.008 \text{ mg-NH}_3 \text{ L}^{-1}$ (*p*-value < 0.001), and $0.004 \text{ mg-NH}_3 \text{ L}^{-1}$ (*p*-value < 0.001), respectively. For the same treatment period, the TAN load (Table 6) appeared to be relatively stable at the W1 and W6 watersheds. The mean annual TAN load from the W7 watershed during PH-II showed an increase of $0.05 \text{ kg ha}^{-1} \text{yr}^{-1}$ compared to the PH-I period (Table 4). Overall, the cumulative mean TAN loads from all watersheds did not show large variations with sequential varying treatments over time (Figure A.4).

3.2.3. Nitrate + Nitrite ($\text{NO}_3 + \text{NO}_2$)

In contrast to TAN, there was a significant response in $\text{NO}_3 + \text{NO}_2$ following timber harvest activities in the MCEW. The response in $\text{NO}_3 + \text{NO}_2$ concentrations was negligible at all treatment sites, with W7 showing negligible effects (*p*-value = 0.08), following the road construction activities, however $\text{NO}_3 + \text{NO}_2$ concentrations during the PH-I period increased significantly (*p*-value < 0.001) at all treatment sites (Table 5). The largest increase in the mean monthly concentration during the PH-I period was a 0.28 mg-N L^{-1} increase observed following clear-cut operations at W1. Similar to the PH-I period, all watersheds experienced significant increases in $\text{NO}_3 + \text{NO}_2$ concentration during the PH-II treatment period. The greatest response in $\text{NO}_3 + \text{NO}_2$ during the PH-II treatment period was a 0.33 mg-N L^{-1} increase at W1, which is possibly related to the competition release herbicide applied within an original experimental harvest unit. As seen in Fig. 3, $\text{NO}_3 + \text{NO}_2$ concentrations increased at W1 during both phases which were then followed by roughly a 5-year recession back to approximately twice the pre-harvest concentrations. Mean annual $\text{NO}_3 + \text{NO}_2$ loads from all watersheds increased considerably, especially, during the PH-I and PH-II treatment periods. Statistically significant increases in $\text{NO}_3 + \text{NO}_2$ loads occurred at all the sites (*p*-value < 0.001) during all treatment phases (Table 6) with the largest increase of $2.19 \text{ kg ha}^{-1} \text{yr}^{-1}$ and $1.04 \text{ kg ha}^{-1} \text{yr}^{-1}$ occurring at the two headwater sites W1 and W2, respectively. The highest mean annual load of $\text{NO}_3 + \text{NO}_2$ occurred at W1 amounting to $3.95 \text{ kg ha}^{-1} \text{yr}^{-1}$ during the PH-II treatment period followed by the load of $1.73 \text{ kg ha}^{-1} \text{yr}^{-1}$ at W4 during the same treatment period (Table 4). Overall, the cumulative mean $\text{NO}_3 + \text{NO}_2$ load from all the MCEW watersheds followed an increasing trend with initial signs of recovery in W1 after 2014 as indicated by a flattening of the cumulative load curve near the end of the data series (Fig. 4).

3.2.4. Total phosphorus (TP)

Mean monthly TP concentrations (Fig. 5) showed no significant changes in the concentrations during the post-road and PH-I treatment periods. The mean monthly concentration at all treatment sites during the post-road phase was $0.044 \text{ mg-P L}^{-1}$, which decreased to $0.021 \text{ mg-P L}^{-1}$ during the PH-I phase. However, a statistically significant increase in TP concentrations ($0.003 \text{ mg-P L}^{-1}$, *p*-value = 0.048) and load ($0.03 \text{ kg ha}^{-1} \text{yr}^{-1}$, *p*-value < 0.001) occurred at the downstream cumulative site W7 during PH-II. TP loads increased significantly (*p*-value < 0.001) at all sites during PH-II (Table 6). This is also apparent from the cumulative annual mean TP load displayed in Fig. 6.

3.2.5. Orthophosphate (OP)

Generally, OP concentrations throughout the study remained near the minimum detectable concentrations. A statistically significant increase in mean monthly OP concentrations of $0.006 \text{ mg-P L}^{-1}$ and $0.007 \text{ mg-P L}^{-1}$ occurred only at the W7 (cumulative downstream) treatment site during both Post-road (*p*-value = 0.021) and PH-I (*p*-value < 0.001) treatment periods, respectively. The mean concentrations at all sites were relatively constant at $0.015 \text{ mg-P L}^{-1}$ during both PH-I and PH-II periods. During the PH-II treatment period, however, small but statistically significant, changes in the mean monthly orthophosphate concentrations occurred at all watersheds, except W6 (Table 5). A maximum increase of $0.004 \text{ mg-P L}^{-1}$ was observed at the W5 (*p*-value < 0.001)

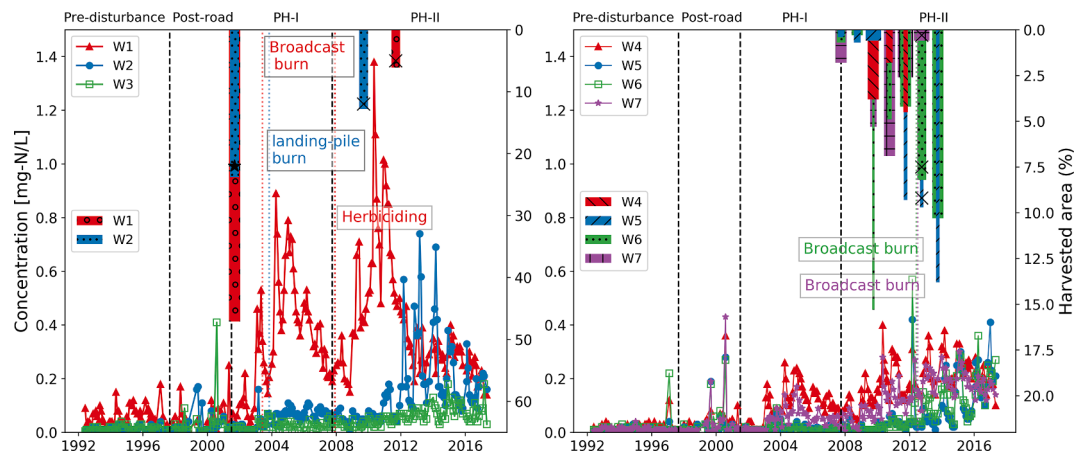


Fig. 3. Observed monthly stream $\text{NO}_3 + \text{NO}_2$ concentrations (mg-N L^{-1}) in MCEW watersheds. The asterisk denotes partial cut and “x” denotes harvest that includes areas adjacent to the stream.

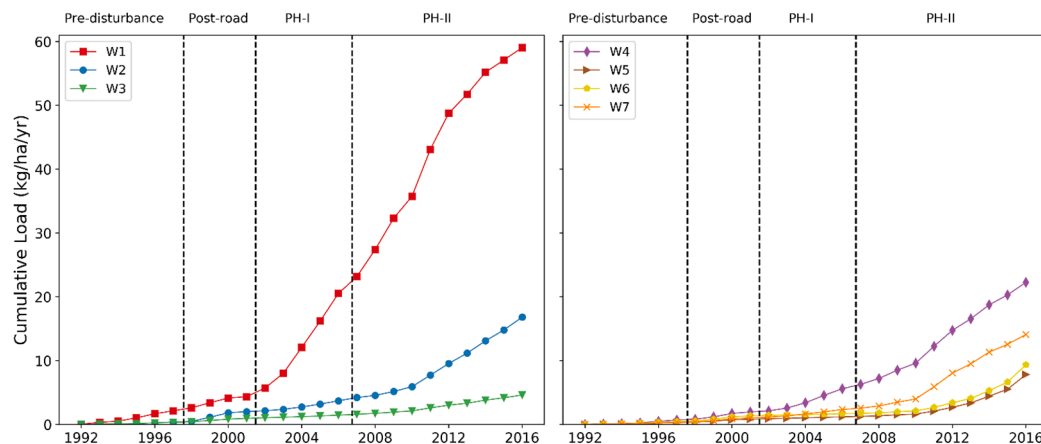


Fig. 4. Cumulative annual mean $\text{NO}_3 + \text{NO}_2$ loads from MCEW watersheds.

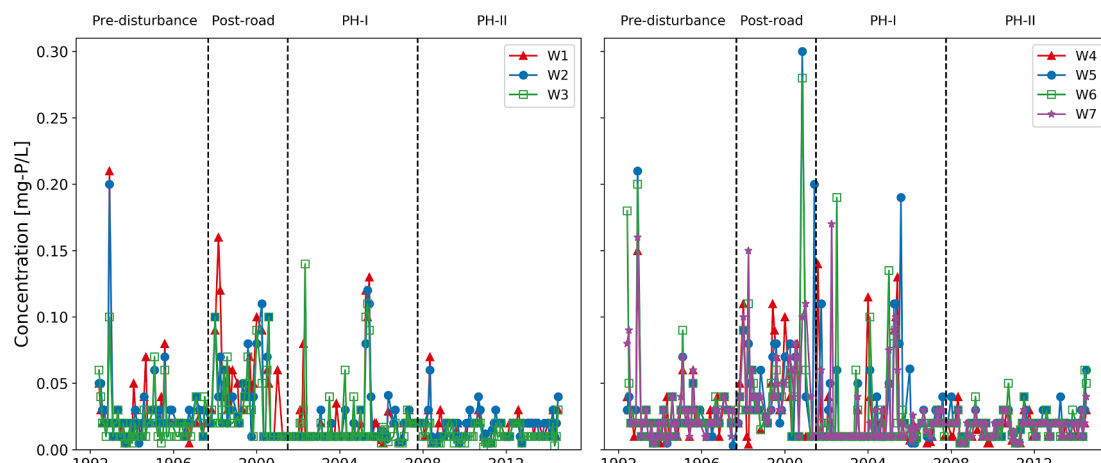


Fig. 5. Observed monthly stream TP concentrations (mg-P L^{-1}) in MCEW watersheds.

treatment site. Overall, trends in OP concentrations and cumulative annual mean load are shown in Fig. 7 and Fig. 8, respectively. Despite relatively small changes in OP concentrations following harvest treatments, the increased streamflow during this time period resulted in increases in OP loads from all of the watersheds, except for W1 and W7, during the PH-II treatment period (Table 6). Maximum load of OP

occurred at W2 and W5 amounting to $0.12 \text{ kg ha}^{-1} \text{ yr}^{-1}$ followed by W4 with $0.11 \text{ kg ha}^{-1} \text{ yr}^{-1}$ (Table 4). The largest cumulative mean annual loads occurred from the W1 and W5 watersheds (Fig. 8) and can be largely attributed to the increased flow.

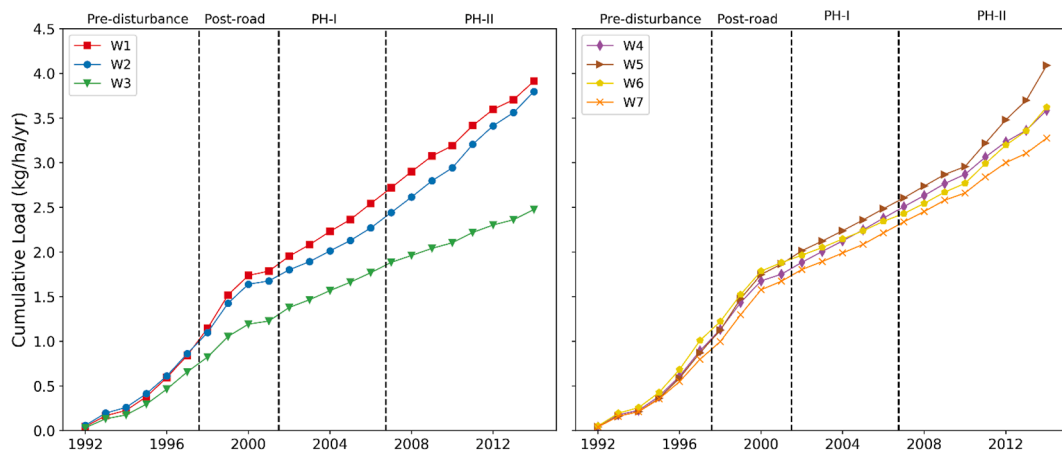


Fig. 6. Cumulative annual mean TP loads from MCEW watersheds.

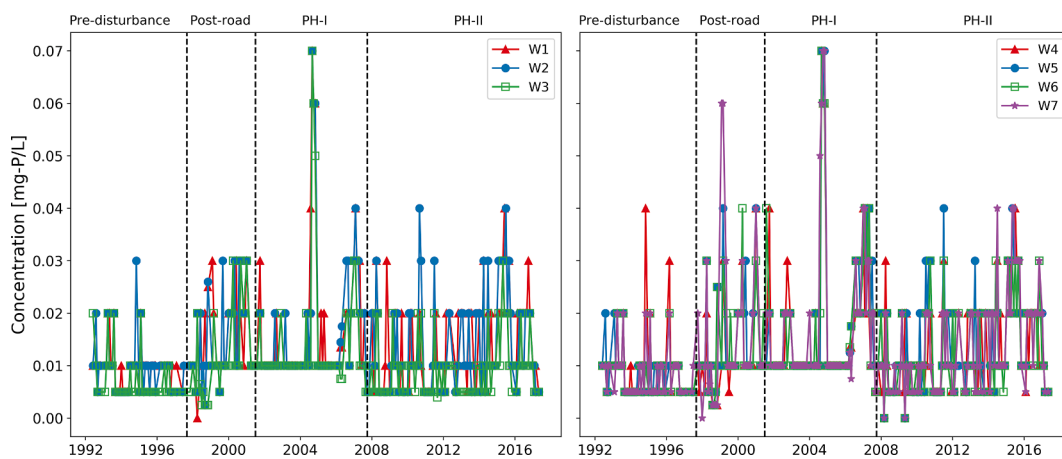


Fig. 7. Observed monthly stream OP concentrations (mg-P L^{-1}) in MCEW.

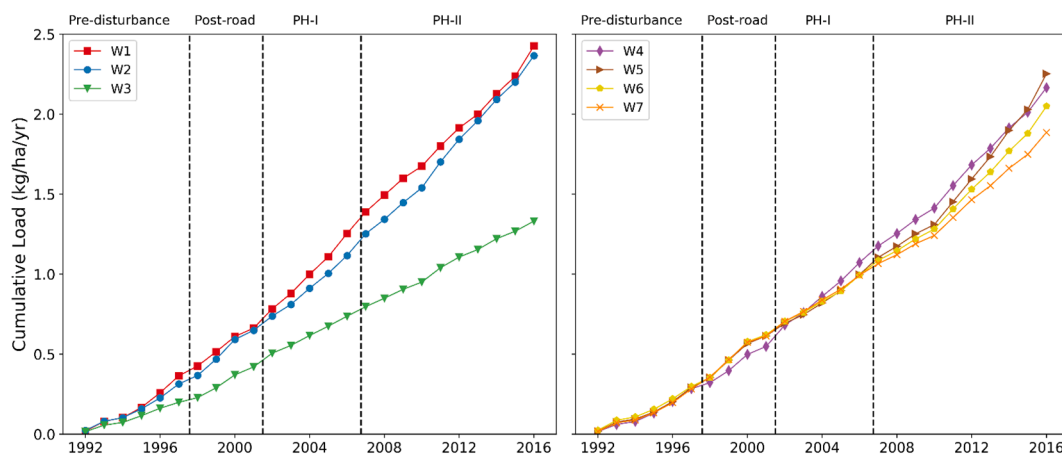


Fig. 8. Cumulative annual mean OP loads from MCEW watersheds.

4. Discussion

The long-term monitoring and assessment of the temporal responses in nutrient concentrations and streamflows from paired and nested watersheds in the MCEW provide important insights into the effects of forest management practices on nutrient loads relative to the response from an undisturbed forest. Through the synthesis of two-and-a-half decade-long nutrient concentration data in MCEW, we found the

greatest response to timber harvest is an increase in $\text{NO}_3 + \text{NO}_2$ concentrations following clear-cutting and site management operations such as broadcast burning and herbicide application. Although OP concentrations remained near minimum detectable levels, both $\text{NO}_3 + \text{NO}_2$ and OP loads increased significantly at all locations as a result of the increased streamflows across the MCEW. TP loads observed in this study showed little to no response to the clear-cut harvest and broadcast burning in the MCEW and these findings are consistent with similar

findings related to suspended sediment loading (Karwan et al.; 2007). Karwan et al., (2007) reported an increase in sediment loads following the clear-cut harvest for a brief period (~1 year) in MCEW which returned to background levels shortly after harvest.

4.1. Streamflow

Road construction resulted in a relatively small increase in streamflow in W1 (10.6%) and W2 (11.4%) watersheds as compared to the increase of 33.4% and 21.5% following the PH-I clear cut and partial cut treatments in the respective watersheds. The streamflow in the larger cumulative watersheds W4 and W7 increased by 20.4% and 10.5% during the PH-I phase and by 21.5% and 12.9% during the PH-II phase, respectively. These results are slightly different from the findings of Hubbart et al. (2007) who analyzed data for slightly different treatment periods than the current study and reported similar increases in water yield from W1 and W2 following harvesting. To further identify and separate the impacts of management from annual precipitation on streamflow response, a streamflow index was calculated as the ratio of the annual streamflow at a watershed outlet and the total annual precipitation observed at the Mica Creek SNOTEL station (SNOTEL 623, elevation- 1375 m) (annual Q:P_Mica) (Fig. 9). No attempt was made to

correct the average annual precipitation by elevation or topography and therefore this index should not be used as an absolute predictor of runoff; however, assuming the relative difference in precipitation across the basin does not differ across years, the efficiency of precipitation capture by the large-volume storage gauge at the SNOTEL site and the accuracy of flow measurements did not change, this index can serve as a reasonable proxy for variability in streamflow response through time. As Fig. 9 shows, as the percent mature forest cover declines due to timber harvest this streamflow index increases. Interestingly, there is also a small temporal increase in the streamflow index in the control watershed (W3) in MCEW.

To assess whether this increasing trend in streamflow from the control watershed was consistent across the region we also investigated the streamflow from a nearby watershed draining into East Fork Pine Creek (PC) gauge station (USGS 12413370) for the same treatment period. PC is a 72 km² watershed located 25 km north of MCEW and is similar in terms of elevation, size, and slope-aspect. The PC streamflow record showed a similar increase in long-term daily streamflow and the streamflow index calculated using MCEW SNOTEL, as observed in W3 in MCEW (see supplemental Figures A.5-A.6). Long-term cumulative outflow plots at each of these stream gauges show an increasing trend in water yield, particularly around 2010, at W3 and W7 in MCEW and PC

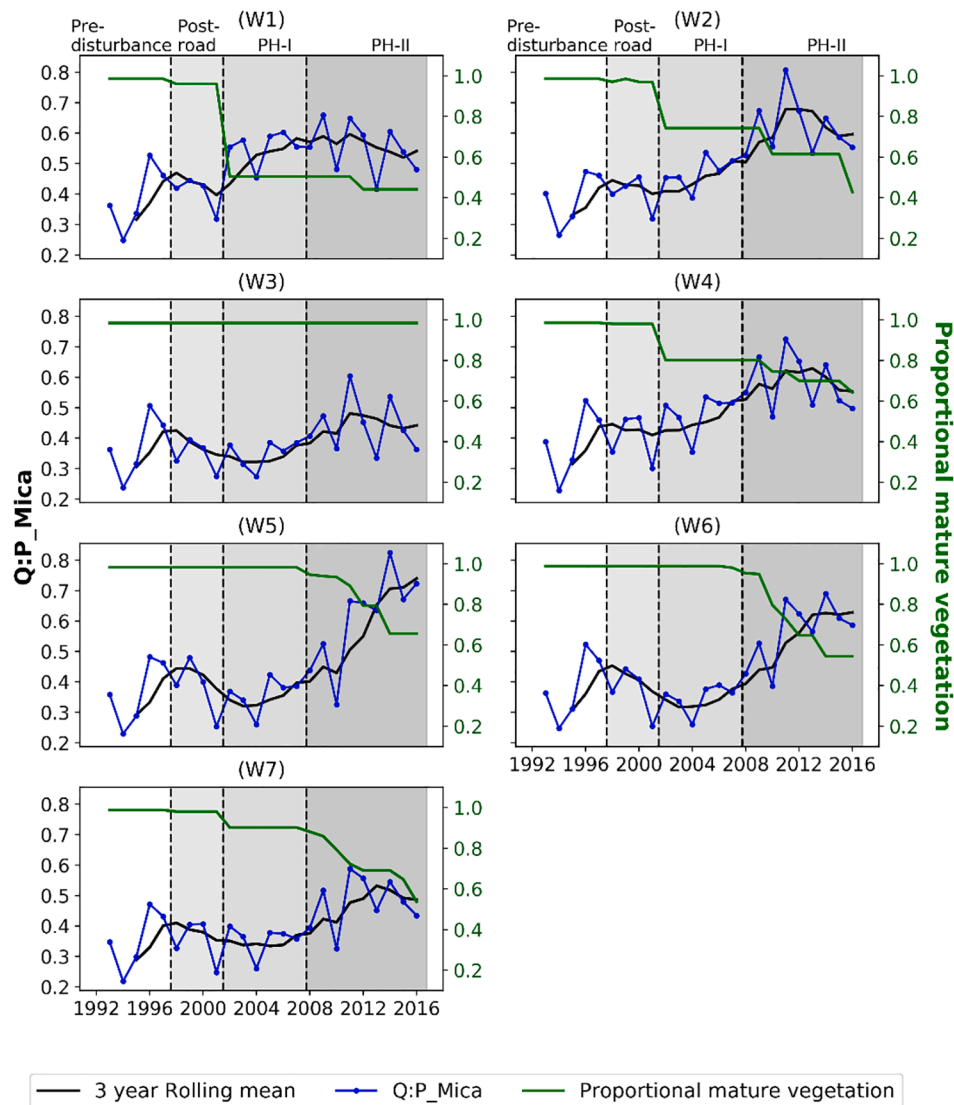


Fig. 9. Annual streamflow index (ratio of observed annual streamflow to the annual precipitation measured at Mica SNOTEL (623) Q:P_Mica) and percent area covered with mature vegetation in MCEW.

(see supplemental Figure A.7).

These observations demonstrate the importance of the BACIPS design to account for variations like streamflow through time. Several factors could help explain the relative increase in streamflow observed at W3, and these include: catchments like MCEW and PC are located in the rain-snow transition zone where the hydrologic response is more sensitive to changes in climate; variability caused by El Niño Southern Oscillation (ENSO)/ Pacific Decadal Oscillation (PDO) patterns; changes in the transpiration dynamics of the aging forest (Koch et al., 2004; Naranjo et al., 2012); observed tree mortality; or minor wind redistribution of snow (Hubbart et al., 2015) from thinned/clear-cut areas adjacent to the mature forest stands. Further data collection and focused research will provide more clarity to the causes and persistence of this observed trend of increasing streamflow.

4.2. Spatio-temporal dynamics of nitrogen and phosphorus

4.2.1. Nitrogen

Similar to the streamflow response, forest management activities in MCEW affected the transport of nitrogen from the landscape. During the Pre-disturbance period, $\text{NO}_3 + \text{NO}_2$ concentrations were low compared to TKN in the MCEW. However, following PH-I and PH-II harvest activities, $\text{NO}_3 + \text{NO}_2$ became the dominant form of nitrogen in MCEW, with concentrations and loads at some sites more than double the TKN concentrations and loads, respectively (Table 5 & Table 6). Overall, the observed stream nitrogen concentration in MCEW increased significantly during the different harvest treatment periods. However, there were clear indications of vegetative recovery and decline in $\text{NO}_3 + \text{NO}_2$ concentration and load particularly at the original experimental PH-I clear-cut sites (e.g. W1).

This increase in stream nitrogen concentration following timber harvest is consistent with other studies that reported increased post-harvest stream nitrogen species. Binkley et al. (2004) reported the mean concentrations of nitrate and dissolved organic nitrogen to be 0.31 mg-N L^{-1} and 0.32 mg-N L^{-1} , respectively in forested streams across the United States. Also, mean concentrations of nitrate and dissolved organic nitrogen for the streams draining coniferous forests across the US have been reported to be 0.15 mg-N L^{-1} and 0.7 mg-N L^{-1} , respectively (Binkley et al. 2004). The nitrogen concentrations found in our study were relatively small compared to mean concentrations for forested streams across the US. For example, the TKN concentrations across all the treatment periods and for all watersheds in MCEW were $<0.18 \text{ mg-N L}^{-1}$. TAN concentrations during the entire study period, and for all watersheds in MCEW, displayed little variability and were $<0.015 \text{ mg-N L}^{-1}$. These values are consistent with the ammonia concentrations reported by Argerich et al. (2013) who investigated trends in stream nitrogen across the US and found that stream ammonia concentrations in 19 of the 21 study catchments had values of $\leq 0.01 \text{ mg-N L}^{-1}$.

One of the challenges in analyzing long-term water quality data is accounting for changes in the instrumentation used to analyze water samples. Detailed information on the Pre-disturbance ranges and detection limits of methods used for measuring different nutrients are tabulated in Gravelle et al. (2009). The minimum detection capabilities of instruments in earlier treatment periods appear to have impeded detection of lower concentrations, and therefore small changes in TKN and TAN concentrations. This is evident from the statistically significant changes that occurred during the PH-I and PH-II treatment periods. These changes were captured because during these treatment periods the NCASI laboratory improved the minimum detection capabilities. Similar to TKN and TAN, $\text{NO}_3 + \text{NO}_2$ concentrations did not exceed the aforementioned mean nitrate concentration of 0.31 mg-N L^{-1} except for the W1 watershed where a similar mean concentration of 0.33 mg-N L^{-1} and 0.4 mg-N L^{-1} was observed during the PH-I and PH-II treatment periods, respectively. Except for W1 and W4, the observed $\text{NO}_3 + \text{NO}_2$ concentrations do not exceed the mean nitrate concentration reported

for the forested watersheds in the western United States, 0.18 mg-N L^{-1} (Binkley et al., 2004).

The naturally consistently high concentrations of nitrogen from the W1 watershed have been largely attributed to the upstream presence of N-fixing alder through additional extensive headwater sampling. Gravelle et al. (2009) have reported the presence of dense alder (*Alnus* spp.) in the riparian areas of MCEW. Several studies have shown a strong relationship between the upstream presence of alder stands, that increase the soil N content, and increased stream nitrogen (Compton et al., 2003; Shafteel et al., 2012). Atmospheric N deposition is another factor that could contribute to increasing losses of N especially in the disturbed forest watersheds where the vegetation is unable to utilize the deposited N. A recent study in the Fish Creek watershed in Wyoming reported that 46% of total N input to the watershed was from atmospheric deposition (Eddy-Miller et al., 2016). In the context of MCEW, to our knowledge, there is no data on atmospheric N deposition that can help elucidate the depositional N contributions to the MCEW watersheds. However, data elsewhere in the region (e.g., National Atmospheric Deposition Program (NADP) site ID02 located in the Priest River Experimental Forest) does suggest increasing trends in total nitrogen deposition largely due to increases in NH_4 deposition. The increased $\text{NO}_3 + \text{NO}_2$ concentration and loads from all the MCEW could be explained by a reduced uptake of nutrients due to reduced vegetation cover, increased mineralization in the unharvested control watershed due to relatively wetter soils, and nitrification in the litter layer. Many studies have demonstrated that such factors lead to increased nitrogen availability and subsequent transport to the streams (Kreutzweiser et al., 2008; Palviainen et al., 2005, 2004; Piirainen et al., 2007). However, the elevated $\text{NO}_3 + \text{NO}_2$ concentration response from W1, following harvest and onset of forest regeneration, appeared to be in a steady recovery (Fig. 3). The first peak response (PH-I) occurred in 2004 following experimental harvest and broadcast burning, with concentrations receding until late 2008 when increases began to be observed until a second peak (PH-II) occurred in 2010. This second peak exceeded the PH-I peak, and it appears to be associated with the 2007 competition release herbicide application since the PH-II harvest in W1 occurred around 2011. Similar to the PH-I phase, the elevated $\text{NO}_3 + \text{NO}_2$ concentration receded to about twice the pre-harvest concentration levels within 5 years. This pattern of $\text{NO}_3 + \text{NO}_2$ response is also reflected at downstream cumulative sites W4 and W7. The stream nitrate recovery observed at MCEW is consistent with previous studies. For example, Oda et al. (2018) reported that nitrate recovery times can range anywhere from 0 to 20 years as a function of both biological processes such as vegetation uptake and hydrological interaction and transport processes.

Interestingly, similar to streamflow, we also observed a long-term increase in $\text{NO}_3 + \text{NO}_2$ concentrations at the control watershed (W3) during PH-II. However, much like streamflow, even though there could be factors unique to the catchment, there is also a rationale to believe overall environmental variability is being captured that otherwise would not in a shorter-term, non-BACIPS study design. When looking at mean $\text{NO}_3 + \text{NO}_2$ concentrations through treatment phases (see Table 4), W3 mean $\text{NO}_3 + \text{NO}_2$ concentrations were 0.07 mg-N L^{-1} during Pre-disturbance, 0.16 mg-N L^{-1} during Post-road, 0.10 mg-N L^{-1} during PH-I, and 0.33 mg-N L^{-1} during PH-II. What is notable is that the other control watersheds (W5 and W6) showed strikingly similar variability during the first three phases of the study before treatment activities occurred in those watersheds during PH-II. At W5, mean $\text{NO}_3 + \text{NO}_2$ concentrations were 0.05 mg-N L^{-1} during Pre-disturbance, 0.14 mg-N L^{-1} during Post-road, and 0.07 mg-N L^{-1} during PH-I. At W6, mean $\text{NO}_3 + \text{NO}_2$ concentrations were 0.08 mg-N L^{-1} during Pre-disturbance, 0.24 mg-N L^{-1} during Post-road, and 0.06 mg-N L^{-1} during PH-I. While it is possible other factors like stand age (Hedin et al., 1995; Vitousek and Reiners, 1975), increased tree mortality, increased decaying downed wood, or increasing alder vegetation could all be contributing factors to the recent W3 increase, further research would be necessary to provide more clarity.

4.2.2. Phosphorus

The results of forest management on phosphorus loads were much more subtle than nitrogen. TP concentration in the streams of the MCEW watersheds appeared to be relatively stable during the study period except in the four watersheds (W2, W5, W6 & W7) during PH-II when very small, yet statistically significant, changes occurred. These results are consistent with the findings of a recent study that evaluated long-term regional TP trends across the US and found that a large number of locations in the western US followed a general decreasing trend, most of which were statistically insignificant (Sprague and Lorenz, 2009). The statistically insignificant, yet smaller than reference period, TP concentrations following harvesting are consistent with the TP budgets evaluated before and after logging activities in watersheds of H. J. Andrews Experimental Forest in the western Oregon Cascades (Martin and Harr, 1989). Martin and Harr, (1989) reported OP concentrations from the clear-cut watersheds (0.014 mg L^{-1}) comparable to those reported in this study ($0.013\text{--}0.014 \text{ mg L}^{-1}$), but the OP loads were approximately 2–3 times the loads reported in this study. Although these higher loads were irrespective of the harvest and could be related to the consistently higher streamflow reported from the watershed both before and after the clear-cut harvest. Other factors that might explain these higher loads include significantly smaller watershed area (~ 11 times smaller than the current study) and the OP inputs from precipitation that ranged between 30 and 43% of the net OP loss from the clear-cut watershed. The inter-annual variability observed in the TP loads during the study periods (Fig. 6) can be partly associated with the changes in flow and suspended sediment loads. Karwan et al. (2007) assessed the effects of timber harvest on suspended sediment load in MCEW and found that the clear-cut harvesting produced a significantly higher suspended sediment load, but its magnitude returned to that of the pre-treatment sediment load within a year after harvesting. In the case of the partial-cut treatment, no significant differences in monthly sediment load were found between the treatment and control sites. The relatively stable TP concentrations and inter-annual variability of TP loads presented here are congruent with the aforementioned study that evaluated the effects of timber harvests on suspended sediment loads.

The pre-harvest OP concentrations found in MCEW were similar to the mean of $0.014 \text{ mg-P L}^{-1}$ for streams draining forests across the US, the mean of 0.01 mg-P L^{-1} for the streams draining coniferous forests in general, and the regional mean concentration of $0.008 \text{ mg-P L}^{-1}$ reported for forest streams in the western US (Binkley et al., 2004). Although OP concentrations from the control watershed W3 also showed a very slight increase, and this could again be related to minimum detectable limits, climatic variation, or some combination of soil mineralization processes. OP concentrations remained similar across the study area irrespective of treatment and time (Fig. 7). Any calculated increased loads of orthophosphate from MCEW was essentially driven by

increases in streamflow.

4.3. Longitudinal and adjacency effects

Although interesting long-term nutrient responses were observed in both upstream (W1, W2, W3) and downstream cumulative (W4, W7) watersheds, nutrient concentrations were generally lower in the downstream watersheds than the upstream headwater watersheds (Fig. 10). This attenuating effect at a longitudinal scale can also be seen in mean annual load estimates (Table 4), where the loads of $\text{NO}_3 + \text{NO}_2$ and OP decreases as one moves longitudinally down the stream network (e.g. during PH-II $\text{NO}_3 + \text{NO}_2$ moves from $3.95 \text{ kg ha}^{-1} \text{ yr}^{-1}$ at W1 downstream to $1.73 \text{ kg ha}^{-1} \text{ yr}^{-1}$ at W4 to $1.24 \text{ kg ha}^{-1} \text{ yr}^{-1}$ at W7). This can be partly attributed to the dilution effect and in-stream processing as the water flows downstream. Studies have shown the magnitude of nutrient concentrations tends to decrease with an increase in the catchment area (Jacobs et al., 2017), and the presence of denitrification and assimilation processes can substantially reduce the nitrate concentrations in streams (Shinozuka et al., 2017). Bernhardt et al. (2003) noted that the absence of in-stream processing would have increased the loads of nitrate-nitrogen from the Hubbard Brook Experimental Forest by about 80–140%. Furthermore, although the streambed occupies a small fraction of the watershed, generally streambed sediments store significant amounts of phosphorus. For example, a study in a nutrient-impacted catchment by Lannergård et al. (2020) reported high proportions of organic and iron-bound phosphorus in the sediments of forested headwater streams.

It is likely that not only does the harvest intensity, but also the location of the harvest relative to the stream plays a critical role in explaining the nutrient concentration dynamics and recovery in the stream. This is evident in W2 (see Fig. 3) where the higher intensity coupled with the harvest adjacent to the stream showed relatively strong (albeit slightly delayed) $\text{NO}_3 + \text{NO}_2$ response and longer recovery times compared to the Phase-I response and recovery. Similarly, in W1 after the competition release herbicide application, the $\text{NO}_3 + \text{NO}_2$ concentration recession curve was interrupted when a PH-II harvest was conducted near the W1 sampling point. Similar response signals caused by the proximity of the harvest to the flume sampling points can be noticed in the $\text{NO}_3 + \text{NO}_2$ response from W5, W6, and W7 (see Fig. 3). However, isolation of these adjacent harvest responses is confounded by the simultaneous PH-II forest management activities occurring further upstream within these watersheds.

4.4. Land use and nutrient response

Although evaluation of the nutrient response between different land uses is beyond the scope of this study, it is important to put these

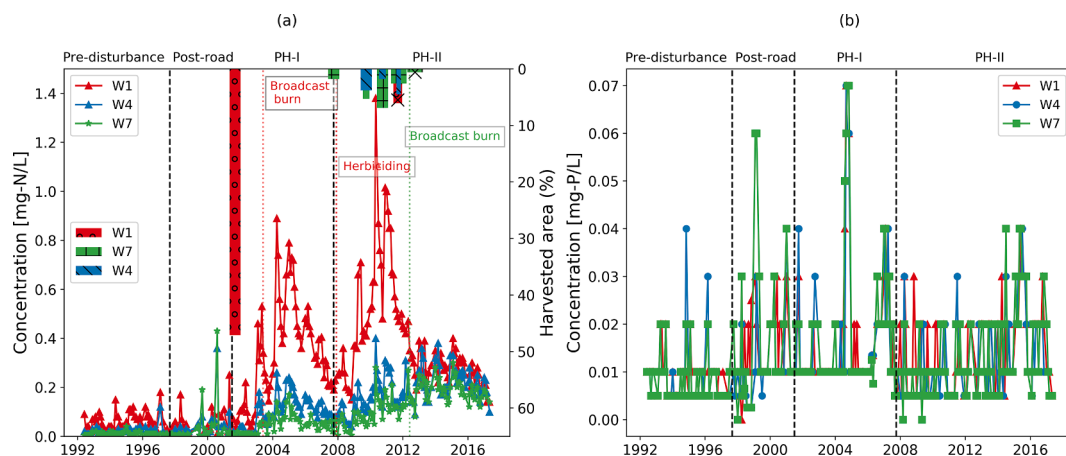


Fig. 10. Response in space and time of commercial forest management practices on $\text{NO}_3 + \text{NO}_2$ (a) and OP (b) concentrations.

findings in a larger perspective. Previous studies have all suggested that compared to the forest-dominated watersheds, the magnitude of nitrate-nitrite N, soluble reactive phosphorus, and TP concentrations and loads are higher in urban (Brett et al., 2005; Duan et al., 2012; Poor and McDonnell, 2007) and agriculture (McColl et al., 1977; Poor and McDonnell, 2007; Quinn and Stroud, 2002; Verheyen et al., 2015) dominated watersheds. Comparatively, the nutrient concentrations in the forested MCEW streams are much lower than streams draining agricultural and urban watersheds. However, the slight increase in nutrient loads in this study is similar to the recent observations of nutrient buildup and subsequent loads from undisturbed forests elsewhere in the US (Loupe et al., 2007; Miller et al., 2005). Due to both a temporary effect and substantial longitudinal nutrient uptake observed in remote forestland streams like MCEW streams, any potential downstream negative effects will depend on proximity to municipalities and other areas of concern (e.g. lakes). A recent survey of undeveloped water bodies across the US indicates that between 2004 and 2014 the percentage of stream length that is naturally oligotrophic (TP < 0.01 mg-P/L) has decreased by 23.8% and the percent of naturally oligotrophic lakes has decreased by 18.2% between 2007 and 2012 (Stoddard et al., 2016). This further emphasizes the need for understanding the sources and processes that affect nutrient dynamics and management approaches for nutrient loads from forested systems. This is becoming an increasing concern for municipalities across the US that are considering localized forest fuel management strategies to lower the risk and impact of wildfire within their municipal water supplies (Smith et al., 2011; Warziniack and Thompson, 2013). Approximately 80 percent of the drinking water supply across the US originates from forested lands providing drinking water to more than 68,000 communities (USFS, 2020). Annual costs of eutrophication of US fresh water from excessive nutrient loading is estimated to be \$830 million and it is likely to be significantly higher in regions affected by wildfires (Bladon et al., 2014). Estimates suggest that after a wildfire the increase in the sediment load can range between 1 and 1459 fold of unburned conditions. Similarly, the increase in the NO₃ and TP loads can range between 3 and 250 fold and 0.3 to 431 fold of unburned conditions respectively (Smith et al., 2011). In contrast to these wildfire impacts, timber harvesting resulted in a relatively low increase in NO₃ + NO₂ (4.9 to 19 fold) and TP (0.66 to 1.25-fold) loads compared to the Pre-disturbance phase. Municipalities must weigh the risk and impact of localized forest thinning strategies on their water quality and quantity relative to the potential impacts of catastrophic wildfire events.

5. Conclusions

Understanding stream nutrient concentration dynamics and nutrient loads from watersheds draining relatively undisturbed, as well as actively managed forests using contemporary management practices, is imperative to improve management of land resources and downstream water quality. A quarter-century of monitoring at MCEW provided an important understanding of the background nutrient concentrations in relatively undisturbed forests as well as the effect that contemporary commercial forest management activities have on nutrient response and loads. Overall, the contemporary commercial forest management practices applied in the MCEW resulted in relatively small increases in nutrient loads and suggest that Idaho Forest Practices Act regulations and BMPs are effective in minimizing the delivery of particulate-bound pollutants. Statistically significant increases in stream NO₃ + NO₂ concentrations and loads were observed following timber harvest treatments. In the case of OP, concentrations remained near minimum detectable concentrations. Longitudinal evaluation of stream nutrient concentrations indicates instream dilution and nutrient assimilation processes. The unharvested control watershed also displayed increases in concentration, streamflow, and loads of NO₃ + NO₂. This increase, however, was smaller than in the harvested watersheds and potentially driven by a combination of climatic variability and subtle forest

changes, further demonstrating the effectiveness of employing a BACIPS design for long-term watershed evaluation. In summary, contemporary forest management activities increased stream NO₃ + NO₂ concentrations and loads following timber harvest activities, but these effects were also attenuated in downstream reaches and reduced through time as vegetation regrowth occurred.

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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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.foreco.2021.119312>.

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