University of Alberta

The role of fine sediment in phosphorus dynamics and stream productivity in Rocky Mountain headwater streams: Possible long-term effects of extensive logging

by

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Abstract

Headwater streams in Alberta's Rocky Mountains are important in regulating aquatic ecosystem function and a range of downstream water resources but are vulnerable to stresses imposed by disturbances including those exerted by logging. The objectives of this research were to determine if the legacy of past forest harvesting impacts could be detected in altered sediment-nutrient dynamics and primary productivity in headwaters in the Rocky Mountains of southwestern Alberta. A descriptive, process-based case study was conducted in an undisturbed-disturbed watershed pair where one watershed had undergone extensive harvesting ending in 1990. The disturbed watershed was found to have higher concentrations of suspended solids and fine streambed material, and considerably greater levels of aqueous and particulate phosphorus (P). Primary productivity was much higher in the disturbed system, likely caused by elevated P levels. This study illustrates the potential for logging disturbance to produce long-lived impacts on stream ecology in critical headwater regions.

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Chapter 1: Thesis Introduction

Headwaters generally refer to the first and second order channels from which larger drainage networks originate and can represent up to 80 percent of the cumulative length of larger basin-scale drainage networks (Benda *et al*., 2005). In larger basins originating from the Rocky Mountain regions, a disproportionate amount of the total stream discharge of these larger basins often originates in these mountainous headwater areas (Owens *et al*., 2005). Thus, the degree of disturbance in headwater regions can have strong influence on both the quality and quantity of the downstream water supplies. However, headwater streams often receive comparatively less attention from land and resource managers than do larger downstream rivers (Macdonald *et al*., 2003; Benda *et al*., 2005).

Of the many anthropogenic stresses placed on headwater stream systems in the northern Rocky Mountains, few are as extensive as forest harvesting disturbance. Forest harvesting and logging road construction can impact headwater streams by affecting numerous physical, chemical and biological characteristics which can directly and indirectly affect water quality and aquatic ecosystems (Megahan and Kidd, 1972; Swanson and Hillman, 1977; Bestcha, 1978; Corbett *et al*., 1978; Troendle, 1983; Anderson and Potts, 1987; Nip, 1991; Binkley and Brown, 1993; Gomi *et al*., 2005). Timber harvesting can increase water yield and alter timing and magnitude of peakflow/stormflow events, due to reductions in evapotranspirative losses, interception and altered snowpack accumulation (Swanson and Hillman, 1977; Troendle, 1983). Compaction of soils associated with skid trails, landings, forest roads and heavy equipment activity can decrease infiltration and redirect surface and subsurface water to streams more rapidly and in larger quantities (Macdonald *et al*., 2003). Accordingly, increases in streamflow volume and timing can alter channel morphology and accelerate in-stream erosion and bank failure. Forestry operations can impact light, temperature, water chemistry, and organic material regimes, but the most deleterious effects stem from the increased input of fine sediment (<2mm) (Beschta, 1978; Golladay *et al*., 1987; Waters, 1995; Wood and Armitage, 1997).

Forest harvesting operations increase the availability of fine sediment sources, usually in the form of unconsolidated material from cut/fill slopes associated with roads, road ditches, road-stream crossings, skid trails, landings, silvicultural activities (i.e., scarification), and other bared areas associated with forestry operations (Reid and Dunne, 1984; Nip, 1991; Megahan and King, 2004). The transport and delivery of fine sediment to watercourses is similarly accelerated by the construction of roads which serve as efficient conduits for sediment transport during rainstorms or snowmelt periods (Macdonald *et al*., 2003). Increases in stream suspended sediment concentrations have been widely reported during and following forestry operations, including in northern Rocky Mountain headwaters (Anderson and Potts, 1987; Nip, 1991). The negative effects of elevated suspended solids concentrations are well documented (Davies-Colley and Smith, 2001; Gomi *et al*., 2005; Owens *et al*., 2005) and are reported to be limited to a short period following forestry activities. Vegetation recovery and soil stabilization can result in rapid declines of point source sediment pollution

(Anderson and Potts, 1987; Nip, 1991; Macdonald *et al*., 2003). Recently however, it has been demonstrated that headwater streams, traditionally considered incapable of storing significant quantities of fine sediment, have a much higher capacity to retain more fine material than previously expected (Zimmerman and Church, 2001; Benda *et al*., 2005). Significant fine sediment retention and storage in the interstitial spaces of coarse beds, in association with large woody debris (LWD) and in pools has been reported in headwater streams in the northern Rockies following disturbance (Megahan, 1975; Benda *et al*., 2005; Little *et al*., 2012). However, little is known about the transport, fate and legacy effects of these materials on water quality and ecosystem health (Jordan, 2006).

Fine sediment generated by forestry activities has historically been considered as primarily directly detrimental to stream biota and water users (Gomi *et al*., 2005) even though fine sediment has also been identified as an important agent of transport and storage of various contaminants (Forstner, 1987; Ongley *et al*., 1992), including nutrients like nitrogen (N) and phosphorus (P). N and P are frequently growth-limiting nutrients in aquatic systems and are often associated with fine sediment (Chessman *et al*., 1992). P especially tends to be bound to and transported by the finest sediment fractions (<63 μm) (Stone and Mudroch, 1989; Stone and Droppo, 1994). Thus sediment plays an important role in the delivery and storage of nutrients in streams (Walling *et al*., 1997). P dynamics have received relatively more attention because P is often the limiting nutrient for primary production in many freshwater ecosystems (Chessman *et al*., 1992; Reddy *et al*., 1999; Bowes *et al*., 2003), and its bioavailable inorganic form (soluble reactive phosphorus (SRP)) is readily adsorbed and transported by sediment particles (Walling *et al*., 1997). In aquatic environments, P adsorbed to fine sediment can undergo reversible desorption reactions thus becoming available for uptake by primary producers (Froelich, 1988; House, 2003). The rate and magnitude of these sorption/desorption reactions are influenced several physicochemical properties of sediment and chemistry of the water (Froelich, 1988). Headwater streams are particularly vulnerable to P enrichment because their typically P-limited state promotes P desorption from sediment and even minor increases in soluble reactive phosphorus (SRP) can alter productivity (Withers and Jarvie, 2008). Fine streambed sediment can strongly influence P concentrations, particularly in small streams with high bed sediment to volume ratios (House and Denison, 2002). Streambeds enriched with fine sediment can release SRP via desorption at the benthic interface as well as participating in sorption/desorption reactions if fine particles are re-entrained (Withers and Jarvie, 2008). P cycling in aquatic systems is governed by a complex set of biophysical and environmental controls. However, the particle size of benthic and suspended materials is a very important factor that influences the source, transport and fate of P in the water column (Lottig and Stanley, 2007).

The enrichment of P in surface waters and subsequent eutrophication results in a host of environmental, social, and economic problems including threats to human health, increased water treatment costs, and losses in species diversity and abundance (Withers and Jarvie, 2008). In mountain headwater streams, which typically are shallow and clear allowing ample light penetration, P enrichment causes increased growth of primary producers like periphyton, filamentous green algae and macrophytes (Dodds, 2003). In headwater streams, where metabolic demand far exceeds supply, increases in primary productivity can be drastic and affect local and downstream ecology and water quality. Increased P availability in oligotrophic systems can drive a shift from an ecosystem reliant on allocthonous inputs and microbial P cycling supporting autotrophic and heterotrophic organisms, to a system with reduced species diversity, dominated by autotrophs (Biggs, 2000). Such changes in species composition threaten biological integrity and sustainability of stream ecosystems (Withers and Jarvie, 2008). Although P-induced increases in primary productivity can damage aquatic ecosystems, periphyton and algae exert significant controls on P cycling. Periphyton assimilates P from the overlying water column and as it diffuses from bed sediments across the benthic interface, reducing aqueous concentrations of P and limiting downstream nutrient flux (Dodds, 2003). P assimilation by periphyton and phytoplankton is responsible for up to 15% of the P flux in riverine systems (Withers and Jarvie, 2008). While primary producers may reduce export of P from disturbed headwaters, they shorten P spirals and may prolong effects of P enrichment resulting from disturbance locally (Newbold, *et al.*, 1983).

The effects of forest harvesting disturbance in headwater systems have been extensively explored in the context of fine sediment pollution and its short-term direct effects on various forms of biota, primarily salmonids. A knowledge gap exists however, concerning the long-term effects of harvesting on sediment regimes in low-order streams, particularly in the context of potential effects of fine sediment storage within streambeds and its subsequent role in providing a long-term source of suspended solids. Similarly, fine sediment-facilitated nutrient enrichment has been primarily studied in agricultural and urban settings, usually in high-order rivers. Little research however, has focused on fine sediment production by forest disturbance and its effects on nutrient dynamics and productivity in oligotrophic headwater streams, particularly in the northern Rocky Mountains. The broad objectives of this thesis were to explore these issues to address these knowledge gaps.

The overall objectives of this study were to determine if sediment associated nutrient storage and transport after extensive forest harvesting disturbance in headwater catchments is an important factor in the long-term legacy of forest harvest effects on water quality and aquatic ecology in northern Rocky Mountain watersheds. Chapter 2 reports on research describing the surface hydrology, water chemistry, suspended sediment regimes and primary productivity of two headwater catchments (disturbed and undisturbed) in the Rocky Mountains of southern Alberta to provide knowledge on the longevity of forest harvest effects on sediment-nutrient dynamics in Rocky Mountain headwater streams. The second data chapter (Chapter 3) examines the particle size distributions, P content/speciation and P-sediment dynamics in the streambeds of the same two Rocky Mountain headwater streams. This information is used to support inferences about the potential for forest harvesting disturbance to alter streambed sediment and nutrient properties, producing impacts on aquatic ecology. The conceptual sediment-P dynamics investigated in Chapters 2 and 3 are illustrated in Figure 1-1. Chapter 4 (Synthesis) summarizes and combines the results of the two data chapters, and provides a more complete perspective on sediment-nutrient dynamics in oligotrophic low order streams. From this, broader inferences are discussed, and future research is suggested.

Tables and Figures

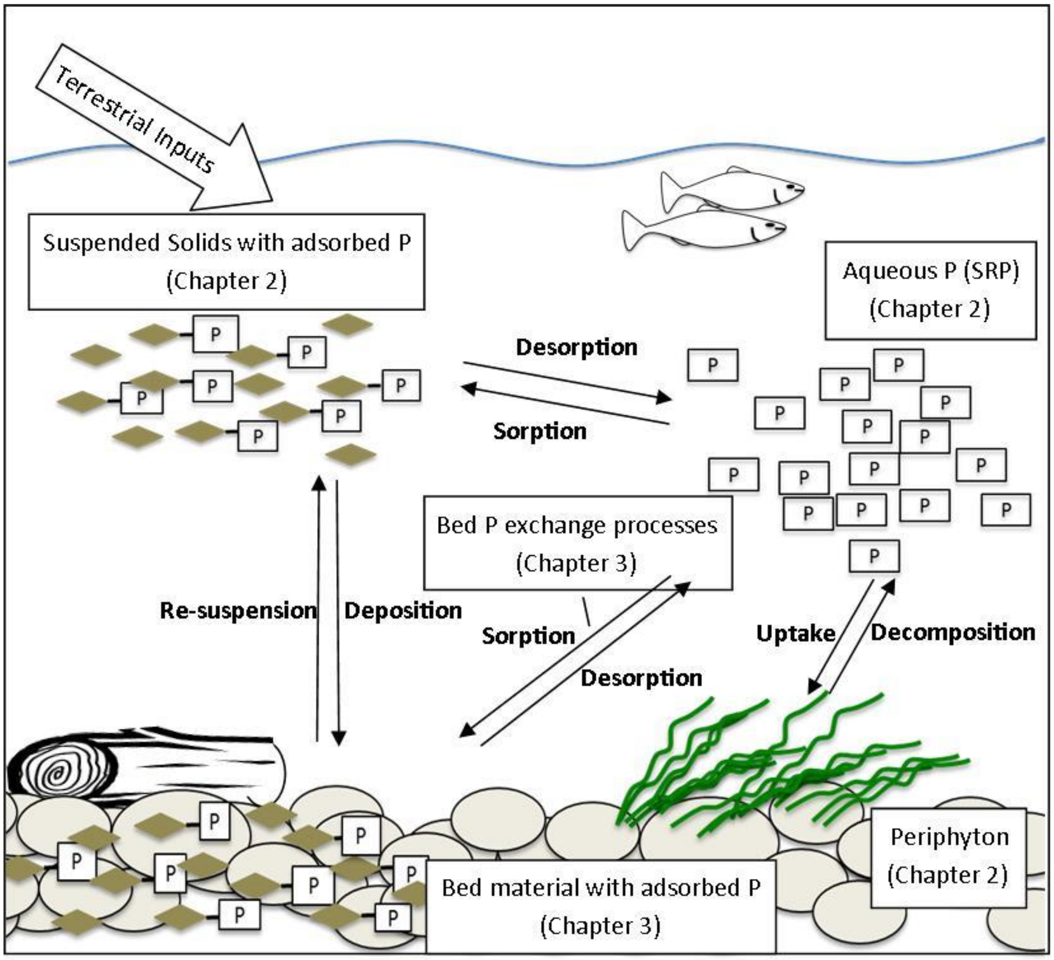

Figure 1-1. Conceptual diagram of stream sediment-phosphorus dynamics, illustrating processes which govern concentrations of each pool of phosphorus/sediment and where each were addressed in the present study.

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Chapter 2: Long term effects of logging on suspended sediment-phosphorus dynamics and primary production in a northern Rocky Mountain headwater watershed, Alberta, Canada

2.1 Introduction

Headwater catchments (the first and second order channels in which a drainage network originates) can represent up to 80 percent of cumulative stream network length (Benda *et al*., 2005) and the disturbance history of these watersheds has a strong influence on the downstream water quality of larger drainage networks (MacDonald and Coe, 2007). Forest harvesting and road construction are known to cause major disturbances to forested watersheds (Webster *et al*., 1992; Binkley and Brown, 1993; Kreutzweiser and Capell, 2001; Macdonald *et al*., 2003), by altering numerous physical, chemical and biological characteristics of aquatic systems, but the increased input of fine sediment has been identified among the most deleterious (Beschta, 1978; Golladay *et al*., 1987; Waters, 1995).

Forestry operations increase sediment availability by the construction of roads, skid trails and other bared areas and accelerate the delivery of sediment to watercourses by decreasing infiltration because of soil compaction and providing conduits for sediment delivery along road ditches, landings, and stream crossings (Anderson and Potts, 1987; Macdonald *et al*., 2003). The impacts of increased fine sediment concentrations have been extensively studied, particularly in regards to negative effects on fish, fish habitat and a broad spectrum of aquatic biota (Bilotta and Brazier, 2008). Additionally, increases in suspended solids can decrease water quality, damage infrastructure and decrease recreational value of streams (Anderson and Potts, 1987; Holmes, 1988). Because of the turbulence and high flow velocity typically associated with mountainous headwater streams, it was previously thought that fine sediment would be efficiently routed out of small watersheds and that negligible amounts of fine sediment would be stored in the river channel (Benda *et al*., 2005). Studies investigating sediment production during and following logging operations in similar headwater settings report rapid post-harvest recovery of suspended solids concentrations to pre-disturbance levels (Anderson and Potts, 1987; Nip, 1991; Megahan and King, 2004). However, few studies in the Northern Rockies have assessed the long term (>10 years) effects of sediment pollution following logging, and the storage of fine sediments in headwater streams has been documented in multiple cases (Benda and Dunne, 1997, Gomi *et al*., 2005, Little *et al*., 2012). These studies suggest that despite high energy flow conditions usually present in these streams, fine sediment introduced during and shortly after logging can be stored and potentially re-entrained in headwater systems long after sediment sources have stabilized and elevated post-disturbance sediment delivery into streams has recovered.

Sediment-associated contaminant transport and storage has been extensively researched in the context of agricultural and urban settings, particularly in higher order systems (Dillon and Kirchner, 1975; Froelich, 1988; Holtan *et al*., 1988; House, 2003; Withers and Jarvie, 2008), but relatively little is known about sediment-nutrient dynamics in headwaters systems, and less so in the Northern Rocky Mountain region. These headwater regions produce significant quantities of high quality water for multiple uses (Emelko *et al.*, 2011) and knowledge of disturbance effects on water quality is necessary to manage this valuable resource. Sediment, particularly the finest fractions (<63 μm) which are most easily suspended and transported, are known to be extremely important transport and storage agents of contaminants and nutrients (Forstner, 1987; Owens *et al*., 2005). Phosphorus and nitrogen are frequently growth-limiting nutrients in many aquatic ecosystems (Chessman *et al*., 1992) and tend be closely associated with sediment for delivery to, and storage within, streams (Walling *et al*., 1997). The source, transport and fate of P is closely linked to fine sediment (<63 μm), and particulate P exists in a number of forms of differing bioavailability (Stone and Mudroch, 1989; Stone and Droppo, 1994). After sediment bearing nutrients are delivered into streams, nutrients can be released by reversible sorption-desorption reactions and become available for uptake by aquatic organisms (Froelich, 1988; House, 2003). This nutrient enrichment can contribute to eutrophication and alterations to downstream ecosystems and water quality (Walling *et al*., 1997; Owens *et al*., 2005). Oligotrophic streams, typical of the Northern Rockies, can be particularly sensitive to small increases in nutrients (Withers and Jarvie, 2008), yet limited research has investigated sediment-nutrient dynamics of Rocky Mountain streams. Given the large influence that headwater Rocky Mountain streams have on higher order downstream river basins, declines in water quality and ecosystem degradation caused by sediment-facilitated nutrient enrichment have significant implications for land management in the Rocky Mountain region.

The broad objectives of this study were to investigate suspended sediment nutrient dynamics in a disturbed and undisturbed Rocky Mountain headwaters catchment pair to explore potential mechanisms regulating long-term effects of forest harvesting activities on stream nutrients and stream ecology. Specific objectives included exploring the comparative a) hydrologic regime, b) suspended sediment and aqueous nutrient water quality, c) suspended sediment associated nutrient availability, and d) algal productivity.

2.2 Materials and Methods

2.2.1 Study Area

The study was conducted in Star Creek and Smith Creek watersheds in the headwaters of the Oldman River basin in the Rocky Mountain region of southwestern Alberta near Crowsnest Pass (49°37'N, 114°40'W) (Figure 2-1). Star and Smith Creeks are first order streams, originating at, or very near, the Alberta-British Columbia border within the Southern Rockies hydrologic region in Alberta (Wagner, 2010). Star and Smith Creek watersheds drain 1035 and 697 ha, respectively, and are situated along the eastern edge of the Flathead Mountain range and the High Rock range, which forms the northern extension of the Flathead range north of highway 3 (Crowsnest highway). The geology and soils of both watersheds are complex, but strongly similar. Both watersheds are underlain by Paleozoic bedrock at high elevations, consisting of Cambrian carbonates, shale and sandstone, Devonian limestone and dolomite, Mississippian shale and carbonate, Jurassic sandstone and shale, and Cretaceous sandstone and shale (OWC, 2012). At mid elevations the Belly River-St. Mary Succession formation, which consists of Upper Cretaceous non-marine mudstone, siltstone and fine-grained sandstone with subordinate, coarser grained sandstone layers, is predominant. Lower elevations are underlain by the Alberta Group, which consists of Late Cretaceous shales (Figure 2-2) (OWC, 2012). The soils of both catchments are imperfectly drained Brunisols with weak horizon development characteristic of higher elevation northern sites (Bladon *et al*., 2008).

Elevation of the Star Creek watershed ranges from 1482-2631 m, while Smith Creek watershed is consistently higher by approximately 200 m across all elevations (1657-2768 m). Aside from minor differences in elevation, the two watersheds have similar longitudinal profiles (Figure 2-3), basin elevation distributions, and slope distributions (Figure 2-4). Star Creek has a northeast aspect and the aspect of Smith Creek is east-northeast. Physical basin characteristics for both study watersheds are summarized in Table 2-1.

Forest vegetation in both watersheds are characteristic of montane ecozones dominated by lodgepole pine (*Pinus contorta* var. *latifolia*) at lower elevations, subalpine ecosystems dominated by engelmann spruce (*Picea Engelmannii*) and subalpine fir (*Abies lasiocarpa*) at mid-elevations, and alpine ecozones at higher elevations, dominated by bare rock and alpine meadow vegetation (Bladon *et al*., 2008). Smith Creek watershed underwent clear-cut forest harvesting starting in the 1960s, through to the mid ‘90s, and during this period at least 44% of the forest cover was removed. An extensive network of logging roads was built during harvest operations and two stream crossings were established. The crossings have since been reclaimed, along with the many of the in-block access roads, however the main haul road and some secondary roads remain and serve as popular trail routes used by recreational all-terrain vehicle (ATV) users. Star Creek watershed is largely undisturbed with no documented history of commercial forest harvesting, however some linear disturbance is present in the form of several ATV trails and seismic exploration cut lines (Figure 2-5). This watershed will be used as a reference to compare with Smith Creek (disturbed).

2.2.2 Study Design

A descriptive, post-hoc reference/impact study catchment design was implemented for this study. Smith and Star Creeks were selected because they are similar with respect to most bio-physical characteristics. The major difference between the watersheds is the extent of disturbance in Smith Creek due to extensive forest harvesting. The absence of pre-disturbance data from these watersheds, or use of replicated reference/impacted watersheds (Swanson and Hillman, 1977) would be considered a major limitation in this study design if the primary objective was the empirical description of differences in sediment, nutrient, and algal production among reference and harvested watersheds. However, the primary focus of this study was the exploration and description of ecohydrological processes and mechanisms regulating geochemical linkages between sediment and phosphorus regimes and their subsequent influence on primary productivity. In this context, Star Creek serves as a generally representative undisturbed reference catchment for Smith Creek, which has been “treated” with a long-term history (50+ years) of extensive forest harvesting. Smith Creek is generally representative of extensively harvested watersheds along the High Rock range north of the Crowsnest Pass region. Two years (2011, 2012) of streamflow, water quality, suspended sediment, and stream algal productivity data were collected from each catchment from April 2011 until October 2012 to enable exploring and describing differences in sediment-phosphorus dynamics and their linkage with aquatic productivity in undisturbed and disturbed Rocky Mountain watersheds.

2.2.3 Surface Water Hydrology

Stream discharge (Q) was continually monitored for the duration of the study to generate annual hydrographs for each watershed. Natural control sections were selected for the establishment of hydrometric gauging stations at the outlet of each of the study basins (Figure 2-6). Stage and stream temperature were monitored continuously (at 10-minute intervals) using HOBO (model U20, Onset Computer Corp.) water-level data loggers deployed in stilling wells. Instantaneous discharge measurements were collected every 2-3 weeks at the gauging stations using standard 10 point area-velocity current metering techniques. Swoffer (Model 2100. Swoffer Instruments Inc.) and Sontek Flowtracker (YSI Inc.) current velocity meters were used to measure stream flow. No discernible difference in velocity measurements between current meters was found, thus the instruments were used interchangeably. Manual stage readings were recorded from permanent staff gauges in order to develop stage-discharge relationships. Stage-discharge relationships were created for 2011 and 2012, and were used in conjunction with the 10-minute stage data to calculate continuous stream discharge estimates.

2.2.4 Suspended Solids

Total suspended solids (TSS) and turbidity were continuously monitored for the ice-free (May-October) periods of 2011 and 2012 using Isco automated water samplers (Model 6712, Teledyne Isco) deployed at the hydrometric gauging stations of Star and Smith Creeks. Daily composite samples (1 L) composed of 4-250 ml sub-samples, drawn from the stream every 6 h, were collected and transported to the laboratory for TSS and turbidity analyses (Silins *et al*., 2009). TSS concentrations ($mg \cdot L^{-1}$) were determined using standard filtration-gravimetric methods (Stednick, 1991) and turbidity (NTU) was determined using a bench top Hach turbidimeter (Model 2100N IS, Hach Co.).

Composite suspended solids samples were collected by deploying *in-situ* time-integrating fluvial sediment samplers to collect sufficient quantities of solids for P speciation analysis from the study streams, which typically have very low suspended sediment concentrations (Phillips *et al*., 2000). The samplers were constructed according to Phillips *et al*. (2000) but scaled down by a factor of 0.75 to ensure constant submersion in the small, low-order streams of the study catchments. Four samplers were deployed in each stream during the ice-free season of 2011 and 2012. In 2011, the samplers were spatially distributed along the length of the stream (Figure 2-6) to provide insight into spatial variability of sediment-P speciation among catchments. Sediment in the samplers was collected in late June after the peak of the spring snowmelt freshet and again in late fall/early winter to evaluate and compare the sediment properties during the first (early melt to post-peak of the annual snowmelt freshet) and second half (post-peak of the annual melt freshet through late summer/fall baseflows) of the ice-free flow season. The sampling protocol was repeated in the spring of 2012. In the falling limb of 2012, four Phillips samplers were clustered within 100 m (upstream) of each hydrometric gauging station to serve as replicates to provide an estimate of variability in P speciation from single composite samplers (Figure 2-6). All composite sediment samples were stored in coolers while sampling and frozen within 6 hours for storage prior to phosphorus speciation analysis.

A sequential extraction procedure (Psenner *et al*., 1984) was used to characterize three operationally defined particulate P fractions; non-apatite inorganic P (NAIP), apatite P (AP), and organic P (OP) (Stone and English, 1993). NAIP is considered to be bioavailable (analogous to aqueous soluble reactive phosphorus (SRP)) and is comprised of loosely sorbed P, reductant-soluble P, and P bound to metal oxides (Psenner *et al*., 1984). The AP fraction includes P bound in mineral lattices, primarily apatite minerals; P contained in such organized mineral structures is considered to be inert and unavailable for uptake (Lukkari *et al*., 2007). OP is present as structural components of organic molecules (e.g., DNA, phospholipids, phosphosaccharides) and is considered to be available for uptake, but only after mineralization (Bostrom *et al*., 1988). Accordingly, NAIP, OP, and AP fractions represent a gradient of P availability from most to least bioavailable for uptake. The extracts of the sequential P extractions were analyzed using a Westco SmartChem 200 discrete wet chemistry analyzer according to the molybdenum blue method (EPA, 1993). Total particulate phosphorus (TPP) was calculated as the sum of the NAIP, AP and OP fractions.

2.2.5 Water Chemistry

Manual, depth-integrated grab samples were collected approximately every 2-3 weeks (concurrently with discharge measurements) at the hydrometric stations. Samples were collected in well-mixed riffles in acid washed (10% HCl), triple rinsed, high density polyethylene bottles (Bladon *et al*., 2008) and kept in a refrigerator at 4°C. The samples were delivered within four days to the University of Alberta Biogeochemical Analytical Service Laboratory for chemical analysis. Samples were analyzed for soluble reactive phosphorus (SRP), total dissolved phosphorus (TDP), total phosphorus (TP), nitrate + nitrite (NO_3^- + NO_2^-), total dissolved nitrogen (TDN), and total nitrogen (TN) concentrations. TN (unfiltered), NO_3^- + NO_2^- (0.7 µm filter), and TDN (0.45 µm filter) concentrations were determined by automated cadmium reduction (Method 4500; NO_3^-:F) using a Lachat QuikChem 8500 multichannel flow injection analyzer (Greenberg *et al*., 1999). TN and TDN samples were digested with potassium persulfate ($K_2S_2O_8$) prior to analysis. Total particulate nitrogen (TPN) concentration was determined indirectly as the difference between TN and TDN. TP (unfiltered), TDP (0.45 µm filter), and SRP concentrations were determined by automated ascorbic acid reduction (Method 4500; P:F) using a Lachat Quikchem 8500 multichannel flow injection analyzer (Greenberg *et al*., 1999). TP and TDP samples were digested with potassium persulfate ($K_2S_2O_8$) prior to analysis.

2.2.6 Primary Production

Primary production of benthic algal communities was estimated from periodic measurement of ash-free dry mass (AFDM) and chlorophyll *a* (Chl*a*) content of algal growth on the streambed. Periphyton samples were collected from each stream during the ice-free seasons of 2011 and 2012 by deploying replicates of unglazed porcelain tiles (138.24 cm^2 per tile) to serve as fixed area artificial streambed substrate in midstream riffles in close proximity (within 50 m) to the hydrometric gauging stations. Fixed-area samples were collected by scraping and rinsing tiles into plastic scintillation vials every 4-6 weeks through the ice-free season and frozen until fluorometric analysis of Chl*a* (Sartory and Grobbelar, 1984) and determination of AFDM (Aloi, 1990) in the laboratory. Four tiles were deployed in Smith Creek for both years of the study, and 2-3 tiles were deployed in Star Creek due to resource limitations and a tile lost during high-flow events.

2.2.7 Statistical Analyses

All data analyses were performed using R statistical software (R Core Team, 2013) using a significance threshold of $\alpha = 0.05$. The data were categorized by flow regime to limit hysteretic variability and better compare the two watersheds. The rising limb and early summer storm flow category (hereafter referred to as RL) is represented by Julian days 121-176 in 2011 and 108-184 in 2012. The falling limb and base flow category (FL) consists of Julian days 177-365 in 2011 and 1-107, 185-285 in 2012.

2.2.7.1 Suspended Solids

<u>2.2.7.1.1 Physical Properties</u>

Because both untransformed and log-transformed TSS and turbidity data were non-normally distributed across year and flow period categories using Shapiro-Wilk tests and Q-Q plots (α=0.05), non-parametric Wilcoxon-signed rank tests were used to test for differences between distributions of TSS and turbidity in Star and Smith Creeks by flow category. The influence of periodically very low TSS concentrations (below the 0.1 $mg \cdot L^{-1}$ detection limit using the filter-gravimetric method) on the seasonal distribution of sediment concentration were estimated using empirical censored distribution functions. The Kaplan-Meier method (Helsel, 2009) uses empirical censored distribution functions to produce revised estimates of water quality data distributions containing censored observations (below detection limits) which has been shown to produce more reliable estimates of sample means and sums rather than rather than representing these observations as ½ of the detection limit as has been typical in previous water quality research (Helsel, 2009). To explore sediment relationships with stream discharge, TSS-Q and turbidity-Q relationships were tested for significance. For regression analyses, TSS concentrations below detection limits were assigned a value equal to ½ of the limit of detection. Overall tests for coincidental regression were used to compare the discharge-TSS relationships between Star and Smith Creeks (Zar, 1999).

<u>2.2.7.1.2 Chemistry of Suspended Solids</u>

P speciation data met assumptions of normality across flow periods for each catchment and Student's t-tests were used to compare mean concentrations for all P species in Star and Smith Creeks. Variation in concentration of particulate P species from upper to lower stream reaches was explored by regressing P concentration against relative stream reach position within each catchment (top to bottom of the stream as a fraction of total stream length). Data from both study hydrologic seasons was combined to ensure significant relationships between catchment position and P concentration. Overall tests for coincidental regression were performed for each P species (except OP) to compare the relationships between catchment position and P in Star and Smith Creeks (Zar, 1999). OP was not significantly related to catchment position and was thus excluded from the tests for coincidental regression.

2.2.7.2 Water Chemistry

Concentration data for many of the various forms of P and N collected were not normally distributed according to P-P plots, Q-Q plots and Shapiro-Wilk tests, which is typical of water-quality data (Bladon *et al*., 2008). Non-parametric techniques were thus necessitated for the analysis of water chemistry. Although some data sets for some parameters in particular time periods met the assumptions of normality (i.e., TP in the FL flow category), all were tested using non-parametric techniques to enable consistent comparisons. Empirical censored distribution functions were fit to data sets with non-detects (censored data) using the Kaplan-Meier method as outlined above. The detection limits for the aqueous N and P forms were as follows: TP-3 $\mu g \cdot L^{-1}$; TDP-3 $\mu g \cdot L^{-1}$; SRP-1 $\mu g \cdot L^{-1}$; TN-5 $\mu g \cdot L^{-1}$; TDN-5 $\mu g \cdot L^{-1}$; NO_3^- + NO_2^--1 $\mu g \cdot L^{-1}$.TP data from Star Creek and TDN data from Smith Creek in the FL flow category of 2011 were censored as were SRP and TP in Star Creek and TDP in Smith Creek for the FL of 2012. Data collected during both the RL and FL of 2011 and 2012 did not differ for all water chemistry parameters using a series of Wilcoxon-signed rank tests, thus observations for the two years were combined to contrast the dominant flow seasons. Wilcoxon-signed rank tests were used to test for differences between the disturbed and reference watersheds for the various water chemistry parameters measured during the RL and FL flow category (α=0.05).

2.2.7.3 Primary production

Primary production data was combined by study year and failed to meet assumptions of normality according to Q-Q plots and Shapiro-Wilk tests at α=0.05. Thus Wilcoxon-signed rank tests were used to compare median AFDM and Chl*a* measures of primary production.

2.3 Results

2.3.1 Surface Water Hydrology

The flow regimes of the Star and Smith Creeks were largely similar over the duration of the study (Figure 2-7). The spring snowmelt freshet dominated the hydrologic regime of both watersheds. The highest discharges occurred in May and June (~ 5 - 15 $mm \cdot d^{-1}$). The timing of the spring melt freshet for both creeks was nearly identical in 2011 (Julian day 121) and 2012 (Julian day 108). The spring freshet progressed slightly more rapidly in Star as noted by comparatively higher peaks in the rising limbs and lower peaks in the falling limbs of 2011 and 2012. Spring and summer rainstorms produced similarly rapid responses in discharges in both streams, with steep rising limbs and rapid post-peak recessions.

Most significant discharge events were closely synchronized between the two creeks with the exception of two spring melt events in 2011 (Julian days 150-153 and 158-161) and the latter part of the falling limb of 2011 (Julian days 200-230) (Figure 2-7). A rain-on-snow precipitation event in Star Creek occurred as snow in the Smith Creek watershed, causing the large discrepancy in discharges between Julian days 158 and 161 in 2011. A temporarily malfunctioning pressure transducer contributed to the erratic discharge readings seen in the 2011 falling limb of Star Creek. Late summer and overwinter base flows were consistently between 0.5 and 2 $mm \cdot d^{-1}$. Cumulative area-weighted discharges were also similar for the two study basins, particularly in the 2012 study period (Table 2-2). In 2011, the Star Creek basin produced 11%, 18%, and 15% greater total Q than Smith Creek in the rising limb, falling limb, and total seasonal discharge (up to Julian day 245), respectively. During the 2012 study period the two basins produced nearly identical quantities of water of ~ 571 mm over 285 days (Table 2-2), with only minor variation among flow periods. The average daily discharge during the 2011 study period was 2.48 $mm \cdot d^{-1}$ and 2.12 $mm \cdot d^{-1}$ for Star and Smith Creeks, respectively. In 2012 the average daily discharges were slightly lower at 2.00 $mm \cdot d^{-1}$ for both Star and Smith Creeks.

2.3.2 Suspended Solids

2.3.2.1 Physical Properties

Slightly greater median daily sediment concentrations and turbidity were generally observed in Smith Creek compared to Star Creek over the two study seasons, though consistent differences in TSS were not evident across flow seasons of the two study years (Table 2-3 and Table 2-4). Median TSS concentration over the two year study period was 13% greater in Smith Creek than in Star Creek (p=0.027, Table 2-3). While TSS was only slightly greater during individual years' RL and FL flow periods in Smith Creek, these differences were significant only during the FL of 2012 (Figure 2-8, Table 2-3). Additionally, the large rain on snow event during spring 2011 (RL) in Star Creek (which did not occur in Smith Creek) was associated with median TSS concentration of 4.66 $mg \cdot L^{-1}$ compared to 2.33 $mg \cdot L^{-1}$ observed in Smith Creek over the RL flow period of 2011 (p=0.003, Table 2-3). Differences in turbidity among seasons and catchments were more consistent than observed for TSS. Median turbidity in Star Creek was 0.36 NTU compared to 0.55 NTU observed in Smith Creek (53% greater) over the two year study period ($p<0.001$, Table 2-4, Figure 2-9). Median turbidity in Smith Creek was also consistently greater across all flow periods in 2011 and 2012 ($p<0.001$ to p=0.005) though the 11% difference evident during the FL of 2011 was not significant (Table 2-4). Despite the generally greater median TSS and turbidity in Smith Creek compared to Star Creek, Log transformed Q-TSS relationships did not indicate strong differences in the sediment/streamflow regime of the two watersheds. While the relationships revealed highly variable, but significant ($p<0.05$) positive relationships between discharge and TSS in both watersheds in 2011 and 2012 (Figure 2-10), this relationship was not significant at α=0.05 when considered over the entire two year study period.

2.3.2.2 Chemistry of Suspended Solids

The concentration of all particulate P species was generally greater in suspended sediment collected from Smith Creek than from Star Creek in both years of study (Figure 2-11). NAIP was the most variable particulate P form among the catchments and it was 36-39% greater in Smith Creek than in Star Creek in both study years (p=0.033 in 2011 and p=0.037 in 2012). No difference in AP and OP concentrations were evident among watersheds in either year (Table 2-5). In addition to differences in NAIP among watersheds in 2012, TPP concentration of suspended solids from Smith Creek (746 $\mu g \cdot g^{-1}$ sediment dwt) was 19% greater (p=0.046) than Star Creek (636 $\mu g \cdot g^{-1}$ sediment dwt, Table 2-5). The proportion of NAIP, AP and OP species was approximately 26%, 55%, and 19% of total PP, respectively for both watersheds across the two study years. These proportions were very similar for both the disturbed and reference catchments in 2011 and 2012 (Table 2-5).

There was considerable spatial variation in the concentration of particulate P forms in suspended sediments from both watersheds. The greatest concentrations of all P fractions where observed in upper most stream reaches of both watersheds and these steadily decreased in the lower reaches of both streams (Figure 2-12). Linear relationships between NAIP, AP and TPP with relative reach location (top to bottom) were significant (p<0.05), while the decreases in OP were not significant. Tests for overall coincidental regressions showed differential linear relationships for TPP, NAIP and AP with relative reach location between Star and Smith Creeks. Suspended sediments from Smith Creek showed higher TPP and NAIP concentrations than Star Creek across all relative reach locations (regression intercepts differed at α=0.05) but the slope of the relationships were similar. Differences in slopes and intercepts of these relationships were evident for AP (Figure 2-12).

2.3.3 Water Chemistry

Strongly differential patterns of P and N water chemistry were observed between Smith and Star Creeks over the two study seasons. Smaller relative differences between watersheds were observed in dissolved P than were evident for the particulate P forms. While median total P (TP), total dissolved P (TDP), and soluble reactive P (SRP) concentrations were generally similar or slightly greater in Smith Creek compared to Star Creek, only SRP was greater during the rising limb (RL) flow period (p=0.005) while only TP was greater during the falling limb (FL) flow period (p<0.001, Figure 2-13, Table 2-6). Distributions of TDP concentrations did not differ among watersheds for either flow category, nor the entire study period. In contrast to phosphorus, large differences in concentration of the various N forms were observed. All N forms were consistently higher in Star Creek across all flow categories (Figure 2-14). In Star Creek, median TN concentrations were 1.75-2 times higher than in Smith Creek. Greater differences in median concentrations of TDN and $NO_3^- + NO_2^-$ than those observed in TN were evident among watersheds. Median TDN and $NO_3^- + NO_2^-$ in Star Creek were, 1.89-2.37 and 2.63-3 times greater than in Smith Creek over the two study years, respectively. Consistent with this observation, all N species were detected at higher concentrations in Star Creek than in Smith Creek during the rising limb flow category (Table 2-6).

2.3.4 Primary Production

Algal productivity was much greater in Smith Creek than in Star Creek over the two year study period. Median daily Chl*a* growth rates of 6.8 and 28.27 $\mu g \cdot cm^{-2} \cdot day^{-1}$ were observed in Smith Creek in 2011 and 2012, respectively compared to 0.03 and 2.44 $\mu g \cdot cm^{-2} \cdot day^{-1}$ over the same time period in Star Creek ($p<0.001$, Figure 2-15, Table 2-7). Similar, but smaller relative differences in ash-free dry mass of algal growth were observed between watersheds. Ash-free dry mass was produced in Smith Creek in 2011 and 2012 at 2.70 and 5.91 $\mu g \cdot cm^{-2} \cdot day^{-1}$, respectively, compared to 1.07 and 1.56 $\mu g \cdot cm^{-2} \cdot day^{-1}$during the same period in Star Creek ($p=0.03$ in 2011 and $p<0.001$ in 2012). Stream primary productivity, measured both as ash-free dry mass and chlorophyll *a* produced by benthic periphyton per unit area per day, was considerably higher in Smith Creek in both 2011 and 2012 (Figure 2-15). Smith Creek produced 2.5 and 3.8 times more periphyton AFDM than Star Creek in 2011 and 2012, respectively, and produced 239 and 11.6 times more Chl*a* in 2011 and 2012, respectively. Additionally, anecdotal evidence of higher benthic periphyton growth in Smith Creek was visually observed, as depicted in Figure 2-16.

2.4 Discussion

2.4.1 Physical Hydrology

The flow regimes of Star and Smith Creeks (unit area discharge volume and timing) were very similar. Observed differences in discharge between the two catchments were mostly produced by a large snowmelt/rain on snow event that occurred in Star Creek in May of 2011 (did not occur in Smith Creek) and the related differences in the falling limb of the annual hydrograph and base flow conditions of 2011. These minor differences were likely driven by slight elevation difference between the two catchments (Smith Creek is approximately 200 m higher in mean basin elevation). The greater elevation of Smith Creek might have contributed to producing slightly more snow in that catchment during a large precipitation event in the region in May 2011 than in the Star Creek watershed. At the time, a significant snowpack remained in the upper portions of both basins, and rain in Star Creek produced a discharge event that was unmatched in volume for the duration of the study. In Smith Creek, the event was comparatively muted as snow fell on the snow-covered upper slopes of the watershed. The differences in the falling limb and base flow discharges in 2011 are likely related, at least in part, to a malfunctioning pressure transducer in Star Creek. In this case, river stage and discharge were likely overestimated despite careful post-hoc data correction. Discharges were nearly identical between the two catchments in 2012 (Table 2-2).

Because these catchments are relatively close to each other, and situated in very similar hydro-climatic settings (topography, elevation, etc.), the river flow characteristics were expected to be similar. Although there are differences in forest age structure between the two catchments, water losses due to evapotranspiration (ET) are likely similar. Harvesting is known to temporarily increase water yield (Stednick, 1996), but lodgepole pine stands in the eastern slopes of Alberta reach peak leaf area index (LAI) and volume growth (major drivers of transpiration) at approximately 25 years of age (Brabender, 2005), younger than most of the Smith Creek watershed. Additionally, watersheds in the

Rocky Mountain regions are typically less sensitive to changes in ET, in terms of water yield response, than wetter hydrologic regions whose water balances are more ET dominated (i.e., Coastal regions, Eastern NA) (Stednick, 1996). If any difference in ET rates between the two catchments exists, it likely has limited influence on water yield because of the comparatively minor control ET has on Rocky Mountain, snowmelt-dominated watersheds. The two catchments have very similar physiographic settings that largely contribute to overall similarity in their hydrologic regimes. Basin slope distributions, stream slopes and aspect, which impact snowmelt timing and rates as well as response to precipitation events are very similar in the two study watersheds. Additionally, the two watersheds share the same general elevation range which strongly influences precipitation and snow accumulation, and further contribute to the similarities observed in surface water hydrology.

2.4.2 Suspended Solids

The results showing higher overall TSS concentrations in Smith Creek are consistent with expectations for a watershed that has undergone such extensive forest harvest disturbance. Studies of sediment production following logging operations in similar hydroclimatic settings in Alberta, British Columbia, and the Northern Rockies of the USA consistently report higher TSS concentrations during and following forest harvest operations (Anderson and Potts, 1987; Nip, 1991; Macdonald *et al*., 2003). The severity and longevity of elevated suspended solids concentrations in harvested watersheds are dependent on multiple factors including the extent of forest harvest disturbance, road and skid-trail density, and condition of stream crossings (Macdonald *et al*., 2003). The persistence of increased sediment concentrations after harvest may depend on forest road use, recovery of vegetation (hydrologic recovery), as well as in-stream storage processes (Macdonald *et al*., 2003). Although some studies have reported recovery of TSS concentrations to pre-disturbance levels within 5 years (Anderson and Potts, 1987), fine sediment has been known to be stored in and behind woody debris dams, within the streambed and in other depositional areas for over 50 years (Gomi *et al*., 2002). Accordingly, this creates a lag in sediment delivery and provides a potential legacy source of fine sediment that can be remobilized during high magnitude flow events (Benda and Dunne, 1997). Thus, the relatively minor increases in suspended solids detected in Smith Creek would likely have been much more pronounced in the decades immediately after harvesting. Although no currently-contributing point source of sediment was evident along Smith Creek, it is plausible that significant amounts of sediment entered the watercourse at two road crossing locations close to the time of active forest harvesting. Swank *et al*. (2001) found that sediment introduced by road crossing erosion was continually released from in-stream storage, resulting in elevated sediment yields for greater than a decade in the absence of significant additional sediment sources. Such a lag in sediment delivery downstream is most likely the cause of the elevated TSS concentrations observed in Smith Creek in this study. Differences in TSS among watersheds were consistent across all flow categories, with the exception of the rising limb of 2011, when Star Creek had significantly higher (2 times) TSS concentrations. While this is contrary to expectations for an undisturbed catchment, this observation is explained in large part, by the rain-on snow precipitation event that occurred in May of 2011 in Star Creek and not Smith Creek (described above). This storm produced a high-flow event which increased sediment yield via surface overland flow and bank scouring/failure in some cases.

Consistent with observations for TSS, turbidity was significantly higher in Smith Creek than in Star Creek over the two study years with the exception of the falling limb of 2011 where no significant difference was detected between watersheds. However, median turbidity was consistently (monotonically) greater than that of Star Creek across all flow periods over the two study years (Table 2-4). The overall differences in sediment production (both TSS and turbidity) among watersheds were not driven by any fundamental differences in geology (Figure 2-2), watershed or stream topography (Figures 2-3 and 2-4) or flow regime between Star and Smith Creeks (Figure 2-11), which supports the likelihood that these differences were produced by the strongly contrasting forest disturbance history between watersheds. While few studies have tracked the long-term production of sediment (TSS and turbidity) after harvesting, one long-term study of TSS and turbidity after harvesting in western Washington (Reiter *et al*., 2009) reported persistence of elevated TSS and turbidity in several watersheds of 30-40 years after road building and harvesting. In this case, watersheds were harvested with less watershed area disturbed (including application of erosion control measures on roads) than occurred in Smith Creek (this study). While differences in TSS and turbidity between harvested and un-harvested watersheds in the present study varied slightly (TSS was not consistently greater in the harvested watershed across two years, while turbidity was), TSS ($mg \cdot L^{-1}$) is heavily biased by infrequent transport of larger particles during high flows, while turbidity is more a reflection of mid-sized and smaller particles (Davies-Colley and Smith, 2001). Thus, the higher TSS concentrations evident in Star Creek (unharvested) during the RL of 2011 are likely more a reflection of high sediment transport during a large rain-on snow event, rather than a reflection of differences or similarities in the overall sediment regime between these two watersheds. The finding of greater median TSS during 3 of 4 flow categories and consistently higher turbidity in Smith Creek in all flow periods across two years supports this conclusion.

Suspended sediment-associated P concentrations were consistently higher in Smith Creek for TPP and NAIP in 2011, 2012 and over both study years collectively. There was no statistical difference between the AP and OP concentrations among watersheds, however Smith Creek had constantly higher mean AP and OP levels across both study years individually and over the two year study period. While the lack of pre-disturbance data for Smith Creek makes it more difficult to conclusively attribute the differences in sediment P to differences in disturbance history among watersheds, there is sufficient evidence to support the concept that the elevated sediment-P levels in Smith Creek were likely the result of the extensive past logging and road construction disturbance within that watershed. Both watersheds are underlain by the same three geologic formations and in similar proportions, thus the ultimate source of P in each catchment is fundamentally the same (Froelich, 1988). Furthermore, the proportions of NAIP, AP and OP which make up TPP are very consistent between the two watersheds (Table 2-6), suggesting that the biogeochemical processes governing P dynamics are likely very similar between the two catchments. Thus, the differences observed in sediment-P concentrations are likely strongly related to the differences in disturbance history, which is the most significant difference between the two catchments with the potential to affect P dynamics. The elevated levels of NAIP and TPP in Smith Creek are consistent with what is expected for a disturbed watershed, and is likely related to the persistence of greater turbidity and TSS concentrations evident in that watershed several decades after harvest disturbance. Fine sediment produced by historic logging/road disturbance is likely responsible for the higher sediment-associated P in Smith Creek. Indeed, while the post-harvest sediment production immediately after harvesting would be expected to be much greater than the contemporary sediment production observed in this study, because sediments can be stored in streambeds for many years or decades (Benda and Dunne, 1997), the greater TSS and turbidity observed in this study (several decades after harvest) is a highly probable reflection of the legacy of past sediment production and channel loading. The presence of greater fine sediments in Smith Creek remained strongly evident in higher turbidity (Table 2-4), and these fine sediments are known to play a disproportionately large role in the transport and storage of P via exchangeable sorption-desorption reactions (Wang *et al*., 2006). Thus, it is likely that the history of disturbance in Smith Creek has left a detectable legacy of greater deposits of fine sediment and subsequently, greater sediment associated P that has the potential to drive increased concentrations of aqueous P.

2.4.3 Water Chemistry

In contrast to sediment associated P, aqueous TP and SRP concentrations were only slightly higher in Smith Creek than in Star Creek. This was evident for SRP in the RL flow category, for TP in the FL flow category, and for both forms of phosphorus when considered over the course of the entire study. However, TDP had very similar distributions and identical median values in both flow categories and the overall study period in both Star and Smith Creeks. While the pattern of slightly higher total and bioavailable P in Smith Creek is consistent with differences in sediment associated P, the differences in sediment associated P among watersheds was much stronger than those evident for aqueous P forms. While aqueous TP and SRP concentrations are linked to sediment-associated P through reversible sorption-desorption processes (Froelich, 1988; Haggard *et al*., 2007), the finding of smaller relative differences among watersheds for aqueous SRP, TDP, and TP than was observed for sediment associated P is consistent with what might be expected if the primary source of catchment differences in aqueous P was produced by reversible sorption-desorption processes. This process is dependent on the stream equilibrium P concentration (EPC_0), which is the aqueous concentration of P at which net adsorption and desorption from benthic solids is equal to 0 (Froelich, 1988). The EPC_0 is influenced by numerous abiotic characteristics including benthic particle size, presence of aqueous divalent cations, and the strength at which P is adsorbed to sediments (Haggard *et al*., 2007). It is likely that the elevated aqueous P concentrations in Smith Creek resulted from P-rich fine sediment (initially introduced near the time of logging/road construction) remobilized from the streambed and marginal zones (House, 2003) undergoing desorption reactions once re-suspension occurs. Thus, it is not surprising that catchment differences in suspended sediment associated P concentrations were considerably stronger than those observed in aqueous P concentrations because of potential (likely) differences in EPC_0 and biotic uptake and cycling.

Star Creek had significantly and consistently higher concentrations of all forms of nitrogen (TN, TDN, NO_2^- + NO_3^-) across all years and flow periods. Nitrogen concentrations in surface waters are strongly related to the rate of nitrogen uptake from terrestrial vegetation (Likens *et al*., 1978; Swank *et al*., 2001). Thus the finding of greater N in Star Creek (undisturbed) was potentially a reflection of differences in forest age structure between the two watersheds and their effect on N uptake of terrestrial vegetation and stream N concentrations. It has been theorized that following disturbance, young aggrading forests tightly retain N as a limiting nutrient, resulting in low stream N concentrations until later seral stages when N demand is reduced (Vitousek and Reiners, 1975; Cairns and Lajtha, 2005). However, several studies have failed to entirely support this concept, reporting N flux to be heavily influenced by in-stream nutrient dynamics, large woody debris recruitment, biotic uptake, abiotic sorption-desorption reactions and successional species turnover (Swank *et al*., 2001; Warren *et al*., 2007). The complex processes that determine N export from watersheds are not fully understood, but it is likely that in Smith Creek a combination of factors including a relatively young, rapidly growing forest, N adsorption to fine sediment (more abundant in Smith Creek), and much greater uptake by primary producers all may have played a role in the lower aqueous N concentrations observed in Smith Creek compared to Star Creek.

2.4.4 Primary Production

Smith Creek had substantially higher rates of Chl*a* and AFDM production overall and in both of the study years. The much greater algal productivity in Smith Creek was most likely driven by the higher concentration of bioavailable P (SRP) in the water column. While solar radiation and water temperature are additional environmental factors known to contribute to algal production, these factors were unlikely to be drivers of the increased productivity observed in Smith Creek. Stream water was consistently warmer and more accommodating to primary producers in Star Creek, particularly in the ice free periods. Water temperatures during lower base flows were very similar between watersheds. Furthermore, while solar radiation reaching the streams was not quantitatively measured, the light regimes in the study reaches of Smith Creek were similar or more shaded than that of Star Creek. Phosphorus (particularly SRP) is well known as a crucial growth-limiting nutrient regulating periphyton production in freshwater ecosystems (Schindler, 1977; Chessman *et al*., 1992; House, 2003). While nitrogen can act also as an important limiting nutrient (Schindler, 1977; Chessman *et al*., 1992), the relatively high N:P ratios and low productivity in Star Creek suggest that N was not a limiting nutrient in these streams. The ratio of NO_2^-+NO_3^- to SRP over the course of the study was 38 in Star Creek. Chessman *et al*. (1992) found that streams with reactive N to reactive P ratios of 17 and greater were nearly always phosphorus-limited. However, the very strong differences in algal productivity evident among watersheds were not likely driven solely by the relatively small differences evident in aqueous P forms between watersheds. Schindler (1977) found primary production increased proportionally with increases in P concentrations in freshwater systems where other potentially limiting factors, such as N and C, are in adequate supply. While overall median SRP concentrations in Smith Creek were 1.2 times greater than that of Star Creek, AFDM and Chl*a* production in Smith Creek was 2.8 and 12.1 times higher in Smith Creek, respectively. Thus, differences in algal productivity were unlikely to have been produced solely by the slightly elevated SRP concentrations in Smith Creek. While the difference in SRP concentrations among watersheds almost certainly influenced the differences in algal productivity among watersheds, the magnitude of differences observed in algal productivity suggests the presence of additional factors. It is possible that differences in P availability among watersheds may be related to another potential source of additional P associated with bed sediments and their potential role in P dynamics and desorption processes in these two watersheds.

2.5 Conclusion

The history of extensive logging and road-building disturbance in the Smith Creek watershed is the most likely cause of elevated primary productivity observed in Smith Creek. The hydrologic regimes of Star and Smith Creeks were largely similar and are unlikely to be key factors driving the large differences in algal growth observed in this study. Rather, higher TSS concentrations and turbidity in combination with higher sediment-associated P concentrations were the most likely contributing factors to the higher primary production evident Smith Creek. This study highlights the particular importance of fine sediments and sediment-nutrient interactions as an important component of forest disturbance impacts on Rocky Mountain headwater streams. Moreover, this study strongly supports the idea that sediment-nutrient enrichment leading to elevated primary productivity can persist for decades following logging disturbance and can represent an important, long-lived legacy of harvest disturbance in some regions.

Tables and Figures

Table 2-1. Watershed characteristics of Star and Smith Creeks.

		Watershed	
Basin Characteristic		Star Creek	Smith Creek
Location (UTM NAD 83 Zone 11N)			
	Northing	5498157	5519986
	Easting	676557	674685
Basin Area (ha)		1035	697
Basin Elevation (m)			
	Mean	1853	2016
	Min.	1482	1657
	Max.	2631	2768
Basin Slope (%)			
	Mean	44	43
	Min.	0	0
	Max.	1908	1143
Stream Slope (%)		13.6	11.3
General basin aspect		NE	E-NE

Table 2-2. Summary of annual unit-area streamflow (mm) by flow category in 2011 and 2012 for Star and Smith Creeks.

	2011		2012	
Component of annual hydrograph	Star Creek	Smith Creek	Star Creek	Smith Creek
Rising Limb (RL)	324.2	288.0	386.7	370.6
Falling Limb (FL)	283.0	231.0	184.3	200.7
Total	607.2	519.0	571.0	571.4

Table 2-3. Median total suspended solids concentration ($mg \cdot L^{-1}$) by year and flow category in Star and Smith Creeks. Probabilities reflect Wilcoxon-signed rank tests between watersheds.

	Stream		
Flow Period	Star Creek	Smith Creek	*p*-value
RL 2011	4.66	2.33	0.003
FL 2011	1.11	1.23	0.197
RL 2012	1.73	2.05	0.107
FL 2012	1.49	2.14	<0.001
Overall	1.65	1.86	0.027

Table 2-4. Median turbidity (NTU) by year and flow period in Star and Smith Creeks. Probabilities reflect Wilcoxon-signed rank tests between watersheds.

	Stream		
Flow Period	Star Creek	Smith Creek	*p*-value
RL 2011	0.53	0.90	0.001
FL 2011	0.37	0.41	0.284
RL 2012	0.41	0.56	0.005
FL 2012	0.30	0.61	<0.001
Overall	0.36	0.55	<0.001

Table 2-5. Mean annual suspended sediment concentrations of four particulate P species (μg·g^{-1} sediment dwt) in 2011, 2012 and the overall study period for Star and Smith Creeks. Values in brackets indicate proportion of each species (non-apatite inorganic P (NAIP), apatite P (AP), organic P (OP)) as a fraction of TPP (total particulate P), *p*-values reflect Wilcoxon-signed rank tests between watersheds.

		Stream		
Year	P species	Star Creek	Smith Creek	*p*-value
2011	NAIP	176.3 (0.27)	244.6 (0.31)	0.033
	AP	377.9 (0.59)	406.1 (0.52)	0.343
	OP	88.3 (0.14)	127.3 (0.16)	0.097
	TPP	642.6 (1.00)	778.0 (1.00)	0.111
2012	NAIP	136.1 (0.21)	185.5 (0.25)	0.037
	AP	342.9 (0.54)	398.1 (0.53)	0.103
	OP	156.7 (0.25)	162.4 (0.22)	0.319
	TPP	635.8 (1.00)	746.0 (1.00)	0.046
Overall	NAIP	156.2 (0.24)	215.0 (0.28)	0.006
	AP	360.4 (0.56)	402.1 (0.53)	0.145
	OP	122.5 (0.19)	144.9 (0.19)	0.091
	TPP	639.2 (1.00)	762.2 (1.00)	0.022

Table 2-6. Median stream water concentrations of P (soluble reactive P (SRP), total dissolved P (TDP), total P (TP)) and N (nitrate+nitrite (NO_2^-+NO_3^-), total dissolved N (TDN), total N (TN))species (µg·L^{-1}) by flow category across 2011 and 2012 in Star and Smith Creeks. Probabilities reflect Wilcoxon-signed rank tests between watersheds.

		Stream		
Component of annual hydrograph	Parameter	Star Creek	Smith Creek	*p*-value
Rising Limb (RL)	SRP	3.0	4.0	0.005
	TDP	3.0	3.0	0.721
	TP	5.5	7.0	0.334
	NO_2^-+NO_3^-	124.5	43.0	<0.001
	TDN	192.5	103.0	<0.001
	TN	194.0	111.0	<0.001
Falling Limb (FL)	SRP	3.0	3.0	0.168
	TDP	3.0	3.0	0.981
	TP	3.0	5.0	<0.001
	NO_2^-+NO_3^-	100.0	38.0	<0.001
	TDN	142.0	60.0	<0.001
	TN	140.0	70.0	<0.001
Overall	SRP	3.0	3.5	0.004
	TDP	3.0	3.0	0.757
	TP	4.0	5.0	0.009
	NO_2^-+NO_3^-	114.0	38.0	<0.001
	TDN	159.0	82.0	<0.001
	TN	161.0	82.5	<0.001

Table 2-7. Median periphyton growth ($\mu g \cdot cm^{-2} \cdot day^{-1}$) represented by ash-free dry mass (AFDM) and chlorophyll *a* (Chl*a*) in 2011, 2012 and the overall study period for Smith and Star Creeks. Probabilities reflect Wilcoxon-signed rank tests between watersheds.

		Stream		
Year	Parameter	Star Creek	Smith Creek	*p*-value
2011	AFDM	1.07	2.70	0.03
	Chl*a*	0.03	6.80	<0.001
2012	AFDM	1.56	5.91	<0.001
	Chl*a*	2.44	28.27	<0.001
Overall	AFDM	1.33	3.76	<0.001
	Chl*a*	1.18	14.24	<0.001

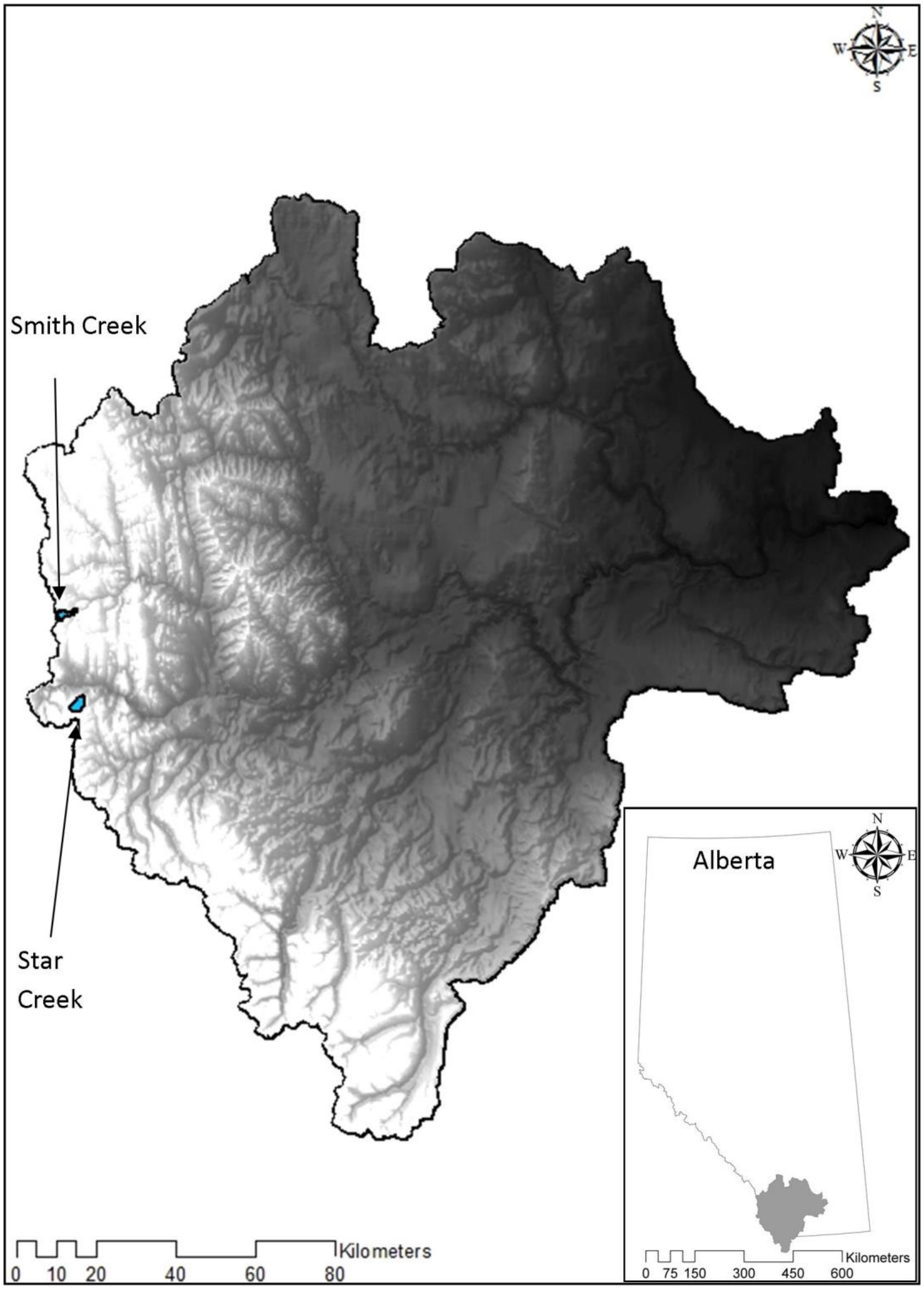

Figure 2-1. Oldman River watershed (shading from dark to light corresponds with low to high elevations, respectively), showing locations of Smith and Star Creek study watersheds. Inset shows location of Oldman River watershed in Alberta.

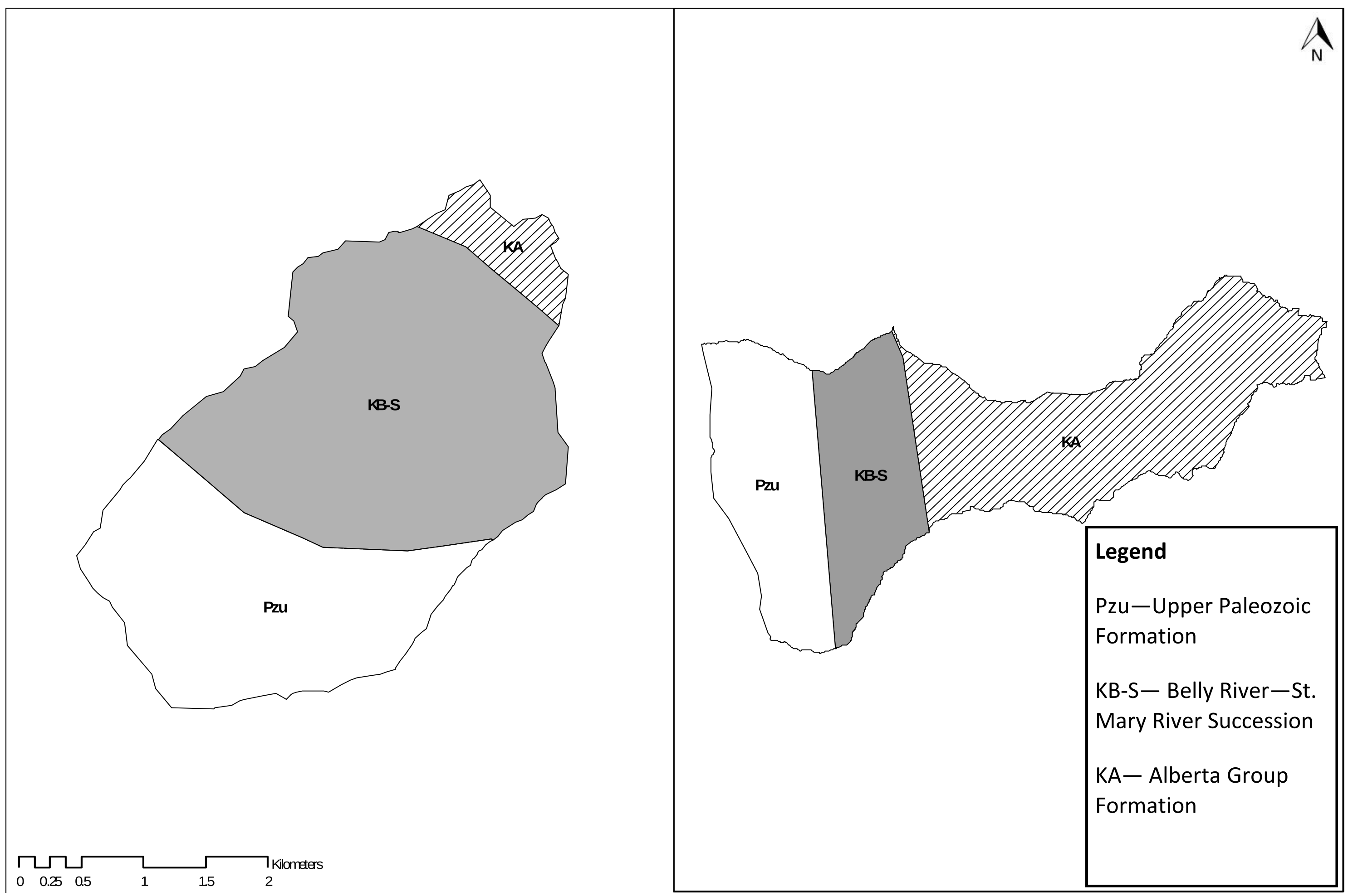

Figure 2-2. Maps of underlying geologic formations in Star and Smith Creek watersheds.

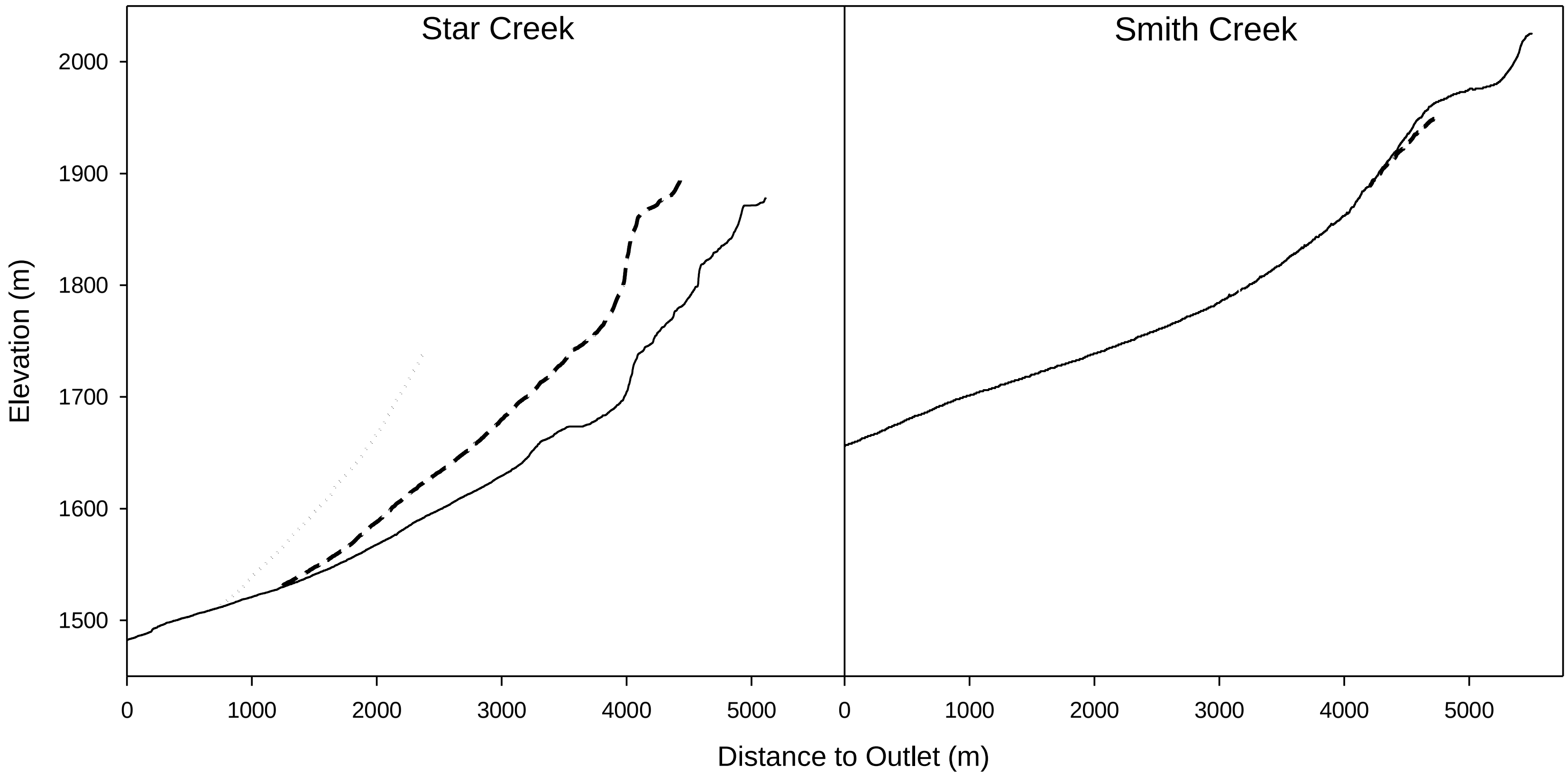

Figure 2-3. Stream elevation profiles for Star and Smith Creeks. Solid line indicates main channel, dashed and dotted lines are tributaries.

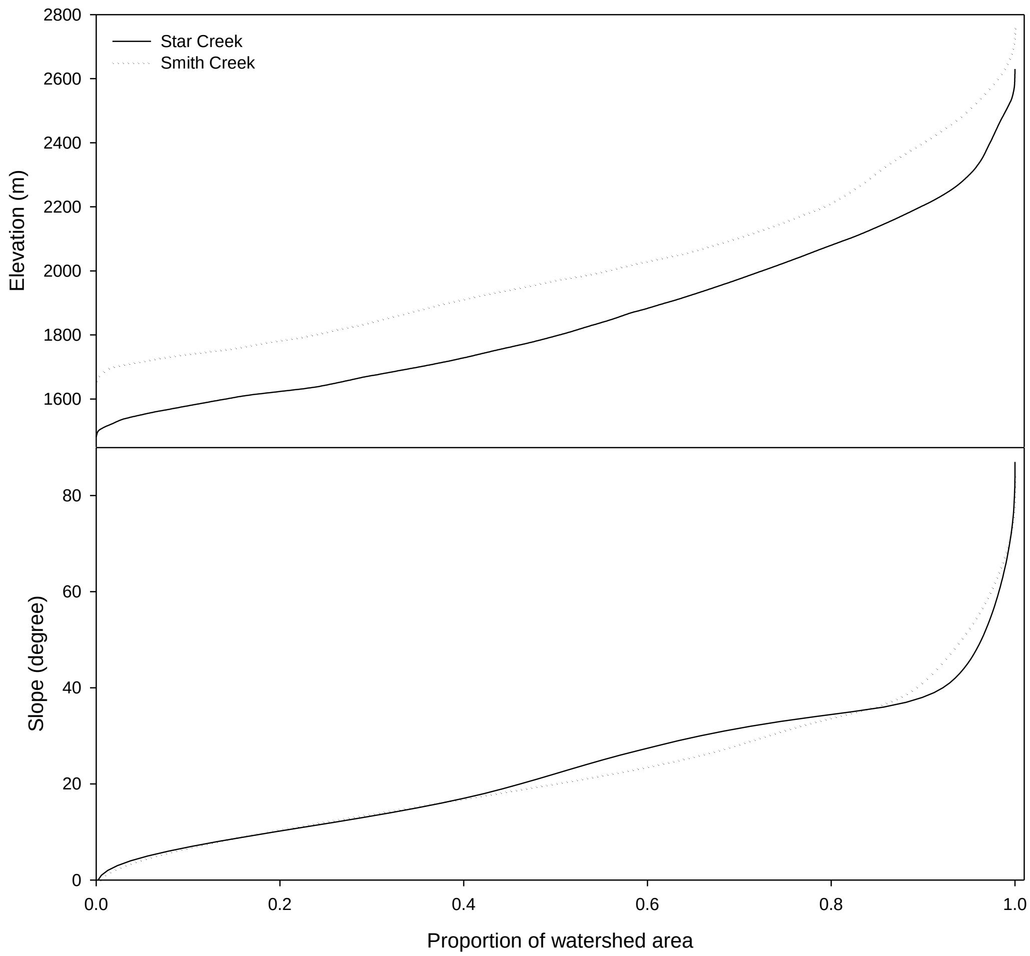

Figure 2-4. Elevation and slope distribution of Star and Smith Creek watersheds.

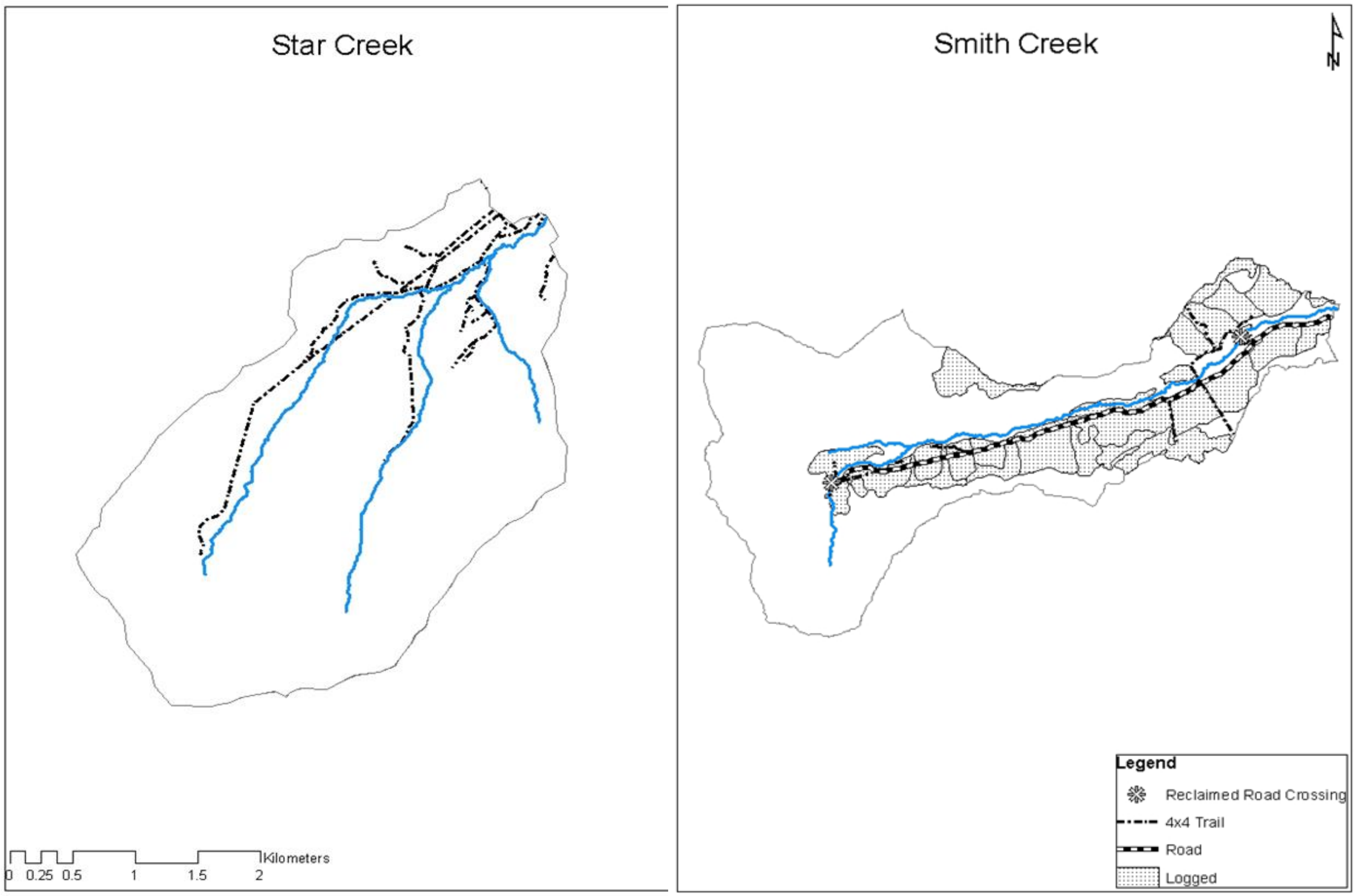

Figure 2-5. Map of historic disturbance in Star and Smith Creek watersheds.

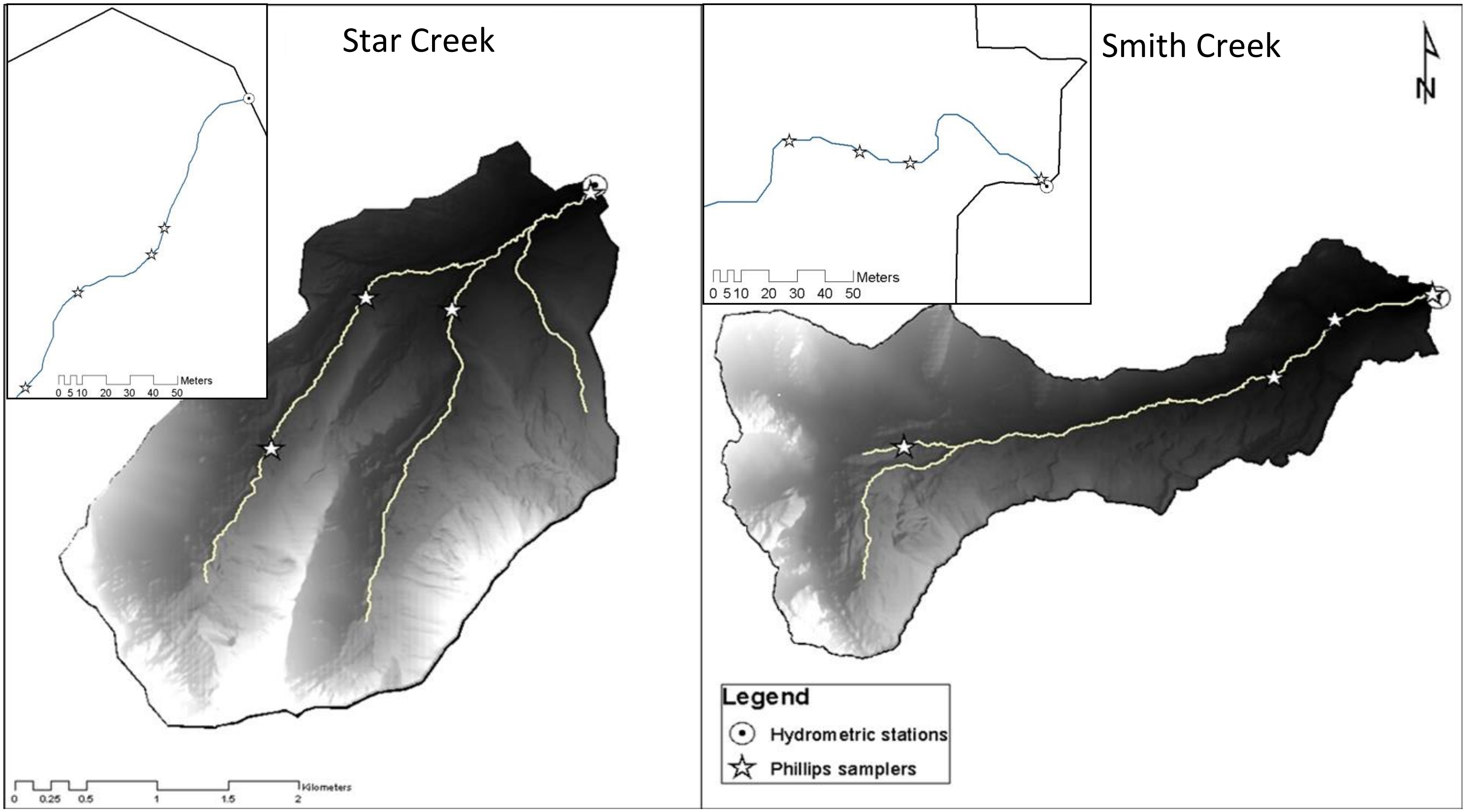

Figure 2-6. Elevation distribution in Star and Smith Creek watersheds, showing locations of hydrometric stations and Phillips' samplers. Shading from dark to light corresponds with low to high locations, respectively. Insets show locations of Phillips samplers clustered near outlets in 2012.

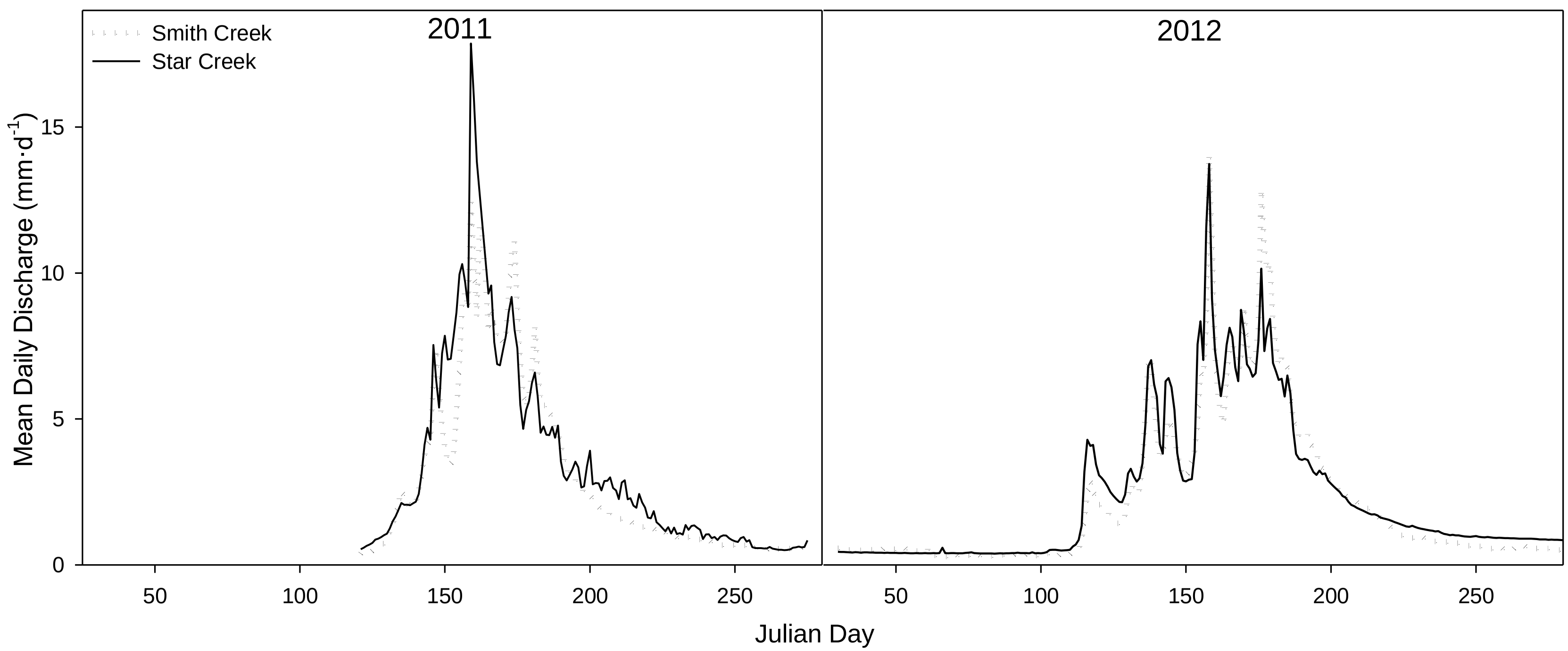

Figure 2-7. Annual hydrographs of Star and Smith Creeks for 2011 and 2012.

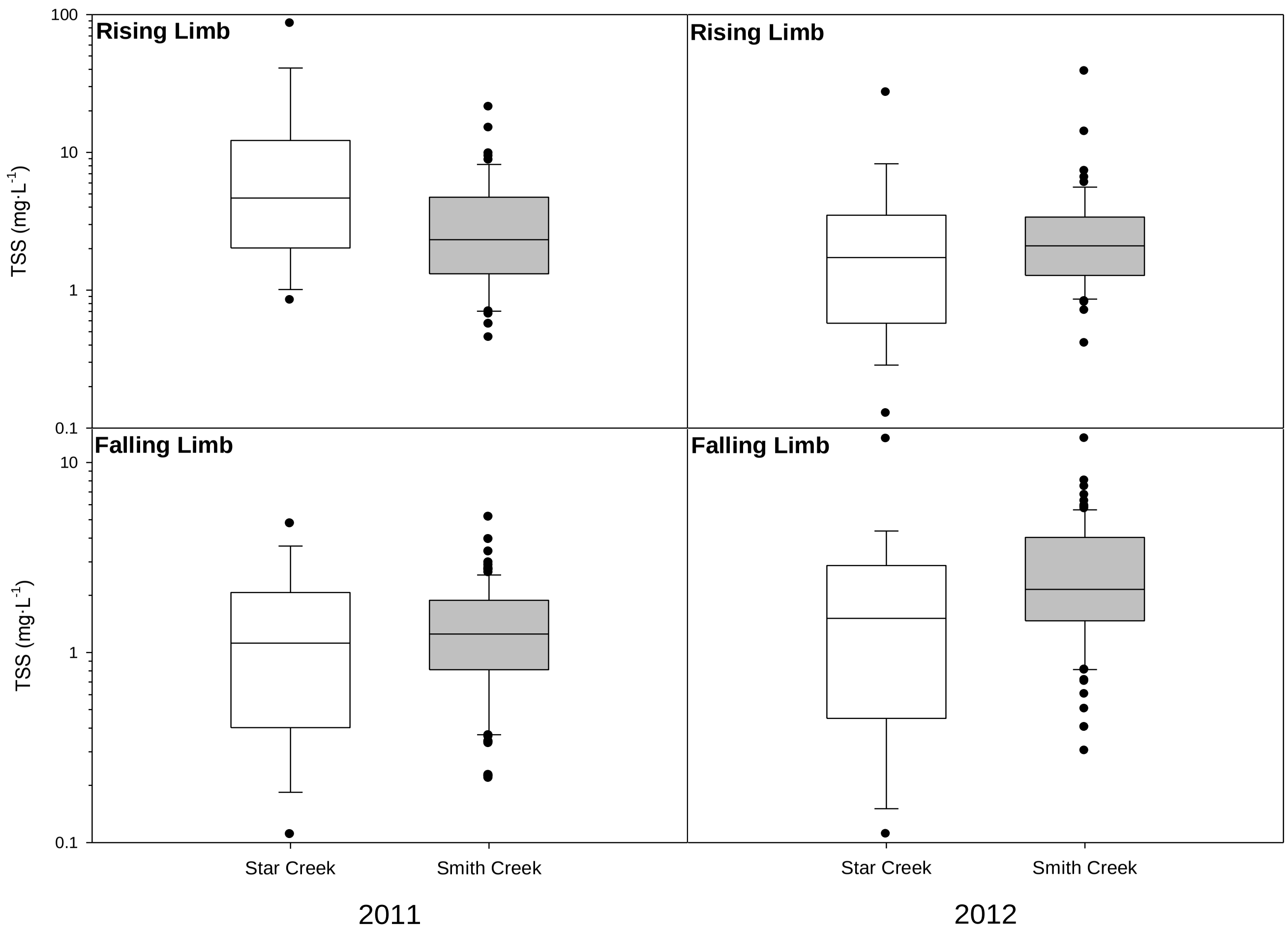

Figure 2-8. Censored distributions of daily total suspended solid concentrations in Star and Smith Creeks by flow category for both years of the study. Horizontal line in boxplots indicates median, upper and lower box position indicates 75th and 25th percentiles, whiskers indicate 95th and 5th percentiles, solid dots indicate outliers.

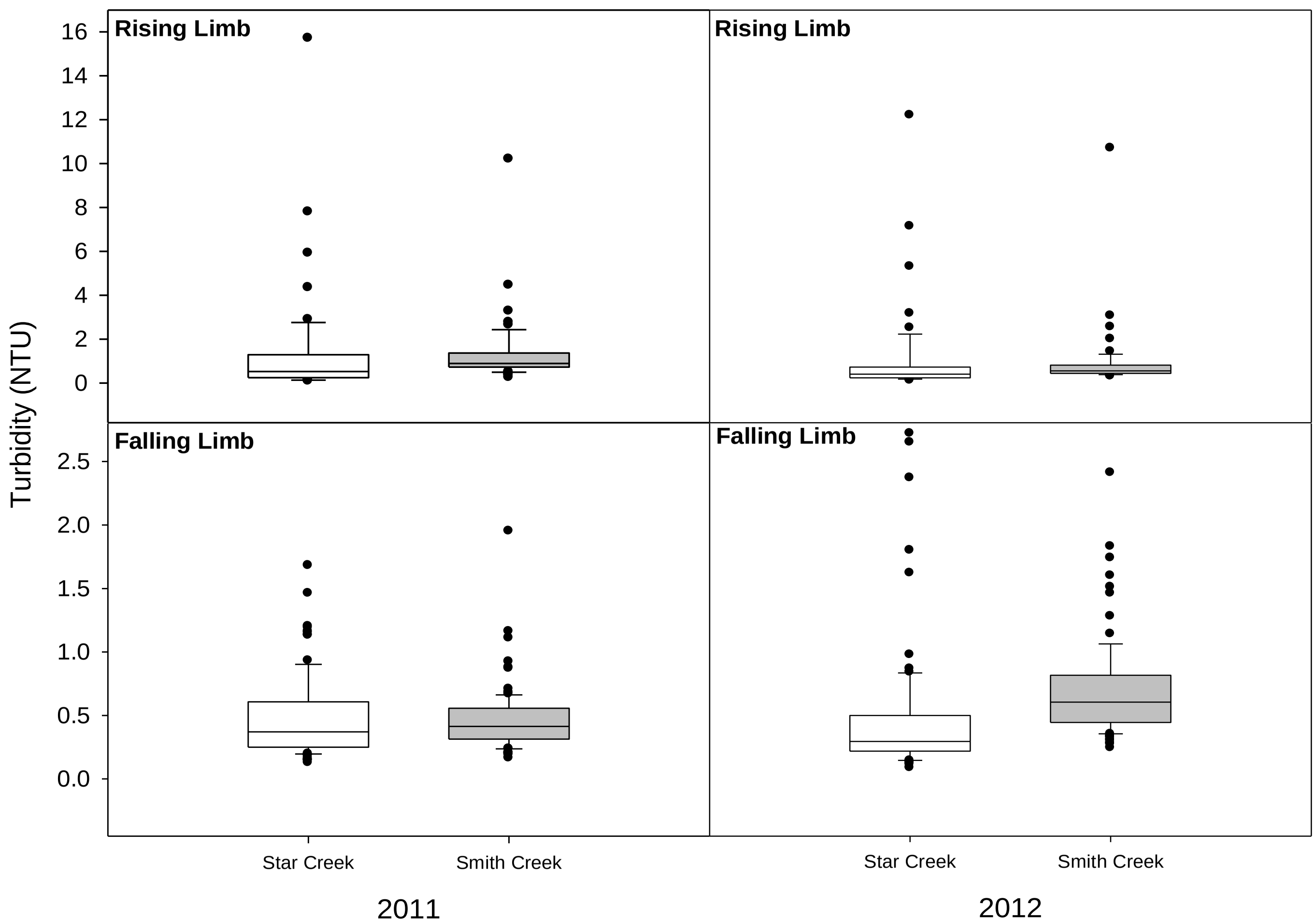

Figure 2-9. Distribution of mean daily turbidity by flow category, for both years of the study in Star and Smith Creeks. Horizontal line in boxplots indicates median, upper and lower box position indicates 75th and 25th percentiles, whiskers indicate 95th and 5th percentiles, solid dots indicate outliers.

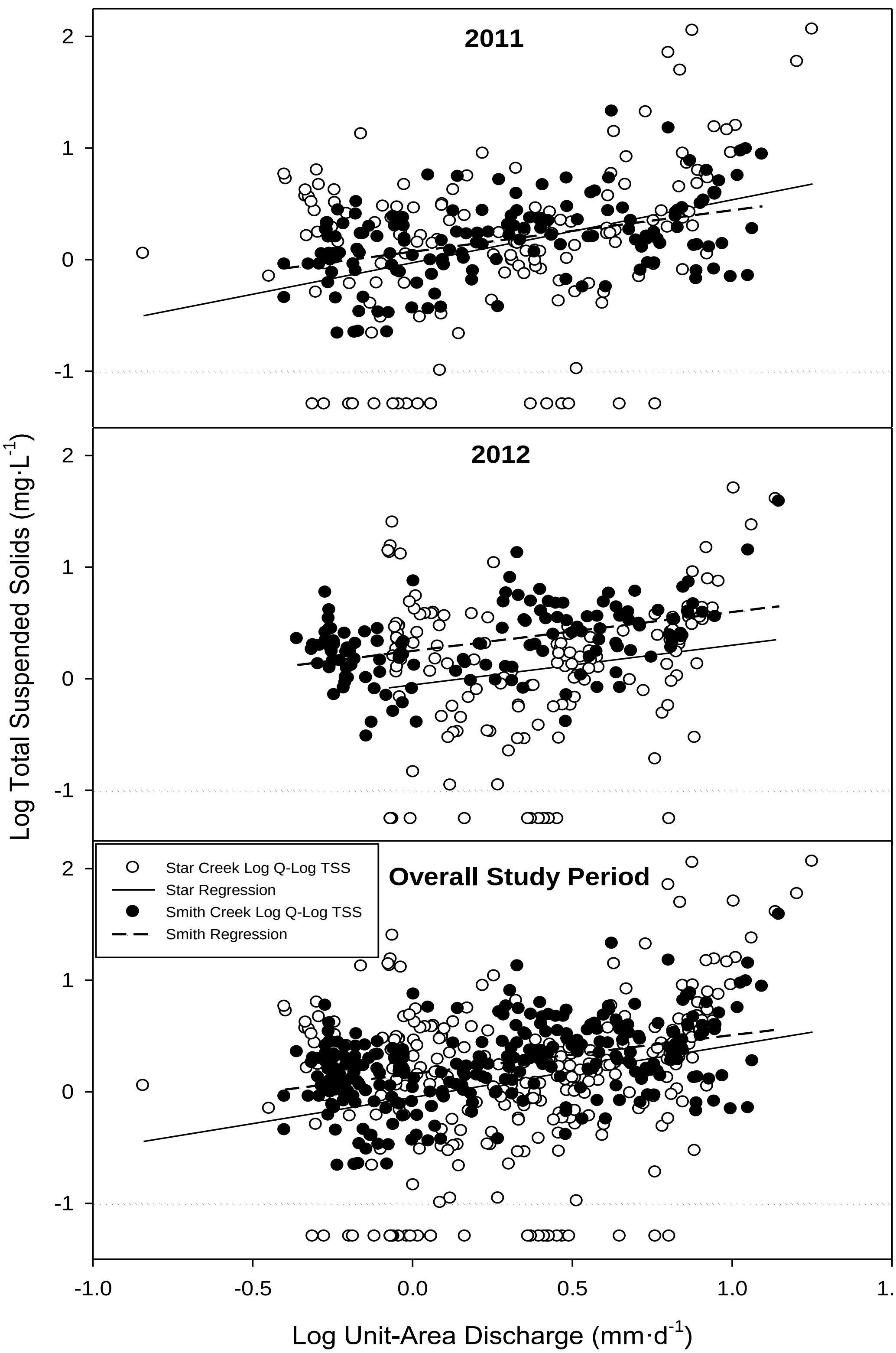

Figure 2-10. Log TSS – Log unit-area discharge relationships for 2011, 2012 and the overall study period for Star and Smith Creeks. Horizontal dotted line indicates limit of detection (LOD =0.1 mg/L). Data below line was estimated using ½ of LOD.

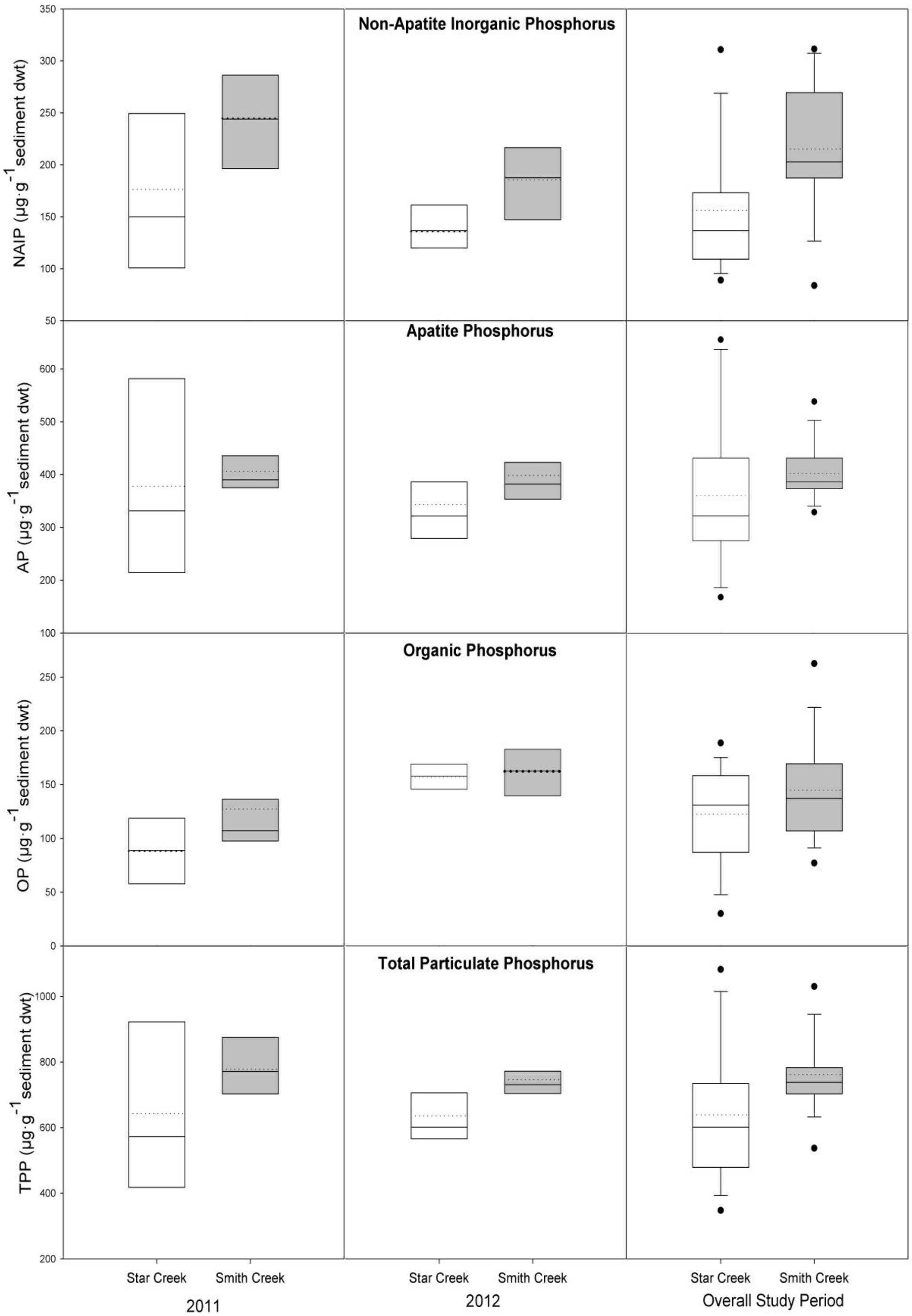

Figure 2-11. Distribution of suspended sediment P concentrations for all P species, year, and overall study period in Star and Smith Creeks. Horizontal line in boxplots indicates median, dashed line indicates mean, upper and lower box position indicates 75th and 25th percentiles, whiskers indicate 95th and 5th percentiles, solid dots indicate outliers.

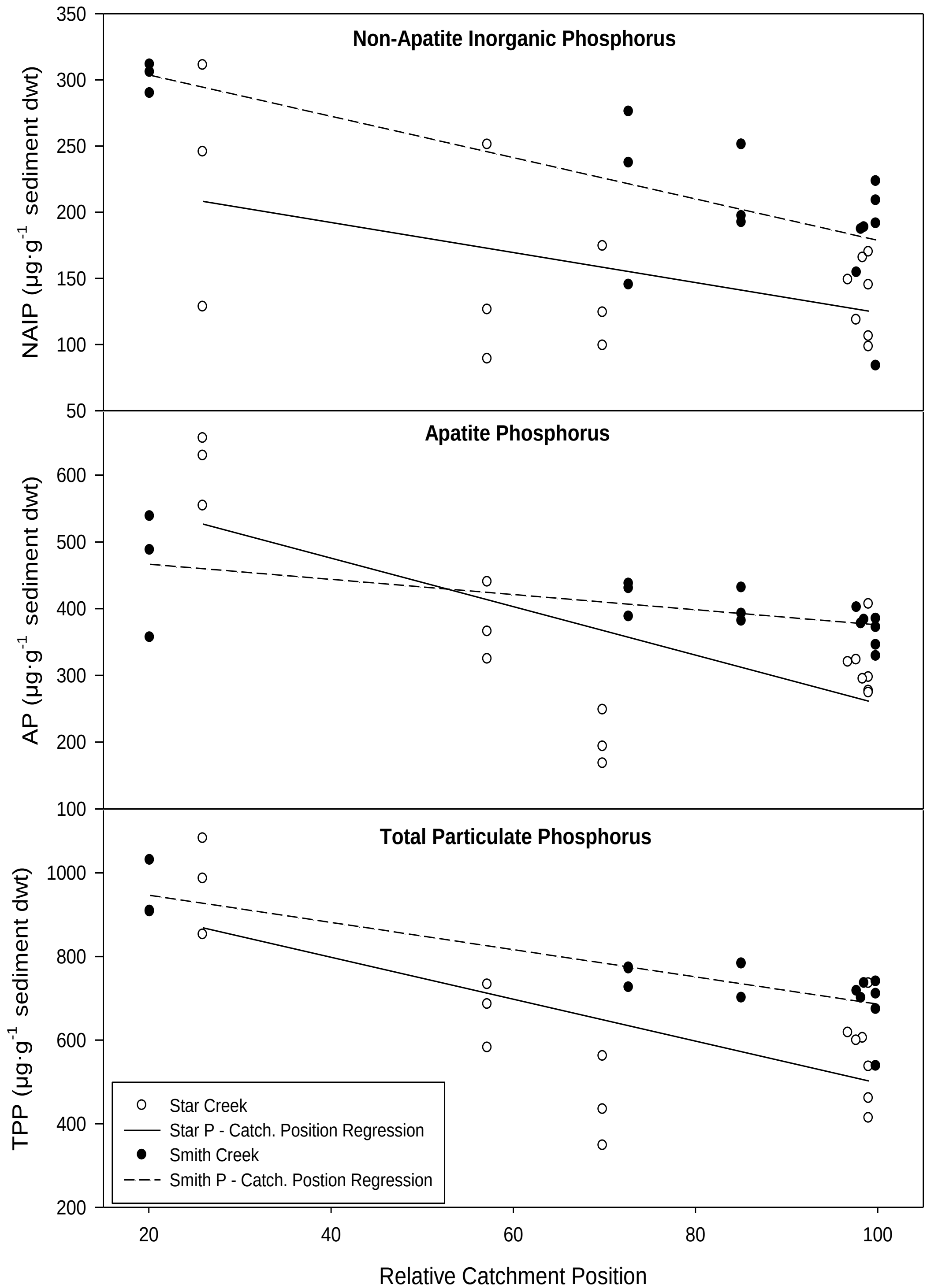

Figure 2-12. Relationship between relative reach position within catchments and concentration of sediment P species. Zero and 100 indicate most upstream and downstream reach positions, respectively.

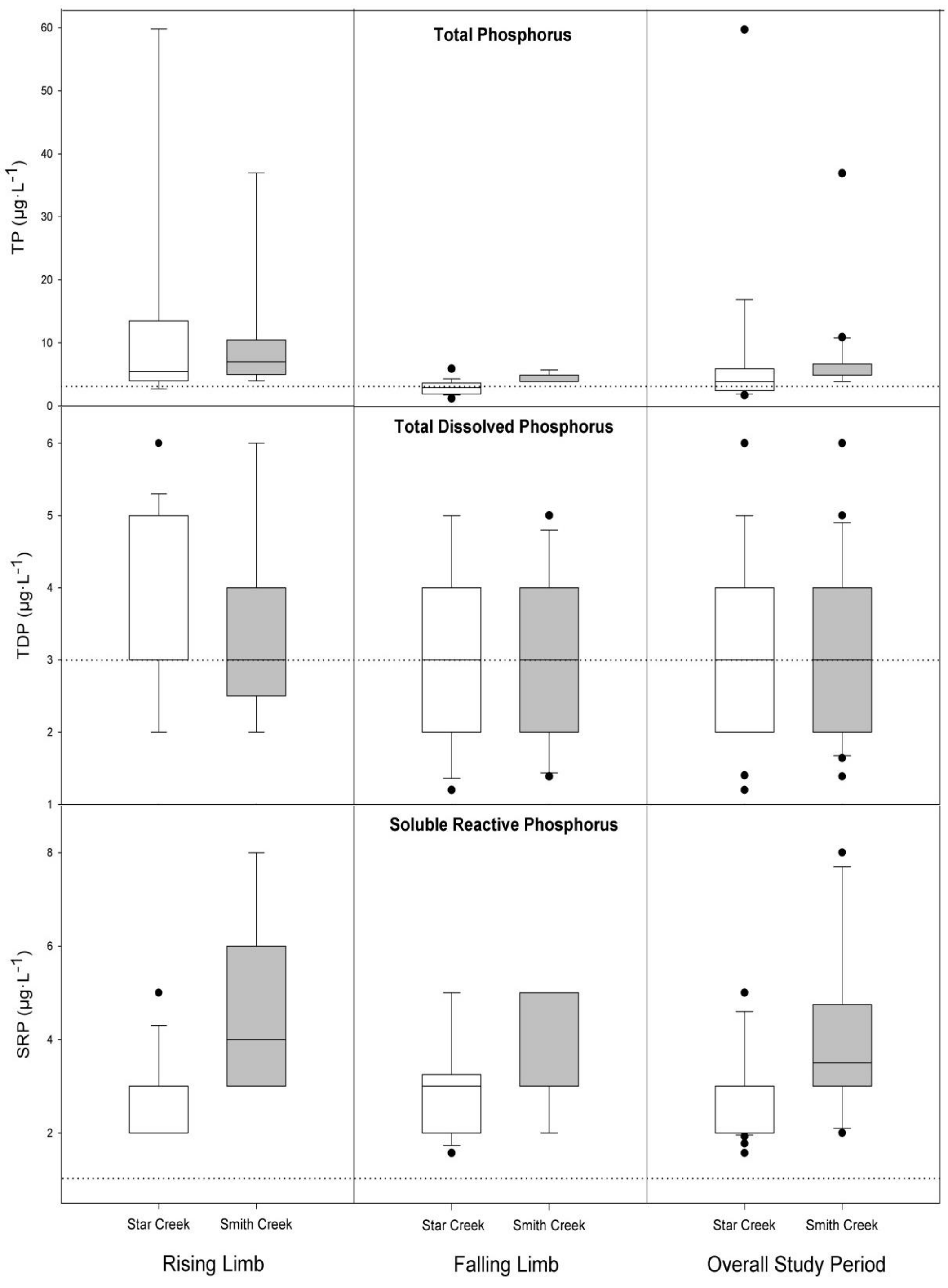

Figure 2-13. Censored distributions of phosphorus stream water species concentrations by flow category and overall study period in Star and Smith Creeks. Horizontal line in boxplots indicates median, dashed line indicates mean, upper and lower box position indicates 75th and 25th percentiles, whiskers indicate 95th and 5th percentiles, solid dots indicate outliers. Horizontal dotted lines indicate limits of detection (LOD); data below this line are estimates generated using cumulative distribution functions. LOD for TP, TDP is 3 $\mu g \cdot L^{-1}$. LOD for SRP is 1 $\mu g \cdot L^{-1}$.

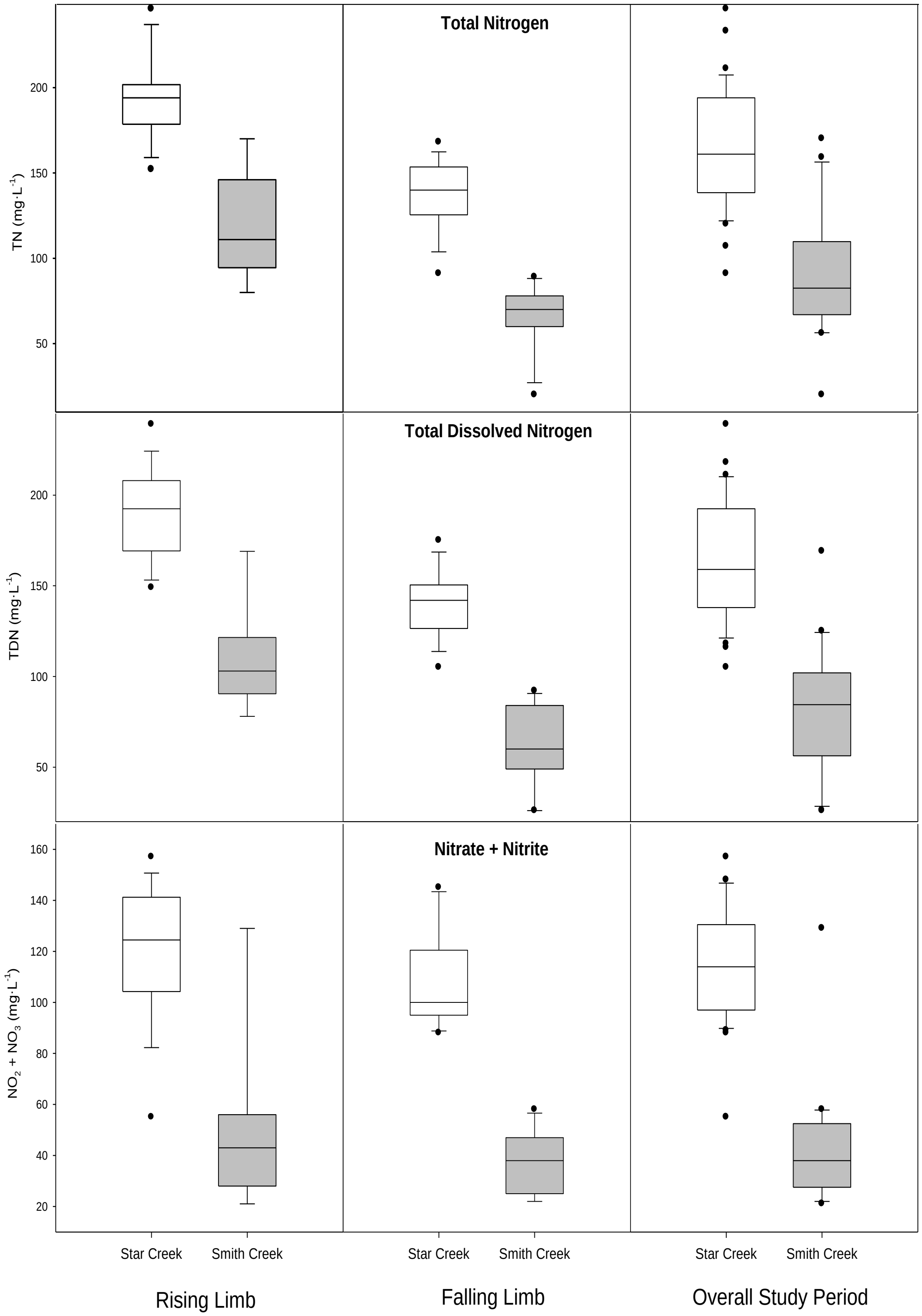

Figure 2-14. Distribution of concentrations of three aqueous nitrogen forms by flow category and overall study period in Star and Smith Creeks. Horizontal line in boxplots indicates median, dashed line indicates mean, upper and lower box position indicates 75th and 25th percentiles, whiskers indicate 95th and 5th percentiles, solid dots indicate outliers.

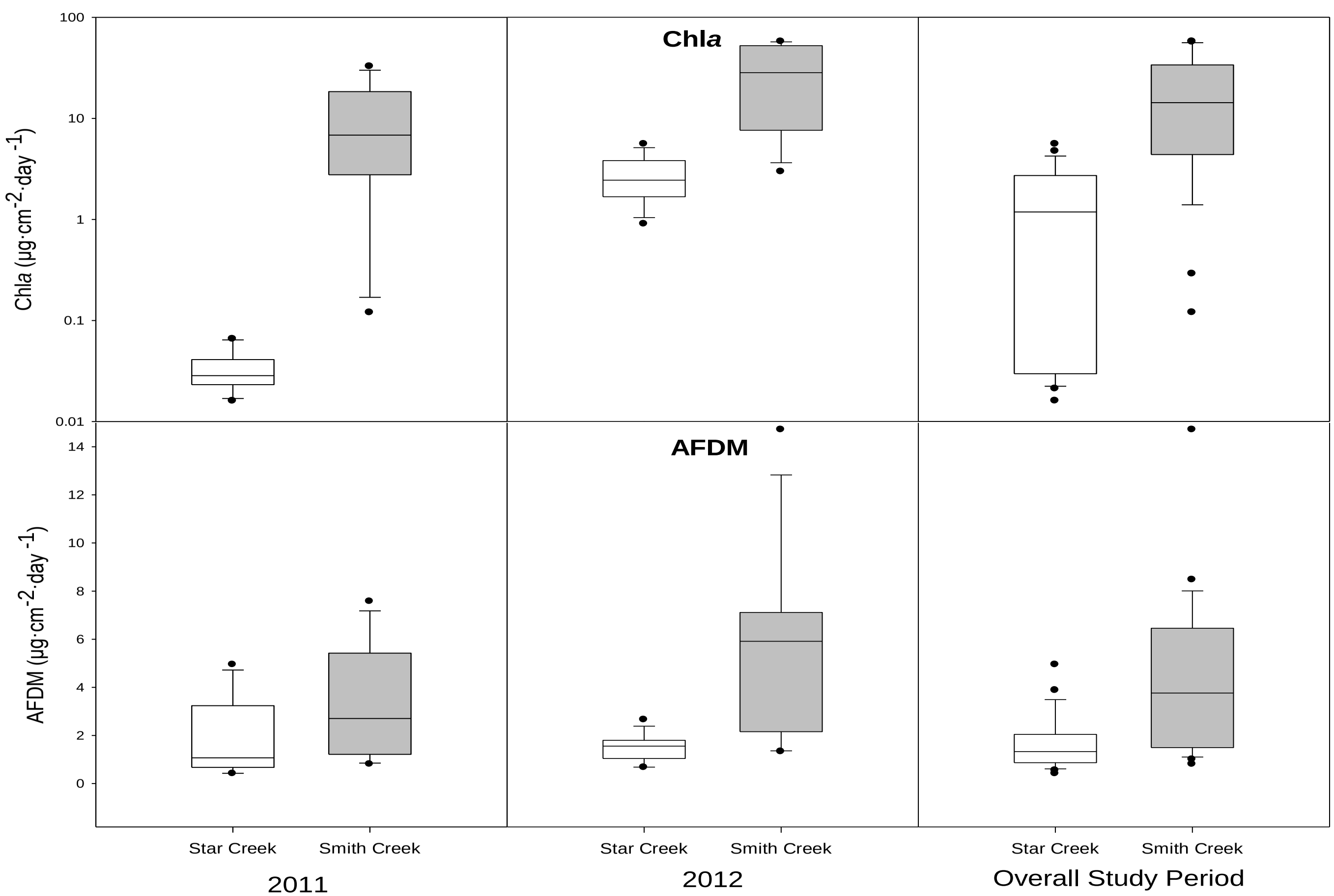

Figure 2-15. Distribution of algal chlorphyll *a* (Chl*a*) and ash-free dry mass (AFDM) production for Star and Smith Creeks by year and overall study period**.** Horizontal line in boxplots indicates median, dashed line indicates mean, upper and lower box position indicates 75th and 25th percentiles, whiskers indicate 95th and 5th percentiles, solid dots indicate outliers.

Figure 2-16. Photographs of the streambeds of Star (top) and Smith Creeks (bottom) depicting anecdotal evidence of higher primary productivity in Smith Creek.

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Chapter 3: Long term effects of logging on streambed sediment-phosphorus dynamics in a northern Rocky Mountain headwater watershed, Alberta, Canada

3.1 Introduction

Headwater regions of watersheds are extremely important for the generation of fresh water, including up to 2/3 of surface water supplies for western North America (Emelko *et al*., 2011), and can represent the majority of cumulative stream network length of larger drainage networks (Benda *et al*., 2005). Forest harvesting disturbance frequently causes fine sediment pollution in forested watersheds by increasing the availability and accelerating the delivery of fine sediment to watercourses (Macdonald *et al*., 2003). Specifically, compacted soils, roads, skid trails, stream crossings and bared areas have been identified as important contributors to fine sediment production during/following logging operations (Anderson and Potts, 1987). The impacts of increased fine sediment concentrations have been extensively researched, particularly in regards to negative effects on fish, fish habitat and a broad spectrum of aquatic biota (Bilotta and Brazier, 2008).

Although numerous studies have examined sediment production during and post-harvest in northern Rocky Mountain headwaters, little research has been conducted on streambed infiltration and storage of fine sediment (Gomi *et al*., 2005). While several studies have shown headwater streams are capable of storing fine sediment following disturbance, particularly in conjunction with deposits of large woody debris (LWD) (Gomi *et al*., 2005; Little *et al*., 2012), the negative impacts associated with increased fine bed sediment (<250 μm) loading have been primarily researched in the context of impacts to salmonid health and reproduction, or alterations of channel morphology and reducing available habitat for a broad range of aquatic flora and benthic organisms (Wood and Armitage, 1997). Although fine sediments, particularly fractions <63 μm are critical vectors for contaminant transport and storage (Walling *et al*., 1997), little attention has been given to bed sediment loading from forest harvest operations as a potentially significant source of pollutants.

Phosphorus (P) is a growth limiting nutrient in many aquatic ecosystems (Chessman *et al*., 1992) that tends to be closely associated to fine sediment for delivery and storage within streams and, when enriched, can contribute to eutrophication and degradation of downstream ecosystems and water quality (Walling *et al*., 1997; Owens *et al*., 2005). Oligotrophic streams, which are typical of the northern Rocky Mountains, can be sensitive to minor increases in P concentrations, which have the potential to cause drastic increases in primary productivity and alterations in ecosystem composition (Withers and Jarvie, 2008). The typically low aqueous concentrations of bioavailable P in headwater systems make P-enriched fine bed sediments an important potential source of P to the water column (House, 2003; Withers and Jarvie, 2008). The biogeochemical cycling of P in rivers is complex and is influenced by many physical, chemical, and biological factors, including the physical composition of streambeds. In headwater streams, which tend to be oligotrophic and have low water volume to bed sediment/benthic area ratios, bed sediment properties affecting P dynamics are of greater significance than in larger systems (House and Denison, 2002; Withers and Jarvie, 2008). Given the disproportionately large influence that headwater streams have on higher order river systems, reductions in water quality and ecosystem degradation caused by bed sediment-facilitated nutrient enrichment have significant implications for land management in the Rocky Mountain region.

There has been limited research on the legacy of forest harvest disturbance in the context of changes in bed sediment composition and P dynamics in northern Rocky Mountain headwater systems. The broad objectives of this study were to investigate the physical and geochemical characteristics of stream bed sediments and their nutrient dynamics in a historically disturbed and undisturbed Rocky Mountain headwaters catchment pair to explore the role of bed sediments as a potentially important component regulating long-term effects of forest harvesting activities on stream ecology. Study objectives included exploring the comparative a) physical characteristics of bed sediments, b) phosphorus-sediment dynamics, and c) influence of large woody debris as a factor promoting sediment-P storage of a disturbed/un-disturbed stream pair in southern of Alberta.

3.2 Materials and Methods

3.2.1 Study Area

Star Creek and Smith Creek watersheds in the headwaters of the Oldman River basin in the Rocky Mountain region of southwestern Alberta near Crowsnest Pass (49°37'N, 114°40'W) were selected as study sites. A detailed description of Star and Smith Creek watersheds is reported in Chapter 2. In brief, Star and Smith Creeks are first order streams, originating at, or very near, the Alberta-British Columbia border, within the Southern Rockies hydrologic region (Wagner, 2010). Both watersheds are similar in size, physical attributes (aspect, slope, elevation) bedrock and surficial geology, soil types and vegetation cover. The most apparent difference in the watersheds is the history of forest disturbance in the Smith Creek watershed. Smith Creek watershed underwent clear-cut forest harvesting starting in the 1960s, through to the mid '90s, and during this period at least 44% of the forest cover was removed. An extensive network of logging roads was built to remove the timber, including two stream crossings. Star Creek watershed has no history of commercial forest harvesting, and disturbance on it is limited to ATV trails and seismic exploration cut lines (Figure 2-5).

3.2.2 Study Design

A descriptive, post-hoc reference/impact study catchment design was implemented for this study. Smith Creek and Star Creek were carefully selected as analogous watersheds, similar in most physical characteristics except for the extensive forest harvesting in Smith Creek. Star Creek served as a generally representative undisturbed reference catchment for Smith Creek, which has been "treated" with long-term history (50+ years) of extensive forest harvesting. Smith Creek is generally representative of extensively harvested watersheds along the High Rock range north of the Crowsnest Pass region. Data on bed sediment particle size distribution, bed sediment P dynamics, and large woody debris were collected from each catchment from April of 2011 until October of 2012 to enable describing differences in sediment-phosphorus dynamics and their linkage with aquatic productivity in undisturbed and disturbed watersheds characteristic of this region.

3.2.3 Bed Sediment Sample Collection

Samples of submerged streambed sediment were collected from six natural depositional sites (behind boulders, debris dams, inside bends of meanders) in the downstream reaches of each stream (within 500 m of gauging stations) in the late summer of 2011 (Figure 3-1) . A freeze-sampling technique, which limits the elutriation of fine sediment by flowing water (Thoms, 1992) was used to collect relatively undisturbed samples of bed sediments from the upper 15 cm profile of the streambeds. The sampler was constructed based on designs and principles reported by Walkotten (1976) and Carling and Reader (1981), but modified to be more portable and durable to suit the remote, coarse nature of the study streams (Figure 3-2). The sampler was driven into the streambed to a depth of 15 cm, then compressed gas (R22a) was passed through the cooling unit (Figure 3-2), causing rapid cooling and freezing of interstitial water and associated sediment to the outside of the sampler. The frozen plug of sediment was then withdrawn from the bed and thawed into a plastic container which was sealed, and frozen until needed for further analysis. The sampling procedure was performed within a bottomless plastic bucket pressed into the streambed to function as a portable stilling well. Three sub-samples were taken from each microsite for separate determination of physical and chemical bed characteristics, yielding 18 streambed samples per stream (6 sites X 3 sub-samples).

3.2.4 Coarse Particle Size Distribution

Particle size distribution by mass for coarse bed sediments (>250 µm) of Smith and Star Creeks was determined using a nested sieve procedure. Frozen, unsorted bed samples were thawed and dried at low temperatures (48°C) for 72 hours, weighed and sieved using a nest of 4 sieves with decreasing mesh sizes (25, 4, 2, 0.25 mm), which sorted the samples into cobble/coarse gravel, fine gravel, coarse sand, medium sand and fine fractions (Wentworth, 1922). The sieves were shaken using a Ro-Tap (model RX-29) sieve shaker for 30 minutes, and each grain size fraction was weighed and used to determine particle size distribution (by mass) for each bed sample. The fine fractions were retained and kept in 20 ml glass scintillation vials in a freezer until needed for further analysis.

3.2.5 Fine Particle Size Distribution

Grain size distribution of fine bed sediments (<250 µm) was determined using a Sedigraph 5100 particle size analyzer. The instrument operates on the principles of Stokes' Law of settling (Jones *et al*., 1988) and uses a finely collimated beam of X-rays to determine concentration of particles in a settling suspension as a function of time (Coates and Hulse, 1985). The analysis yields cumulative mass for a corresponding equivalent spherical diameter (ESD) (Jones *et al*., 1988) based on assumed particle density of 2.65 $g \cdot cm^{-3}$ (quartz). Fine sediment (2.5 g) was prepared by adding 25 ml 0.05% sodium metaphosphate to chemically disperse flocs, and then mechanically dispersed using a sonic bath for one minute prior to being loaded into the Sedigraph 5100. The sample was kept fully suspended in solution using a magnetic stirring table prior to analysis (the sonic bath excluded). The range of particle ESDs ranged from 300 to 0.98 μm at 19 intervals according to the Wentworth (1922) grain size classification scheme.

3.2.6 Phosphorus Speciation

Fine bed sediment samples were thawed and dried at low temperatures (48°C) for 72 hours, gently crushed with a mortar and pestle to remove sediment conglomerates, weighed, then sieved to 250 μm to remove any large organics. A sequential phosphorus extraction procedure was used to determine the relative composition of particulate P species representing a range of loosely to tightly bound P (Psenner *et al*., 1984). The procedure yields three operationally defined particulate P fractions; non-apatite inorganic P (NAIP), apatite P (AP), and organic P (OP) (Stone and English, 1993). Total particulate phosphorus (TPP) was calculated as the sum of the NAIP, AP and OP. A more detailed description of the P extraction procedure appears in Chapter 2.

3.2.7 Phosphorus Sorption Characteristics

Phosphorus sorption isotherms and equilibrium phosphate concentrations (EPC_0) were determined for fine (<250 μm) bed sediments from each stream using a six-point batch isotherm technique based on methods previously reported by House and Denison (2000), Lai and Lam (2009) and Wang *et al*. (2006).

A constant mass of sediment (0.25 g) was added to six 50 ml centrifuge tubes containing 25 ml of phosphate solution of differing concentrations (0, 10, 25, 50, 140, 200 $\mu g \cdot L^{-1}$). The solutions were prepared using de-ionized water and potassium dihydrogen phosphate (KH_2PO_4). Toluene (50 µl) was used to inhibit microbiological activity and 0.5 ml of 0.5M $CaCl_2$ was added to adjust ionic conditions to better resemble native stream water for each solution. Six samples from each creek were prepared in this fashion and the experiment was run in triplicate (n=18 per creek). Samples were placed on a shaker table at approximately 50 rpm for a 24-h equilibration period at room temperature. The tubes were then centrifuged at 5000 rpm for 10 minutes, and an aliquot of the supernatant was analyzed for P concentration using a Westco SmartChem 200 discrete wet chemistry analyzer using the molybdenum blue method (EPA, 1993).

Quantities of inorganic P adsorbed to or desorbed from the sediment were calculated using the equation: $P_{ads} = [(P_{initial} - P_{final}) * 0.025L]*wt^{-1}sediment$ (Wang and Li, 2010; Froelich, 1988). P sorption isotherms were created by plotting the linear relationship between $P_{initial}$ values (x axis) and P_{ads} on (y axis). Negative values of P_{ads} represent net desorption of native P from sediment to solution and positive values are interpreted as P adsorbed from solution to the surface of the solid particles (Froelich, 1988). Equilibrium phosphate concentrations (EPC_0) are represented by the $P_{initial}$ concentration where $P_{ads} = 0$.

3.2.8 Large Woody Debris

Surveys of large woody debris (LWD) were conducted on Smith and Star Creeks under low flow conditions in late summer of 2012. Five 10m stream reaches were surveyed in the alluvial sections of each stream, spaced 20m apart. There are no standard criteria for the size of material that constitute LWD, but the dimensions used here are the same as used in similar research (Hauer *et al.*, 1999); LWD was defined as logs ≥ 0.1m in diameter and ≥ 1.0m in length. To be recorded as LWD, pieces had to be within the stream channel and below the bank-full stage (identified by the point on the bank where terrestrial moss and rooted vegetation cease) (Gomi *et al*., 2001). LWD straddling the end of survey sections had only the portion within the survey section recorded, allowing a more accurate volume of LWD per unit length of stream to be calculated. LWD extending above the bank-full stage mark were similarly excluded. The length of each piece was measured with a tape measure and the diameter at the midpoint was measured with calipers. Only the visible length of each piece was recorded, as excavation would be required to get complete measurements of embedded pieces. Difficulty accessing embedded tops and butts necessitated the use of the midpoint diameter for calculation of volume for each LWD piece. Piece volume was calculated using a modified Smalian's volume formula (ESRD, 2006), with the midpoint diameter of the piece representing the average of the butt and top diameters. The short nature of the LWD (average length approximately 2.3 m) reduces the importance of taper in the volume calculations. Mean LWD volume$\cdot$km^{-1} was calculated for each stream.

3.2.9 Statistical Analysis

All data analyses were performed using R statistical software (R Core Team, 2013) using a significance threshold of $\alpha = 0.05$. All data sets were tested for normality using Q-Q plots and Shapiro-Wilk tests (α=0.05). Coarse bed particle size distribution data met assumptions of normality and the means of each fraction from Star and Smith Creeks were compared using Student's t-tests. Bed sediment P concentrations from Star and Smith Creeks met assumptions of normality and mean concentrations of all P species in each creek were compared using Student's t-tests. The P sorption curves for the bed sediment from each creek were compared using overall tests for coincidental regression (Zar, 1999), as well as individual comparisons of slopes and intercepts. Large woody debris data failed to meet assumptions of normality at $\alpha = 0.05$, thus a Wilcoxon-signed rank test was used to non-parametrically compare distributions of LWD volumes in Star and Smith Creeks.

3.3 Results

3.3.1 Coarse Particle Size Distribution

The particle size distributions of bed sediments representing coarse particle size fractions (>250 µm) were similar between Smith and Star Creeks. The mean proportion of the coarser bed sediments (coarse gravel, fine gravel, coarse sand and medium sand) comprising stream beds did not differ (p=0.07-0.49) between Star and Smith Creeks (Table 3-2, Figure 3-3). However, while not significantly different at α=0.05, mean fine bed sediments (<250 µm) in Smith Creek comprised 16.3% compared to only 6.3% of bed sediments in Star Creek (Table 3-2) and the distribution of sediment in the fine sediment class contained notably greater proportion of finer particles (Figure 3-3).

3.3.2 Fine Particle Size Distribution

The distribution of bed sediments comprising fine particles (<250 µm) were also generally similar between Star and Smith Creeks across the range of particle ESDs from 300 to <0.98 µm (Figure 3-4 and Table 3-3). The percentage of particles finer than a given ESD was very similar (within ~3%) throughout the range of particle sizes (Table 3-3) with the exception of the smallest particles (<3.91 µm), where the bed sediments of Smith Creek contained a greater proportion of these smallest particle sizes than were observed in Star Creek. The greater proportion of the finest sediment class was most clearly evident in the distribution of the finest sediments (Table 3-4) where the mean particle size of the 10th percentile of the fine sediment distribution in Star Creek was approximately double that of Smith Creek (2.0 compared to 1.1 μm ESD, respectively).

3.3.3 Bed Sediment Phosphorus

The concentration of all P species was greater in bed sediment collected from Smith Creek than from Star Creek in 2011 (Figure 3-5). Among the three fractions of sediment associated P making up TPP in bed sediments, NAIP was most strongly different between catchments. NAIP of sediments from Smith Creek were 99% greater than that observed in Star Creek (124 $\mu g \cdot g^{-1}$ compared to 62.6 $\mu g \cdot g^{-1}$ sediment dwt for Smith and Star Creek, respectively; p<0.001). AP, OP and TPP concentrations were all similarly higher in Smith Creek bed sediments (Table 3-5). Bed sediment AP in Smith Creek (640.6 $\mu g \cdot g^{-1}$sediment dwt) was 98% greater than that of Star Creek (324.1 $\mu g \cdot g^{-1}$sediment dwt; p<0.001). OP concentration in Smith Creek was 135.0 $\mu g \cdot g^{-1}$sediment dwt, 90% higher than the bed sediments of Star Creek (71.1 $\mu g \cdot g^{-1}$sediment dwt; p<0.001). Bed sediment TPP concentration, which is the sum of NAIP, AP and OP was 900.3 $\mu g \cdot g^{-1}$sediment dwt in Smith Creek, which was 97% greater than that of Star Creek (457.8 $\mu g \cdot g^{-1}$ sediment dwt; p<0.001). The proportions of NAIP, AP and OP species comprised approximately 14%, 71%, and 15% of total PP associated with the bed sediments of both watersheds, and were very similar for both the disturbed and reference catchments (Table 3-5).

3.3.4 Phosphorus Sorption Characteristics

The relationships between aqueous P concentration and P sorption to sediments differed between the bed sediments of Star and Smith Creeks (Figure 3-6, p<0.001). No difference in slopes of the relationships between initial P concentration and P sorption where observed (p=0.41), indicating that the rate at which the bed sediment of Star and Smith Creeks adsorb/desorb P across the given range of aqueous P concentrations was not different. However, the regression intercept of the sorption relationship in Star Creek was greater than that evident in Smith Creek (p<0.001). As a consequence, the EPC_0 (concentration of aqueous P at which the sediment neither adsorbs nor releases P) of Smith Creek bed sediment was 74% greater than that of Star Creek (68 $\mu g \cdot L^{-1}$ and 39 $\mu g \cdot L^{-1}$ for Smith and Star Creek, respectively).

3.3.5 Large Woody Debris

The stream channel in Smith Creek contained >5 times greater mean LWD volumes ($m^3 \cdot km^{-1}$) than was observed in Star Creek (Figure 3-7). In 2012, Smith Creek contained a median LWD volume of 68 $m^3 \cdot km^{-1}$, compared to 5.4 $m^3 \cdot km^{-1}$ LWD observed in Star Creek (p=0.03).

3.4 Discussion

3.4.1 Particle Size Distribution

The results of the coarse particle size distribution (>250 µm) analysis indicated that the streambeds of Star and Smith Creeks had similar composition of coarse materials. Although none of the size fractions showed statistically significant differences in percent composition by mass, the finest fractions of these bed materials did comprise slightly more of the streambed in Smith Creek than in Star Creek; the median fine sediment composition in Smith Creek bed sediments was approximately 16%, while it was only 6% in Star Creek. The strong similarity of coarse bed sediment fractions from Smith and Star Creek were, however, consistent with the physiographic and hydrological similarity of factors controlling bed and channel geomorphology (Benda *et al*., 2005). The surficial and bedrock geology of the two catchments are very similar, as are the stream gradients and stream discharges (in 2011/2012). Thus, it was not surprising that the coarse streambed sediment fractions did not differ between the Star and Smith Creeks.

This general similarity of streambed materials among watersheds was also evident in the finer sediment fractions (<250 μm). However, the streambed of Smith Creek was comprised of a greater proportion of extremely fine particles (<5 μm) compared to Star Creek. Given the similarities in physical and hydro-climatic conditions of the catchments, it is possible that the difference in fine particle size composition was the result of past disturbance in the Smith Creek watershed. The soil disturbance associated with logging and road building is known to increase the abundance of fine particles by accelerating erosion and promoting the breakdown of soil aggregates and weathering of primary particles (Megahan and King, 2004). This is supported by Bilby (1985), who determined that the majority of sediment produced by a logging road was less than 4 μm in diameter. Logging practices can result in soil compaction, placement of unconsolidated material on slopes, and concentration of surface runoff which contributes to the increased delivery of fine sediment to watercourses. Logging of 15% of the Salmon River watershed in central Idaho resulted in drastic increases in proportion of fine materials in the streambed (Binkley and Brown, 1993). Although it has been suggested that high-energy headwater streams have little ability to store very fine sediment (Benda *et al*., 2005), a number of studies have documented long-term in-stream storage in Rocky Mountain headwater settings (Benda and Dunne, 1997, Gomi *et al*., 2005, Little *et al*., 2012). A combination of limited streamflow, a step-pool channel morphology, and high surface roughness associated with boulders and LWD, limits sediment transport and promotes deposition and storage (Benda *et al*., 2005). Considering this, it seems likely that the greater proportion of fine sediments observed in Smith Creek were likely caused by the combination of a) historic logging disturbance promoting terrestrial erosion, weathering, and delivery of sediments into Smith Creek, and b) more efficient subsequent storage of fine material within the streambed of Smith Creek because of the greater LWD observed in the channel of that stream.

3.4.2 Bed Sediment Phosphorus

Bed sediment-associated P concentrations of all four P species were greater in Smith Creek than Star Creek in 2011. While the lack of pre-disturbance data for Smith Creek makes it more difficult to conclusively attribute the differences in sediment P to differences in disturbance history among watersheds, there is sufficient evidence to support the concept that the elevated sediment-P concentrations in Smith Creek was the result of the extensive past logging and road construction disturbance within that watershed. Both watersheds are underlain by the same three geologic formations in similar proportions and the surficial geology is very similar (Figure 2-2, Table 3-1), thus the ultimate source of P in each catchment was likely the same (Froelich, 1988). Furthermore, the proportions of NAIP, AP and OP which make up TPP were very consistent between the two watersheds (Table 3-5), suggesting that the biogeochemical processes governing P dynamics are likely very similar between the two catchments. The elevated levels of sediment associated P are consistent with what is expected for a disturbed watershed, and is likely related to the past delivery and subsequent storage of fine sediments generated by historic logging and road disturbance.

Fine sediments stored within streambeds are known to exert control over the geochemical cycling of P in rivers, and anthropogenic disturbance, like the harvesting in Smith Creek, has been reported to increase loading of sediment (House, 2003). Bed sediments can subsequently act as a source of P, which can desorb and diffuse upward into the water column, where the bioavailable component (SRP) becomes available for uptake by various plants and microorganisms (Withers and Jarvie, 2008). The rate at which P desorption occurs depends on multiple factors including aquatic P concentrations, flow velocities, turbulence, biological assimilation rates at the benthic interface, pH and redox conditions (House and Denison, 2002; Withers and Jarvie, 2008). Under oligotrophic conditions such as in Star and Smith Creeks, low aqueous P concentrations create a steep dissolved P concentration gradient between the benthic interface and overlying stream water, facilitating desorption and upward diffusion/convection of P from bed sediments (Withers and Jarvie, 2008). The bed sediments of Smith Creek were significantly richer in P than in Star Creek, thus the potential for flux of P from bed sediment to the water column where NAIP would be available for biological assimilation was likely also greater. However, the processes governing the release of P from bed sediments are not simply controlled by chemical gradients and diffusion/convection. Periphyton and phytoplankton can influence bed-P dynamics, particularly in headwater systems which tend to be fast moving, shallow, and transmissive to light (Dodds, 2003). Withers and Jarvie (2008) reported that benthic periphyton affected P cycling by intercepting P diffusing from bed sediments, creating biogeochemical conditions favorable for dissolved P removal at the benthic interface (increased O_2 concentration, decreased pH). These same conditions also promote production of OP through decomposition processes at the benthic interface (House, 2003). The net influence of these biotic effects are to limit P export and increase storage, shortening the length of nutrient spirals as well as limiting or dampening the upward diffusion of bioavailable P from bed sediments (Newbold *et al*., 1983). Thus, the significantly greater levels of primary productivity observed in Smith Creek in Chapter 2 are consistent with the relatively high levels of bed-P observed in Smith Creek. Benthic organisms in Smith Creek likely contributed to P storage in that system.

3.4.3 Phosphorus Sorption Characteristics

The results of the P sorption analysis indicate that the fine bed sediments in Smith Creek had a higher EPC_0 which was likely related to the greater concentrations of sorbed P observed in Smith Creek. While the artificial laboratory conditions under which the sorption experiments were conducted differ from actual stream conditions in either Star or Smith Creek (warmer temperatures, still water, differing ionic composition, absence of biotic influences), the sorption experiment does yield information on relative differences in potential streambed P dynamics between these watersheds. The higher EPC_0 in the bed of Smith Creek (Figure 3-6) suggests that at relatively low aqueous P concentrations, Smith Creek bed sediments are more of a source of P than are the bed sediments of Star Creek. This is consistent with the results of the P speciation analysis, which showed approximately double the concentrations of all P species existed in Smith Creek bed sediments compared to that of Star Creek The most likely mechanism driving this difference among watersheds is the differences in fine particle distribution evident between Smith and Star Creek Withers and Jarvie (2008) reported particle size having a major influence on P sorption for two reasons: the smallest particles have larger specific surface areas for P sorption, and clay sized particles (<2 μm) tend to have higher proportions of Fe and Al cations which are highly affinitive to phosphate ions. The higher EPC_0 of Smith Creek suggests bed sediments in that watershed had the potential to release greater quantities of dissolved P particularly if favorable conditions arise, such as remobilization of bed material or decreased concentration of overlying aqueous P. (House, 2003).

3.4.4 Large Woody Debris

The channel of Smith Creek was found to have >5 times more LWD (by volume) than Star Creek. The difference in LWD volume between the streams was likely driven by a number of factors including disturbance history, but also riparian (stream-adjacent) stand age and topography (Benda *et al*., 2005). Increased frequency of mass-wasting events and debris flows following forest harvesting have been identified as mechanisms of increased LWD input in the

Rocky Mountains of the Pacific Northwest (May, 2002), but in the drier, more stable conditions of the east slopes of the northern Rocky Mountains mass-failure is infrequent (Anderson and Potts, 1987). Post-harvest windthrow in riparian buffers can be a significant mechanism for LWD recruitment in Rocky Mountain streams (Bahuguna *et al.*, 2010) and the harvest in the Smith Creek watershed likely contributed to increased LWD inputs into the channel. Additionally, un-harvested stands adjacent (within 30 m) are generally older in Smith Creek than in Star Creek, thus are more prone to natural mortality and tree fall. A more extensive LWD survey would be required to determine the relative importance of the specific factors contributing to the greater LWD in Smith Creek and the association with fine sediment storage. Numerous studies have reported that large wood from riparian stands can allow high energy headwater streams to accumulate and store significant volumes of fine sediment (Gomi *et al*., 2001; Jackson and Sturm, 2002; Benda *et al*., 2005). LWD increase sediment retentiveness (deposition/storage) by reducing flow velocities, creating step-pool channel morphology which limit shears stress, increasing surface roughness, and promoting pool formation (Benda *et al*., 2005; Little *et al*., 2012). In headwater streams in Idaho, Megahan (1975) reported that less than 10 percent of sediment stored behind LWD obstructions was released over a three year period, highlighting the potential for LWD to retain sediment in mountainous headwater systems.

Although the higher volume of LWD in Smith Creek did not directly cause the production of finer sediment, it likely did provide a mechanism through which the fine sediment generated by harvesting operations was retained within the otherwise high stream energy conditions of Smith Creek.

3.5 Conclusion

The greater proportion of very fine sediment and its role in storage of sediment-associated P in the bed of Smith Creek is potentially a legacy of the history of extensive logging and road-building disturbance in that watershed. Star and Smith Creek watersheds have similar physical attributes that govern geomorphological processes including surficial and bedrock geology, thus catchment differences not related to harvesting were unlikely to have been significant drivers of the differences observed in streambed particle size distributions and P dynamics. Indeed, a series of biogeochemical processes including a) greater proportion of fine sediments, b) their association with greater storage of bioavailable (NAIP) and other forms of particulate P, c) greater aqueous P exchange potential (EPC_0) of sediments, and d) greater LWD capable of promoting more efficient fine sediment storage in Smith Creek establish a series of strong mechanistic linkages with the chronic sediment inputs associated with harvesting history of that watershed. The abundant LWD in Smith Creek provided a mechanism that enabled more efficient long-term storage of fine sediment in a headwater system with high potential for sediment transport. The increased proportion of very fine sediment in the bed of Smith Creek functions as a storage mechanism for P, which is capable of desorbing into aqueous P. The sorption curves for the fine sediment from the two streambeds support the assertion that the P stored in Smith Creek can be more readily released and can become available for uptake by plants and microorganisms, ultimately driving an increase in primary productivity. This study highlights the potential for sediment-driven nutrient P enrichment related to forest harvesting operations to affect Rocky Mountain headwater systems (and potentially contribute to downstream cumulative effects) decades after logging disturbance.

Tables and Figures

Table 3-1. Surficial geologic deposits in Star and Smith Creeks (%).

	Stream	
Geological Deposit	Star Creek	Smith Creek
Bedrock	24.4	21.5
Talus	14.7	11.1
Colluvium	36.7	11.0
Slightly/Moderately Leached Till	22.9	54.1
Other	1.4	2.5

Table 3-2. Mean coarse particle size fractions (% by mass) for bed sediments from Star and Smith Creeks in 2011.

Particle Size Fraction	Stream: Star Creek	Smith Creek	*p*-value
Coarse Gravel	17.2	17.3	0.49
Fine Gravel	42.5	35.4	0.10
Coarse Sand	12.7	10.0	0.07
Medium Sand	21.3	20.6	0.41
Fine Sand	6.3	16.3	0.09

Table 3-3. Fine particle (<300 µm) size distribution of bed sediments in Star and Smith Creeks in 2011. Values indicate percentage of fine particles that are smaller than given diameter.

	Stream	
Particle Diameter (µm)	Star Creek	Smith Creek
0.98	6.4	9.6
1.95	9.8	12.4
3.91	16.0	16.8
7.81	24.5	24.4
15.63	36.4	34.8
31.25	49.1	48.3
37	53.6	53.3
44	59.0	59.4
53	65.5	66.9
62.5	71.7	73.8
74	77.6	80.1
88	82.3	85.0
105	85.9	88.7
125	89.0	91.7
149	90.8	93.6
177	92.1	94.9
210	93.4	96.2
250	94.7	97.4
300	96.1	98.8

Table 3-4. Particle sizes (µm) of fine sediment distributions from the beds of Star and Smith Creeks in 2011.

	Stream	
Percentile	Star Creek	Smith Creek
90	138	113
75	69	65
50	32	33
25	8.1	8.1
10	2.0	1.1

Table 3-5. Mean bed sediment concentrations of four particulate P species ($\mu g \cdot g^{-1}$ sediment dwt) in 2011 for Star and Smith Creeks. Values in brackets indicate proportion of each species (non-apatite inorganic P (NAIP), apatite P (AP), organic P (OP)) as a fraction of total particulate P (TPP), *p*-values reflect Student's t-tests between watersheds.

	Stream		
P species	Star Creek	Smith Creek	*p*-value
NAIP	62.6 (0.14)	124.6 (0.14)	<0.001
AP	324.1 (0.71)	640.6 (0.71)	<0.001
OP	71.1 (0.16)	135.0 (0.15)	<0.001
TPP	457.8 (1.00)	900.3 (1.00)	<0.001

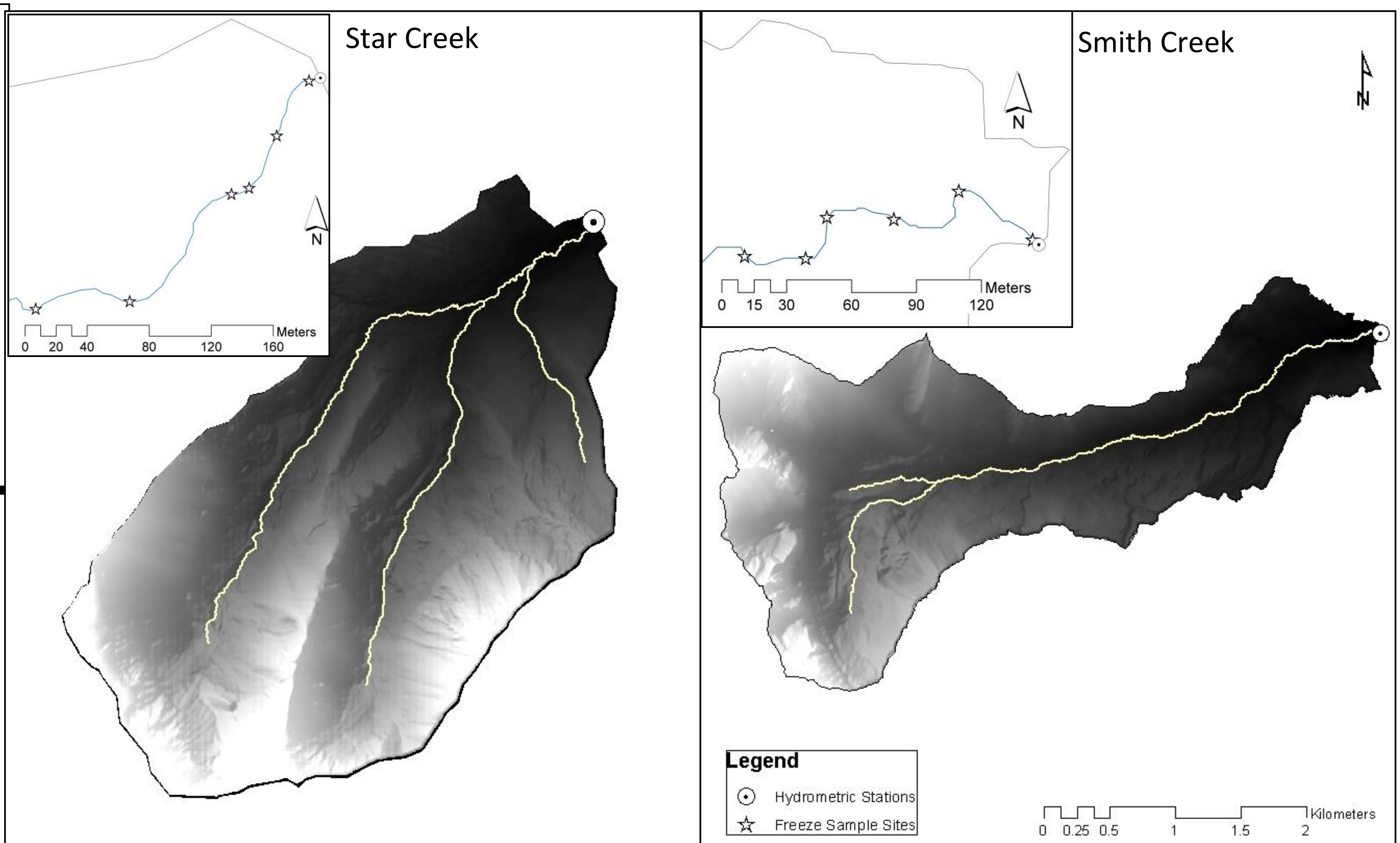

Figure 3-1. Map of Star and Smith Creek watersheds showing locations of hydrometric stations and freeze sampling sites (inset).

Figure 3-2. Freeze sampler. Design of freezing unit (top left), design of entire sampler (top right) and photos depicting the sampler in use, with bed sediment frozen to sampler (bottom).

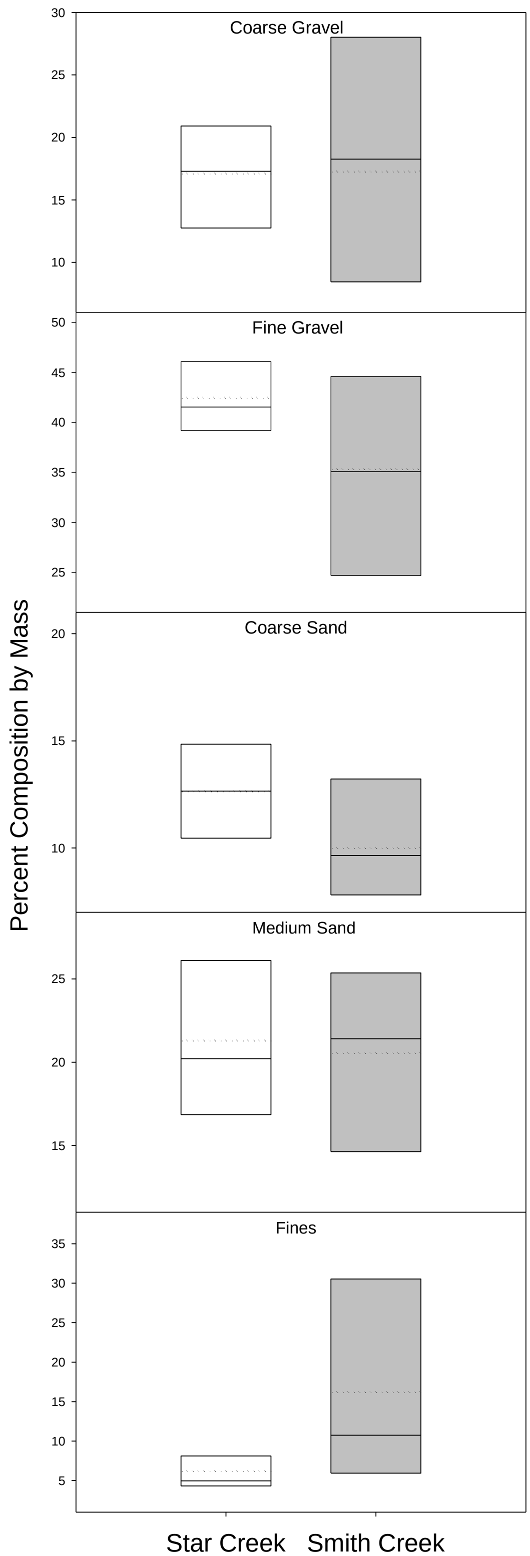

Figure 3-3. Distribution of coarse particle size distributions of bed sediments in Star and Smith Creeks in 2011. Horizontal line in boxplots indicates median, dashed line indicates mean, upper and lower box positions indicate 75th and 25th percentiles.

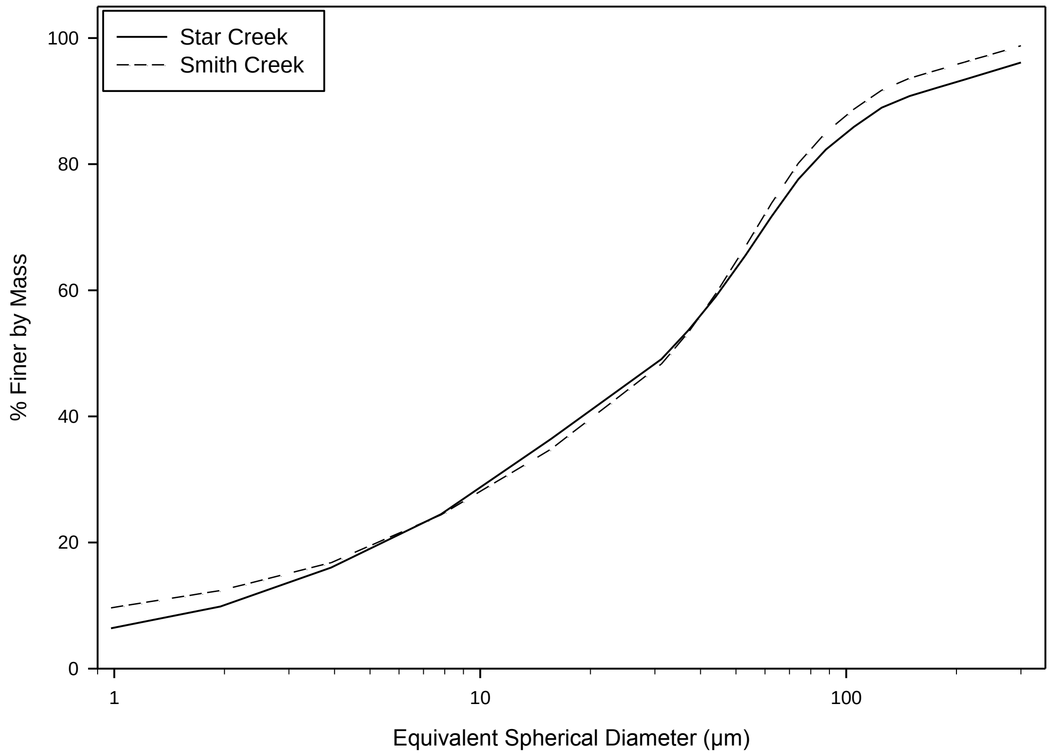

Figure 3-4. Particle size distribution for fine fractions of bed sediments of Smith and Star Creeks in 2011.

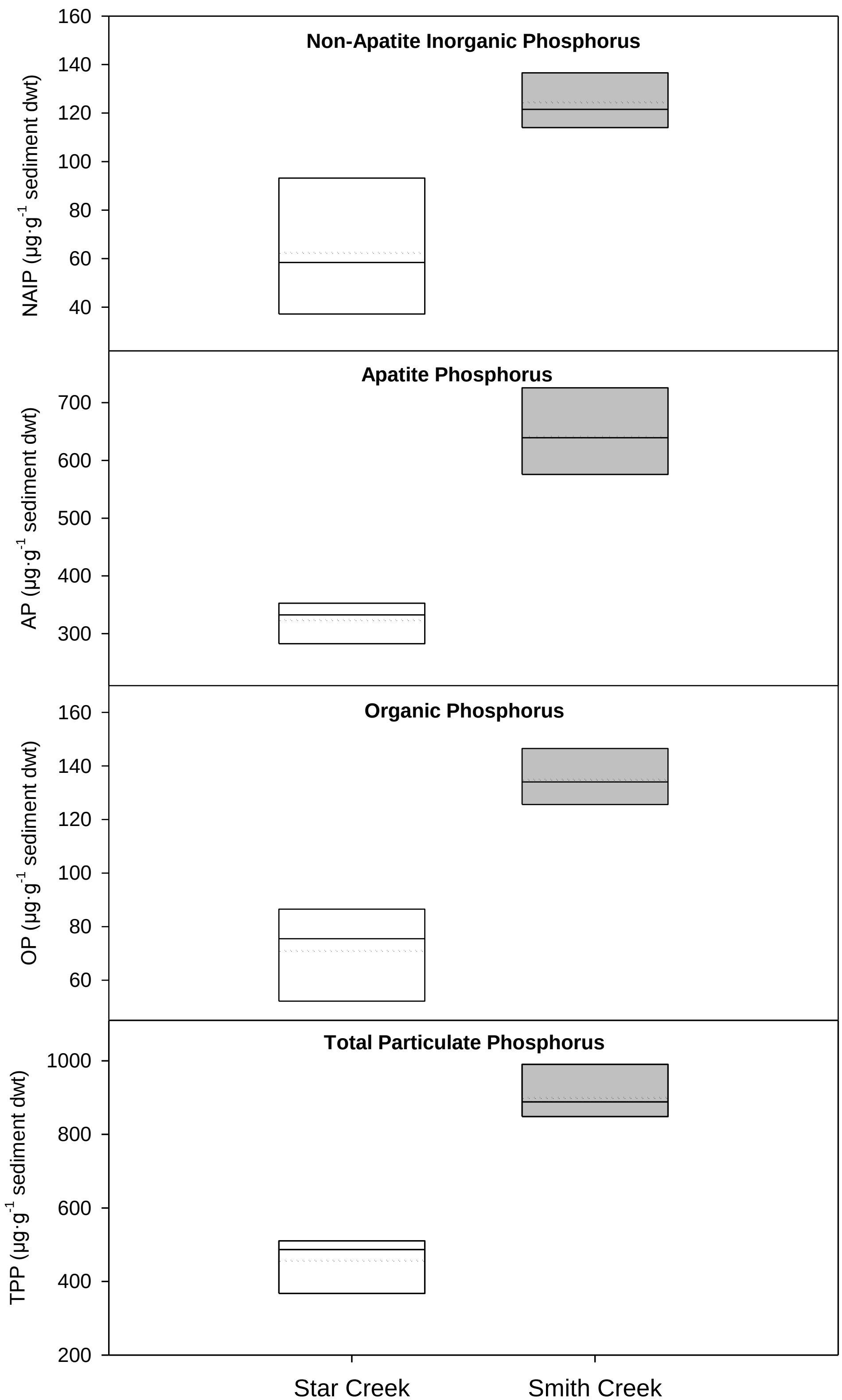

Figure 3-5. Distribution of bed sediment P concentrations for all particulate P species in Star and Smith Creeks in 2011. Horizontal line in boxplots indicates median, dashed line indicates mean, upper and lower box positions indicate 75th and 25th percentiles.

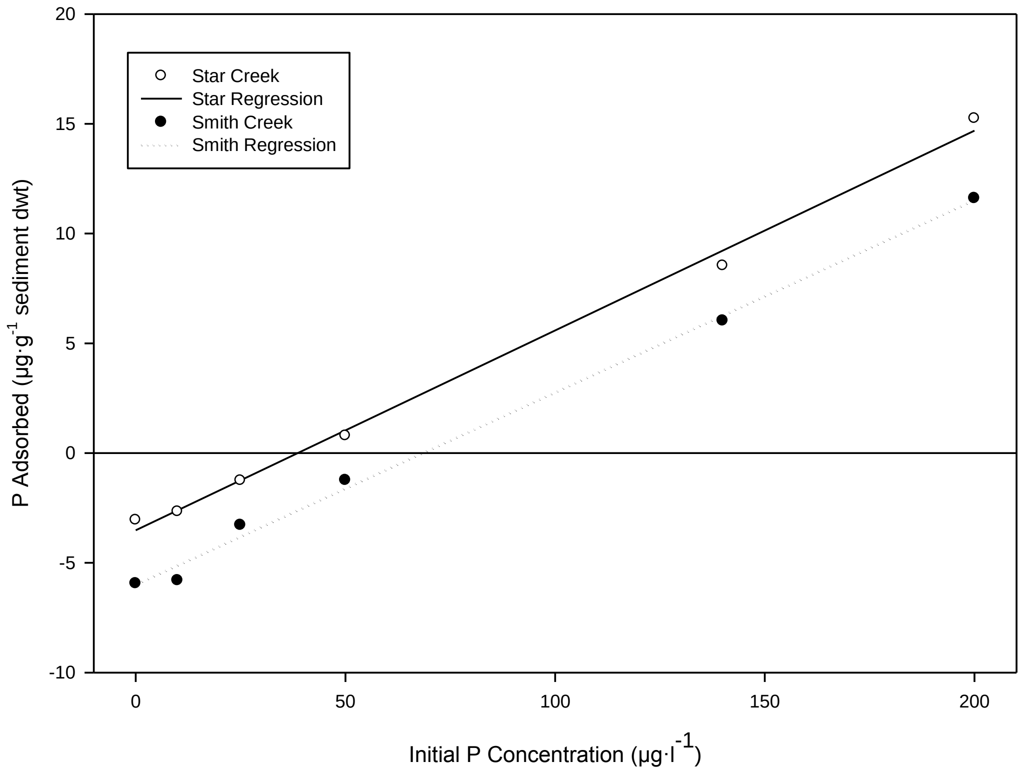

Figure 3-6. Bed sediment P sorption curves for Star and Smith Creeks in 2011. EPC_0 Star = 39 µg·L^{-1}. EPC_0 Smith = 68 µg·L^{-1}.

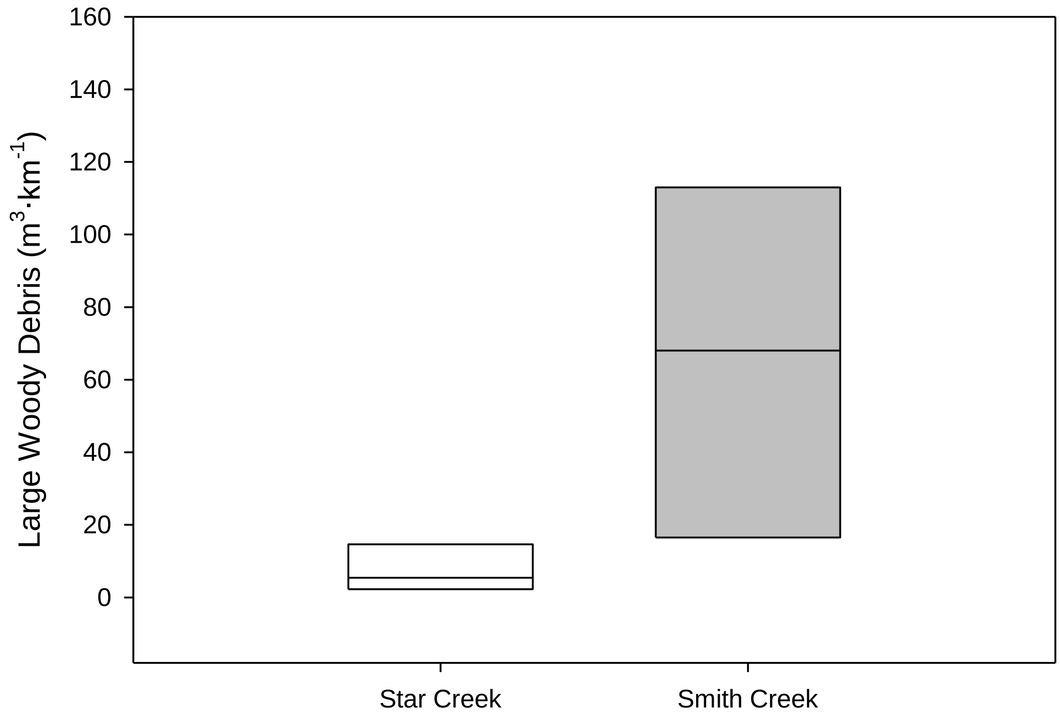

Figure 3-7. Distribution of large woody debris (LWD) volumes in Star and Smith Creeks in 2012. Horizontal line in boxplots indicates median, upper and lower box positions indicate 75th and 25th percentiles.

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Chapter 4: Synthesis

The overall objectives of this study were to determine if forest harvesting disturbance leaves a legacy of significant, detectable effects on sediment-nutrient dynamics and primary productivity in northern Rocky Mountain headwater systems many decades after conclusion of harvest activities.

The first data chapter of this study (Chapter 2) reported on differences in the surface hydrology, aqueous chemistry, suspended sediment-nutrient dynamics, and primary productivity of Star and Smith Creeks to infer how past logging may still be affecting stream biogeochemistry and ecology. The results of Chapter 2 highlight the potential longevity of harvesting disturbance in headwater streams of the northern Rockies. The surficial hydrology of Smith Creek appeared to have recovered in the decades post-disturbance, consistent with previous findings in Alberta, but evidence of alterations to the suspended sediment and nutrient regimes was detected in the logged catchment. The slightly higher levels of TSS in Smith Creek were possibly a result of past sediment introductions which occurred at, or close to, the time of disturbance. In-channel storage associated with LWD and subsequent re-mobilization and transport of sediments were likely important factors in the downstream propagation of upstream harvest impacts. Channel storage and transport of sediment and sediment associated P were likely the fundamental drivers of the differences observed between disturbed/undisturbed watersheds in sediment and aqueous P concentrations and ultimately, in primary productivity among the study watersheds. The suspended solids of Smith Creek, a portion of which were generated during/post-harvest, had higher associated concentrations of all species of P. It is plausible that the difference in sediment-P concentrations was a result of the finer distribution of suspended solids generated by advanced weathering and erosion processes associated with logging. These finer sediments are capable of adsorbing greater quantities of P due to higher surface area-volume ratios. The greater concentrations of TSS and higher concentrations of sediment-associated P in Smith Creek (particularly bioavailable NAIP) established the likely mechanism for the dramatically elevated primary productivity evident in that watershed.

The second data chapter (Chapter 3) reported on the comparative particle size distributions, P concentrations and P-sediment dynamics as well as LWD content in the streambeds of Star and Smith Creeks to determine if past logging disturbance was reflected in altered streambed geomorphological and chemical characteristics with the potential to affect stream ecology. The results illustrated the potential for logging disturbance to be reflected in streambed composition and particulate P concentrations >20 years after the cessation of operations. The streambeds of the disturbed and undisturbed catchments were not different in terms of compositions of coarse particles (>250 μm), but Smith Creek had a greater proportion of extremely fine particles. The similarity in the physical basin characteristics and geomorphology of the two catchments were likely important factors driving the similarities observed in the coarse size distributions of the two streambeds. Weathering, erosion, and sediment delivery processes accelerated by logging and road construction are probable drivers for the increased proportion of very fine sediment in the bed of Smith Creek. However, fine sediment loading in Smith Creek was likely greater closer to the time of harvest operations, but in the decades since much of that sediment was probably re-entrained and transported either out of the system or re-deposited some distance downstream. Benda and Dunne (1997) describe the in-stream movement of non-uniformly introduced sediment as waves of transport and storage moving downstream at the potentially large temporal scales. The high levels of LWD in Smith Creek would have assisted in the long-term storage of fine particles and retarded or buffered some of this downstream transport out of the watershed. The fine sediment particles, particularly the very finest fractions, are effective vectors of storage and transport of phosphorus (Froelich, 1988), and the greater bed sediment-P observed in Smith Creek can be attributed to the elevated loading of these particles. Although the difference in fine sediment loading was relatively small, the increase in surface area available for P sorption is much larger, and establishes mechanistic explanation for the finding of a near two-fold difference in bed sediment P among watersheds.

Chapters 2 and 3 of this study are closely connected because of the dynamic linkages between aqueous nutrients, suspended solids, bed sediments, and primary producers which govern the biogeochemical conditions of a stream (Figure 4-1). The elevated aqueous P and TSS concentrations in Smith Creek discussed in Chapter 2, were influenced by streambed sediment composition and bed-P loading. The significantly greater concentrations of P observed in the bed of Smith Creek, along with a higher EPC_0, indicate that under conditions of low aqueous P (such as during low-flow periods), P will diffuse upward into the water column and increase dissolved P concentrations. The increased fine sediment loading of the streambed of Smith Creek also provides a source of sediment for re-entrainment and remobilization during periods of high flows and bed-load moving events. The occurrence of this process was reflected in the observations of higher turbidity and TSS concentrations in Smith Creek, particularly when considering no significant point sources of sediment were located during thorough reach-walks. This re-suspension of fine sediments also promotes sorption/desorption reactions of P by allowing sediment from deeper bed strata to mix with stream water, which tends to have considerably lower P concentrations (see Chapter 2 results). This increased chemical gradient accelerates release of P from fine sediment and ultimately increases aqueous concentrations of dissolved P. This concept is consistent with the elevated concentrations of suspended sediment P and aqueous TP and SRP detected in Smith Creek. The concentrations of TPP and NAIP were consistently greater in suspended solids from Smith Creek and were incrementally more elevated in the streambed sediments for all P fractions. However, the differences in aqueous P concentration between Star and Smith Creeks were much more subtle. Over the course of the study, there was no difference in TDP concentration, and neither SRP nor TP concentrations were more than 25% greater in Smith Creek. The relative decrease in P concentrations from bed sediment to suspended sediment to the water column is driven by numerous biogeochemical influences, including biological uptake. Northern Rocky Mountain headwater streams like Star and Smith Creeks are typically oligotrophic, and the biological demand for P exceeds supply. As P becomes available (particularly NAIP) for uptake via desorption from sediment, a significant proportion is rapidly assimilated by the nutrient-starved aquatic flora and fauna. The interception of desorbed bioavailable P by primary producers provides an explanation for the effective dampening of aqueous P concentrations. If this process did not occur, we would expect to see aqueous P concentrations resembling the EPC_0s in Star and Smith Creeks. The drastic difference in primary productivity between Star and Smith Creeks is further evidence of the complex dynamic interactions between sediment, P, and aquatic ecology. I propose that the increased growth of benthic periphyton in Smith Creek was driven by the P desorbing from fine bed sediment (and remobilised, suspended sediment), which may have been introduced by decades of logging disturbance. These processes are depicted in Figure 4-1 and illustrate the important roles likely played by LWD in fine sediment storage and streambed particulate P desorption in facilitating periphyton growth. The P-limited nature of these streams ensures the uptake of the P by primary producers is nearly complete, and results in dampening the concentrations of TP and SRP in the water column. This process was reflected in the relatively low concentrations of aqueous SRP and TP compared to bed and suspended sediment NAIP and TPP concentrations in both Star and Smith Creeks. Furthermore, although Smith Creek had a much higher concentrations of bed-sediment P, this difference was much less pronounced in aqueous P, presumably because of uptake by primary producers. The additional P in Smith Creek appeared to support dramatically more primary production, which in turn likely limited the aqueous P concentrations to levels only slightly higher than that of undisturbed catchments.

The results of this study offer a number of potentially important insights on the diversity of impacts that extensive logging can have on headwater stream systems in the northern Rocky Mountains. Extensive forest harvest disturbance can impact the sediment regime of headwater systems for much longer than previously reported in the Northern Rockies. A small number of studies have examined sediment dynamics following logging operations in the northern Rockies (Swanson *et al*., 1986; Anderson and Potts, 1987; Nip, 1991) but none investigate the legacy left after 30+ years. A few studies have examined long term changes in sediment dynamics following logging disturbance and road construction (Beschta, 1978; Reiter *et al.*, 2009), but these occurred in the coastal headwaters of the Pacific Northwest and the differences in climate, soils, and geology limit the direct applicability of their results to the continental Rocky Mountain region.

The results of this study also highlight potential for long term streambed storage of fine sediment produced by forestry operations. Long term bed storage processes have been observed following other watershed disturbances which introduce large volumes of fine sediment (i.e., landslide, wildfire) (Benda *et al*., 2005; Gomi *et al*., 2005, Little *et al*., 2012), but this concept has been largely ignored in the context of logging disturbance. Moreover, this study illustrates the critical importance of fine sediment as a vector for the transport and storage of P, in addition to directly impacting habitat of salmonids and various other biota, which is traditionally the focal point regarding sediment pollution following logging. The P-enrichment observed in the suspended and bed sediments of the disturbed catchment in combination with the drastically elevated primary production clearly illustrates the potential for fine sediment pollution to significantly affect aquatic ecology in headwater streams for decades following logging disturbance. This important series of potential ecohydrological impacts from forest harvesting, or the long-term duration of these impacts have not been previously described in the research literature to my knowledge.

These findings reinforce the importance of implementation of forestry and road construction best-management practices in northern headwater catchments. The apparent longevity of introduced fine sediment (even in steep, high-energy creeks) clearly illustrates the importance of strict sediment control protocols during all phases of forestry operations (planning, operations, and reclamation). The sediment-facilitated increases in biologically available P, and subsequent increase stream productivity should draw attention to the importance of protecting water bodies even in the absence of obvious anthropogenic values (i.e., game fish). Headwater stream systems serve as critical habitat areas for numerous species, and require careful management to ensure maintenance of local ecological integrity and downstream water quality.

4.1 Future Research

During the process of performing this research a number of related questions were generated. If such research was pursued in the future, it would serve to refine and expand upon the inferences suggested by the present study.

1. Fingerprinting of fine sediment stored in the streambed of the disturbed watershed. It may be argued that the discrepancies in LWD volumes between Star and Smith Creek may be the driving factor in the differences observed in fine bed sediment loading, not differences in past sediment introductions. If the fine sediment in the streambed of Smith Creek could be attributed to specific locations of past point sources (i.e., historic road crossings), it would help disentangle whether the LWD in Smith Creek enabled the retention of fine sediment produced by logging, or very efficiently trapped natural suspended solids thus enriching the streambed with fine sediments.

2. Determine if the increased productivity of Smith Creek is reflected in ecological changes in higher trophic levels. It has been reported that increases in periphyton growth can serve as a springboard, producing changes in ecosystem structure by favoring the success of particular species of invertebrates, ultimately reducing species diversity (Withers and Jarvie, 2008). Comparisons of invertebrate surveys in Star and Smith Creeks would enable better understanding of how changes in P dynamics can be propagated in headwater ecosystems. Furthermore, close linkages between invertebrate abundance and diversity with fish are well-documented. Future research linking post-disturbance sediment-P dynamics to higher trophic levels including fish would provide potentially an important link between forestry disturbance and fisheries management.

Tables and Figures

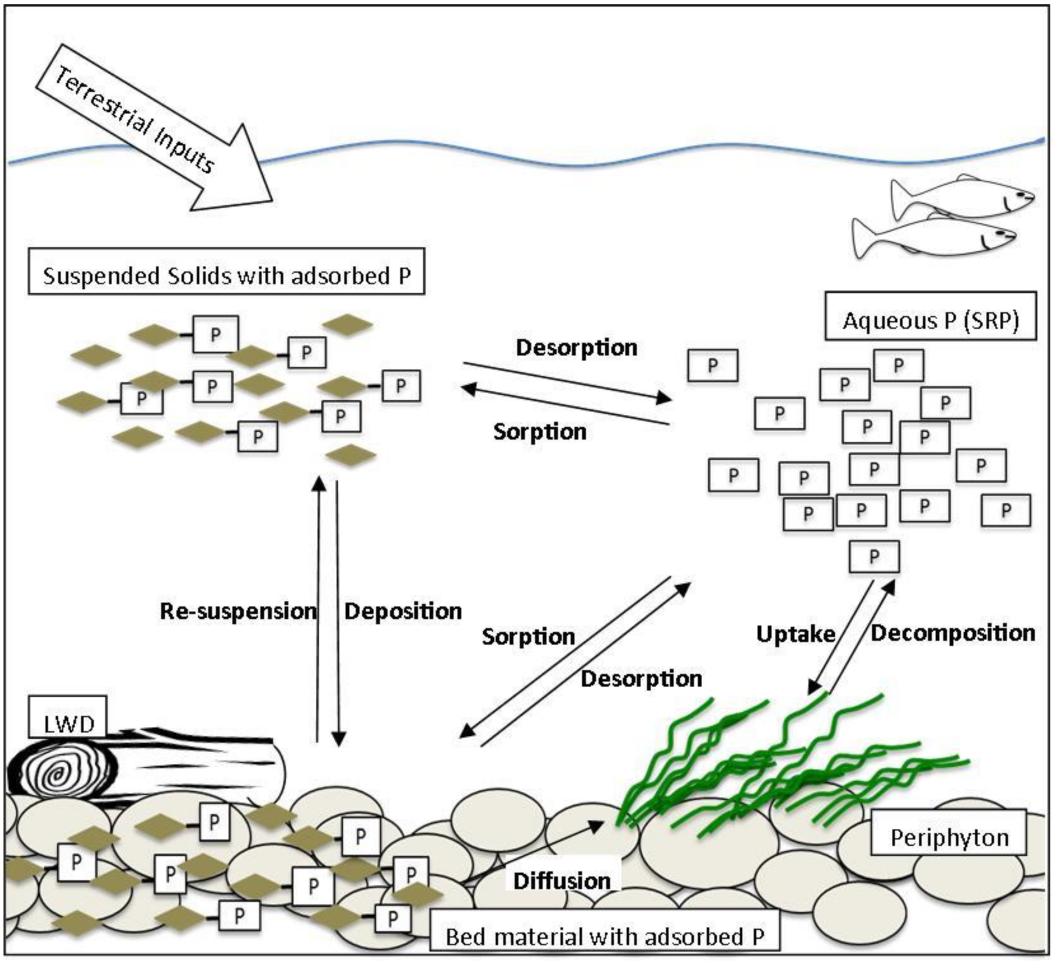

Figure 4-1. Conceptual diagram of stream sediment-phosphorus dynamics, illustrating various processes which govern concentrations of each pool of phosphorus.

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