Eastside Type N Riparian Effectiveness

Study Design

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Table of Contents

Abstract

In 2001 the Washington State Forest Practice Board approved a comprehensive set of new forest practice rules (WDNR, 2001) based on the *Forests & Fish Report* (WDNR, 1999). One of the goals of these rules is to protect water quality, including aquatic life,[1] in streams on non-federal forest lands in Washington State. To this end, the Cooperative Monitoring Evaluation and Research (CMER) committee has been tasked with performing research in support of an Adaptive Management Program (AMP). This research includes the study described herein to evaluate the effectiveness of the eastern Washington Type N riparian management rules (WAC 222-30-022).

This study addresses three questions related to the effects of timber harvest along non-fish-bearing (Type N) streams in eastern Washington:

1. What is the magnitude of change in water temperature, canopy closure, and stream cover of Type N channels in the first two years after harvest?
2. What is the magnitude of change in stream flow and suspended sediment export from the Type N basins in the first two years after harvest?
3. What is the relationship between observed changes in resource condition and forest management activity?

The study, intended to be a companion to Type N studies in western Washington, is focused on water quality. The design allows for additional questions to be addressed where doing so will provide scientifically defensible information relevant to adaptive management, and does not jeopardize our ability to satisfactorily address the original questions as was done in both westside Type N studies.

As with the westside Type N studies, this study incorporates a Multiple, Before-After/Control-Impact (MBACI) experimental design. Spatially blocked sets of treatment and reference sites will be identified and data collection will be conducted for at least two years pre-harvest and two years post-harvest, with a one-year harvest window. Longer-term monitoring will be required to determine the overall trajectory of the response and to capture a broader range of climate conditions and greater potential for episodic changes.

Small sample size, relative to observational studies, is an issue for most experimental studies and especially so for field-based studies like this. However, experimental studies are essential to testing the effectiveness of specific riparian prescriptions. Given our limited amount of basin-scale replication, the results of this study should not be viewed solely in isolation, but rather as a part of the larger body of research on forestry effects.

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[1] It is required that all indigenous fish and nonfish aquatic species be protected in waters of the state (WAC 173-201A-200(1).

Context and History

The Washington State Legislature and the Washington Forest Practices Board (WFPB) have adopted forest practices rules designed to maintain and restore salmonid populations and meet the requirements of the federal Clean Water Act. They also set up a formal science-based Adaptive Management Program (AMP) to provide science-based recommendations and technical information to assist the WFPB in determining when it is necessary or advisable to adjust rules and guidance to achieve resource objectives (WAC 222-12-045). The resource objectives are intended to ensure that forest practices, either individually or cumulatively, will not significantly impair the capacity of aquatic habitat to: a) support harvestable levels of salmonids; b) support the long-term viability of other covered species; or c) meet or exceed water quality standards (WAC 222-12-045(2)(a)) (Table 1).

The WFPB has empowered the Cooperative Monitoring Evaluation and Research committee (CMER) and the TFW Policy committee (Policy) to participate in the AMP (WAC 222-12-045(2)(b)). CMER has been tasked with completing a programmatic series of work tasks in support of the AMP; these tasks are laid out in an annual work plan that is approved by Policy. The Type N (non-fish-bearing) Riparian Effectiveness Program has been given a high research priority because of the many gaps in the scientific understanding of headwater streams, their aquatic resources, and the response of riparian stands, amphibians, water quality, and downstream fish populations to different riparian management strategies (CMER 2014). To date, no CMER studies have been conducted to determine whether the eastern Washington Type N riparian prescriptions described in WAC 222-30-022(2) are effective in meeting the resource objectives.

This study is part of the formal AMP for the Washington Forest Practices Habitat Conservation Plan (HCP) and the state forest practices rules and is one component in the Type N Riparian Effectiveness Program in the 2015 CMER work plan (CMER, 2014). The study was listed as a priority project in the 2012 Forest Practices Habitat Conservation Plan (FPHCP) Settlement Agreement between conservation community, the timber industry, and Washington State regarding Clean Water Act assurances. As an effectiveness monitoring project, the Eastside Type N Riparian Effectiveness Project (ENREP) is intended to determine whether "the rules produce forest conditions and processes that achieve resource objectives" and "whether less costly alternative prescriptions would be effective in producing conditions and processes that meet resource objectives or where more conservative prescriptions may be necessary".[2]

In 2013 a Technical Writing and Implementation Group (TWIG) was appointed to develop ENREP objectives, critical questions and study design options. In June of 2013 Policy approved the objectives and critical questions while asking the TWIG to consider additional issues including the effects of harvesting along seasonally dry stream reaches. The TWIG evaluated research alternatives and proposed a preliminary research effort (the "Forest Hydrology Study (FHS) Extension") to examine the spatial and temporal consistency of channel wetting and drying in a set of Type Np (N-perennial) basins known to include both perennial and seasonally-

[2] Forest Practices HCP Appendix N - Schedule L-1 Key Questions, Resource Objectives, and Performance Targets for Adaptive Management. http://www.dnr.wa.gov/programs-and-services/forest-practices/forest-practices-habitat-conservation-plan. December 2005.

dry reaches along Np channels. In the regulatory context, a Type Np channel can contain seasonally-dry reaches as long as there is upstream perennial flow connected to fish-bearing waters through a continuous channel. In the absence of connected upstream perennial flow, a seasonally dry reach is classified as Ns. A basin containing an Np channel is a Type Np basin. The ENREP research alternatives document was approved by Policy in November 2013.

Over the summer of 2014, the TWIG supervised data collection under the FHS Extension. In the original FHS survey, 21% of the Type Np channel network was dry by late summer. The FHS Extension focused on a subset of these same basins that were known to have seasonally dry reaches along Np channels. The FHS Extension showed that by later summer 35% of the total Np channel length was dry in this subset of Np channels with dry reaches. Approximately one-third of this was dry over the course of the entire summer while the remaining two-thirds dried incrementally. Additionally, the Extension showed that there was general consistency in the location of drying from year to year. The TWIG concluded that ENREP could feasibly include harvest treatments that targeted seasonally dry reaches. Policy was informed of the FHS Extension results in February 2015 and directed the TWIG to develop an ENREP study design.

In July 2015 the TWIG presented a draft study design to CMER that included a treatment focused on harvesting dry reaches in accordance with current FFR rule. That design met with resistance within CMER, and in August 2015 Policy retracted their previous guidance and directed the TWIG to develop treatments based on a structure recognized as best available science (BAS). In 2016, it was determined that the study would have two study components; one focused on perennially wet Np channels and one focused on seasonally wet Np channels.

In 2016, a study design for the perennial component of the project was completed and submitted for ISPR review. Shortly thereafter, in 2017, a TWIG was also convened for the dry component. In response to ISPR comments, and in response to dry TWIG concerns, field reconnaissance was conducted in summer 2017 to locate potential study basins in the Northern Rockies ecoregion. Three study basins were located across the region's precipitation gradient that show varying spatial and temporal extents of wet and dry reaches within each basin.

Given the comprehensive nature of ISPR comments about the scope of inference for the study, the inherent mix of wet and dry reaches that occurs in most Northern Rockies basins, and the experimental necessity and practical efficiencies of addressing wet and dry reaches simultaneously, the wet and dry TWIGs forwarded a recommendation to CMER to once again combine the two studies. This study plan now addresses a combined study.

This study will use BAS to address the question of whether riparian processes and functions provided by Type Np buffers are maintained at levels that meet FPHCP resource objectives. The FPHCP resource objectives listed in Schedule L-1 which are relevant to this study are listed in Table 1.

Table 1: FPHCP Schedule L-1 Resource Objectives.

Condition or process	**Functional objective**	**Performance Target**
Stream temperature	Provide cool water by maintaining shade, groundwater temperature, flow, and other watershed processes controlling stream temperature.[3]	Water quality standards (—current and anticipated in next triennial review)
Shade	Same as above.	Eastside high elevation: Shade available within 50 feet for 50% of stream length
LWD	Develop riparian conditions that provide complex habitats for recruiting large woody debris and litter.[4]	
Litterfall	Same as above.	Eastside Type N: at least 70% of recruitment available from within 50'.
Sediment	Provide clean water and substrate and maintain channel forming processes by minimizing to the maximum extent practicable, the delivery of management induced coarse and fine sediment to streams by protecting stream bank integrity, providing vegetative filtering, protecting unstable slopes, and preventing the routing of sediment to streams.	
Hydrology	Maintain surface and groundwater hydrologic regimes (magnitude, frequency, timing, and routing of stream flows) by disconnecting road drainage from the stream network, preventing increases in peak flows causing scour, and maintaining the hydrologic continuity of wetlands.	

3 Stream temperature is affected by the interaction of a complex set of factors, including shade, air temperature, pool depth and frequency, flow, and groundwater influences. These factors are addressed in resource objectives for other conditions or processes (e.g., hydrology, sediment, LWD) in addition to the targets selected for stream temperature.

4 Litter is defined to include leaves, needles, twigs, branches, and other organic debris that is recruited to aquatic systems and riparian forest floor.

Background

Discharge

Forest management can affect headwater stream hydrology (Moore and Wondzell 2005). The removal of forest canopy reduces evaporation from canopy interception and transpiration which changes the magnitude and timing of water delivery to the soil, soil moisture (Lewis *et al.* 2001; Keim and Skaugset 2003; Johnson *et al.* 2007) and snowmelt dynamics (Jones and Post, 2004; Marks *et al*., 1998). At the same time, forest roads can extend the surface channel network and intercept subsurface flow thereby increasing the surface water volume and the speed at which it enters the channel (Wemple *et al.* 1996; Wemple and Jones 2003).

Watershed studies from the Pacific Northwest and elsewhere have generally revealed that annual water yields increase in the short term then gradually decline following recovery from timber harvest, though the magnitude and timing of change is affected by a large number of factors (Bosch and Hewlett 1982; Stednick 1996; Jones and Post 2004; Brown *et al.* 2005; Moore and Wondzell 2005). In the Pacific Northwest, basins with 80% clearcut harvest have been shown to yield 483-615 mm more water per year in the Oregon Coast Range (Harr *et al.* 1975; Harris 1977; Harr 1983), 290-410 mm in the Oregon Cascades (Harr *et al.* 1979; Harr *et al.* 1982; Harr 1983, Harr 1986), and 360 mm on Vancouver Island (Heatherington 1982). Paired watershed studies have reported absolute daily streamflow changes of -2 to 8 mm/day following 100% forest removal with strong seasonal variations in the response including reductions in summer base flow (Hicks *et al.* 1991; Jones and Post, 2004). It is generally accepted that in rain-dominated areas, measurable annual runoff can increase by as much as 6 mm per year for each percent of the basin harvested above some threshold (Moore and Wondzell 2005). Following these initial flow increases immediately after harvest, decreased runoff below baseline values has been both observed and simulated with physically-based models decades after harvest due the recovery of vigorous young vegetation with higher evapotranspiration rates (Perry and Jones 2016; Du *et al*. 2016). These flow declines have also been noted to occur during dry summer low discharge conditions, both due to changes in riparian forest conditions (Hicks *et al*., 1991) and development of upslope forest plantations with the ability to transpire more water than older forests, resulting in decreased discharges for 10 to 50 or more years following harvest (Perry and Jones 2016). Flow alterations resulting from harvest may affect stream temperature dynamics by increasing or decreasing the potential of streams to buffer increased heat fluxes, especially during the period of maximum temperatures when discharges are low.

In contrast to rain-dominated or rain-on-snow-dominated systems, eastern Washington tends to be dominated by snow-with-rain or snowmelt systems (Reidy Liermann *et al.* 2012). Forest cover has a large effect on snow accumulation and melt dynamics. Snow accumulation in clearcuts is much larger than in intact forests due to reduced canopy snow interception losses from sublimation and evaporation, and increased turbulence over forest gaps. Melting is faster in clearcuts because snow is exposed to more shortwave radiation and turbulent (sensible and latent) energy transfer relative to snowcovers beneath forest canopies (Marks *et al.*, 1998), although snowfall magnitude, aspect, elevation, slope, windspeed, and clearcut size all affect the magnitude and timing of change (Berris and Harr 1987; Storck *et al.*, 2002; Varhola *et al*. 2010; Ellis *et al*. 2013). In snowmelt systems, water yields also increase following harvest, but the timing of hydrologic change is different. For example, a paired watershed study in the Mica Creek Experimental Watershed in northern Idaho determined that a 50% clearcut and a 50% partial cut increased yield by more than 270 mm and 140 mm per year, respectively; the largest increases came during the snow melt season in late spring (Hubbart *et al.* 2007). In rain-dominated systems, the relative effect of harvest typically decreases with event magnitude; but in snowmelt-dominated systems, peak flows are likely to increase in both frequency and magnitude (Green and Alila, 2012; Kuras *et al.* 2012) which may in turn influence channel disturbance and sediment transport dynamics, and hence alter interactions between flows, channel substrate, and energy fluxes that control temperature dynamics, especially during low flow conditions.

Thermal Processes

Summer stream temperatures vary both within stream reaches and across the landscape due to variation in physical, hydrologic, and climatic conditions that drive thermal processes (Poole and Berman, 2001; Johnson, 2004). Stream temperatures also vary as a result of differences in past riparian management practices and disturbance (Brown and Krygier, 1970; Brosofske *et al.* 1997; Wilkerson *et al.* 2006; Dent *et al.* 2008; Pollock *et al.* 2009; Kibler *et al.* 2013). For example, although stream heating is commonly associated with increased shortwave radiation loading due to decreased shade (Moore *et al.* 2005a), increased flows as noted above may serve to buffer temperature increases from increased energy gains. Increased peak flows, sedimentation, and/or wood recruitment following harvest may alter shading from riparian vegetation or large wood, channel dimensions, and/or hyporheic flow and storage which can all affect the degree stream heating. Stream heating is further complicated in small, spatially intermittent channels because hyporheic flow and understory shading can more readily influence stream temperature. Much of the scientific uncertainty in the effectiveness of riparian management practices is caused by the complexity of heating and cooling processes in streams as summarized above (Poole and Berman, 2001; Moore *et al.* 2005a).

In order to understand the components that drive stream temperature change, it is useful to consider a one-dimensional energy balance model for well mixed streams without inflows or outflows (per Caissie *et al.* 2007):

$$\frac{\partial T_w}{\partial t} + v\frac{\partial T_w}{\partial x} - \frac{1}{A}\frac{\partial}{\partial x}\left(AD_L\frac{\partial Tw}{\partial x}\right) = \frac{W}{\theta\rho A}H_n \tag{1}$$

where, T_w represents the water temperature, t is the time, v is the mean water velocity, x is the distance traveled, A is the cross-sectional area, W is the river width, D_L is the dispersion coefficient in the direction of flow, θ is the specific heat of water, ρ is the water density, H_n is the net heat flux, and ∂ preceding a parameter denotes its partial differential. For a given point in time, and excluding the effect of longitudinal dispersion, the equation can be simplified to:

$$\frac{dT_w}{dx} = \frac{1}{v}\frac{H_n}{\theta\rho D} \tag{2}$$

where D is depth and d preceding the parameter denotes its differential. The model indicates that as water travels through the reach its temperature changes as a function of H_n, which is the sum of the heat exchange processes including net shortwave radiation, net longwave radiation, sensible and latent heat exchange, and bed heat conduction (Johnson, 2004; Moore *et al.* 2005a). For a given net heat flux in equation 2, the change in water temperature is proportional to the surface residence time (x/v) and inversely proportional to water depth. This was observed by Hawkins and colleagues (1997) in the Sierra and Shasta-Trinity mountains.

Tributaries can also modulate stream temperatures. The effect of tributaries depends on the temperature difference between the inflow and receiving stream temperatures and on their relative contribution to discharge, which can be modeled according to a simple mixing equation (Moore *et al.* 2005a):

$$dT_w = \frac{Q_{trib}}{Q_{main}}\left(T_{trib} - T_{main}\right) \tag{3}$$

where, Q is discharge. This model can be extended to account for the effects of other discrete inflows (e.g., springs). When changes in flow are not discrete, the model becomes:

$$\frac{dT_w}{dx} = \frac{Q_{in}}{Q_{sw}}\left(T_{in} - T_{sw}\right) \tag{4}$$

where, Q_{in} / Q_{sw} is the functional relationship between the diffuse inflow and surface water flow along the reach, and T_{in} and T_{sw} are, respectively, the diffuse inflow and surface water flow temperatures. This equation can be used to model the effects of diffuse groundwater inputs and direct precipitation to the channel.

Hyporheic exchange occurs along a subsurface flow path where surface water mixes with subsurface water and then returns to the stream (Gooseff *et al.* 2003). Water in the hyporheic zone is sheltered from exchanges along the air-water interface so it directly affects the spatial/temporal distribution, rather than the magnitude, of stream heat energy, and is often viewed as an important temperature buffer (Poole and Berman, 2001). Hyporheic water temperature (T_{hyp}) can be modeled in terms of the residence time (Gooseff *et al.* 2003):

$$\frac{\partial T_{hyp}}{dt} = \int_0^t \frac{\partial T_w(t-\tau)}{\partial t} g^*(t)dt \tag{5}$$

where, τ is a lag time and $g^*(t)$ is a function of the distribution of exchange rates.

Thermal processes can be accounted for using an additive effects model (Moore *et al.* 2005a) in which the rate of change in water temperature over a reach length (x) is a function of the incoming water temperature, net energy exchange across the air-water surface, and change in temperature associated with inflows, including the desynchronizing effect of hyporheic exchange (Figure 1).

Riparian timber harvest can affect stream temperatures through alterations of shade, sediment and logging debris recruitment, hyporheic flow, and discharge. In forested environments with perennial streams, shade provided by riparian vegetation is commonly the single most important variable influencing summer stream temperature (Brown, 1969; Johnson and Jones, 2000; Danehy *et al.* 2005; Groom *et al.* 2011) although channel morphology can exert a distinct influence, especially at small scales. Harvest of riparian trees can reduce canopy cover and shade, thereby increasing the amount of shortwave radiation reaching the stream (Brazier and Brown, 1973; Moore *et al.* 2005b). The retention of riparian forest buffers ameliorates some of the loss of shade from adjacent timber harvest, though effectiveness of the buffer varies with buffer width, tree height, and tree density (DeWalle, 2010). In very small headwater streams, recovery of low-stature herbaceous vegetation following complete harvest of the riparian zone has been noted to produce high shade levels in the years immediately following harvest (Klos and Link, 2018).

Harvest adjacent to the stream can result in delivery of logging debris to the channel which can provide short-term stream cover and temporarily offset some of the loss of forest canopy shade (Jackson *et al.* 2001). Logging debris also increases hyporheic exchange capacity through the creation of debris jams and channel steps (Wondzell, 2006), but can also increase retention time and hence the potential for heating. Changes in discharge can also affect temperature by diluting or concentrating the incoming heat load into a larger or smaller volume of water (Poole and Berman, 2001).

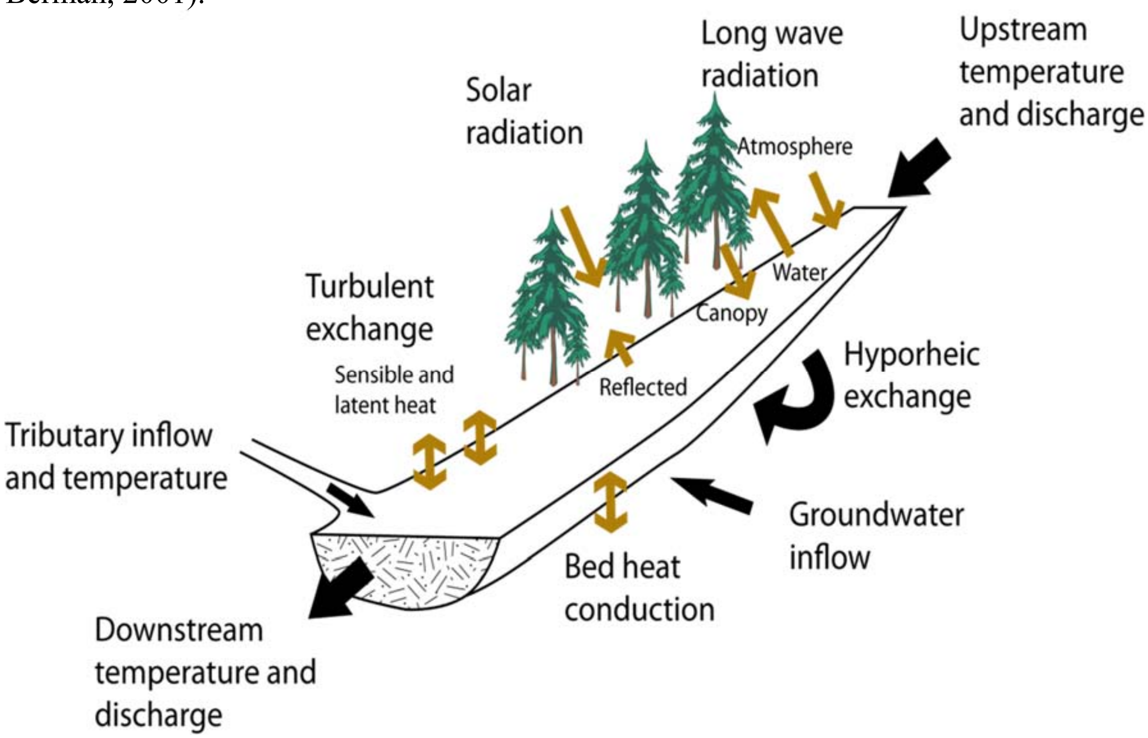

Figure 1: Conceptual model of stream heating and cooling processes (Moore *et al.* 2005a). Black arrows indicate energy fluxes associated with water exchanges.

Studies of the effects of various riparian buffer configurations on stream temperature on fish-bearing streams in the Pacific Northwest have reported increases of 2.5 to 5.0 °C following clearcut harvest (Moore *et al.* 2005a). A well-replicated study of small and medium-size streams (<283 L/s mean annual flow) reported responses ranging from −0.9 to 2.5 °C for two different buffer strategies in the Oregon Coast Range that included 20 foot no-cut buffers with thinning out to 69 ft (Groom *et al.* 2011). The magnitude of change was most strongly correlated with stream shade levels, though temperature change did vary with reach length and gradient. In contrast, Janisch *et al.* (2012) reported temperature changes ranging from −0.1 to 3.6 °C with no correlation to stream shade in small, non-fish-bearing headwater streams in western Washington. Janisch and colleagues suggested that very small (watershed area ≤8.5 ha) headwater streams with spatially intermittent surface flow differ from larger streams with spatially continuous flow throughout the stream network because hyporheic flow and understory shading can more readily influence stream temperature response in the former. Similarly, Gravelle and Link (2007) reported stream temperature increases of up to 3.6°C in clearcut perennial nonfish-bearing reaches, but increases of only 0.8°C in a similar reach with ~125 meters of seasonally dry channel, and non-significant minor temperature decreases in downstream buffered fish-bearing streams. Gravelle and Link (2007) hypothesized that cooler than expected stream temperature in the downstream reaches may have been due to increased midsummer streamflows resulting from evapotranspiration declines following timber harvest. Thus, stream temperature response is not only affected by heat exchange across the air-water surface interface, but is also a function of channel configuration, and hydrologic and basin physical conditions (Johnson, 2004; Hannah *et al*. 2004; Gravelle and Link 2007; Subehi *et al.* 2009; Leach and Moore, 2010).

Preliminary results from the CMER Type N Experimental Buffer Treatment Study in Competent Lithologies ("Hard Rock study") are largely consistent with previous research; even small (less than 10%) decreases in shade caused measurable increases in summer maximum stream temperature. Following harvest, shade decreased in all three buffer treatments with the greatest reduction in shade observed in the 0% treatment (no buffer) and the least in the 100% treatment (entire reach buffered). Increased maximum, minimum, and diel range of daily stream temperatures were observed at nearly all sites and the magnitude of temperature response was correlated with several metrics of post-harvest shade. Average changes in maximum stream temperature at the bottom of the study area in the first two years post-harvest were 0.9 and 0.6°C in the 100%, 1.4 and 1.0 °C in the FP (50% of the reach length buffered), and 3.1 and 2.7 °C in the 0% treatment (i.e., no trees in the riparian management zone (RMZ).

Although large (relative to surface flow) inputs of groundwater and hyporheic exchange can buffer the response of stream temperature to energy gains (Arrigoni et al., 2008, Johnson and Jones 2000; Story *et al.* 2003; Wondzell 2006), subsurface flow through dry reaches was not observed to have a consistent effect on downstream water temperature in the Hard Rock study. Models accounting for the influence of solar radiation on water temperatures are similarly mixed in their representation of subsurface flow and energy transfer processes. Stonestrom and Constantz (2003) outline a generalized model for tracing the movement of groundwater near streams that is based on heat exchange at the groundwater-surface water interface. A major assumption underlying this model is that groundwater is unaffected by solar radiation. However, bed heat conduction (as shown in Moore *et al.* 2005a) may transfer energy to subsurface flow along dry reaches. Kurylyk *et al.* (2015) develop a system of equations for accounting for subsurface water warming resulting from surface perturbations such as timber harvesting. They demonstrated the theoretical sensitivity of relatively shallow groundwater less than 10 m below the surface to warming, and emphasize that groundwater warming is particularly important for very small streams where subsurface energy fluxes may comprise a large portion of the stream energy budget.

In summary, timber harvest can alter a complex variety of interrelated processes that affect stream temperature dynamics. Specifically, harvest can decrease stream shade that may increase water temperatures. However, increased flow due to dramatically decreased evapotranspiration following harvest may serve to buffer the effects of increased heat fluxes which may have a reverse effect after vegetation recovers, evapotranspiration increases, and flows decline below preharvest conditions. Harvest can also increase the potential for both increased sedimentation and peakflows which can either individually, or in tandem alter channel morphology, wood recruitment, and/or riparian vegetation. Altered channel conditions may also affect temperature dynamics by changing the amount of water surface exposed to shortwave radiation loading, hyporheic flow and residence times, and shading by nearby bank vegetation and large wood. To adequately understand how harvest affects stream temperature, this study must employ a design and methodology to quantify these complex and interrelated conditions that directly and indirectly control stream temperature.

Wood and Sediment

Wood and sediment are natural components of headwater streams and timber harvest can affect the recruitment, storage, and transport of both wood and sediment that can in turn affect hyporheic storage and flow dynamics.

Timber harvest can affect wood recruitment by changing the number of trees with recruitment potential (Beechie 1998) or the frequency of mortality-inducing events including windthrow (Grizzel and Wolff, 1998; Jackson *et al*. 2001). The functions provided by wood depend upon the characteristics of both the wood (e.g. size and position relative to the channel) and the channel (e.g. width, gradient and composition of bed material) (Bilby and Ward, 1989; Gomi *et al*. 2001, Maxa 2009). While large woody debris (LWD, greater than 10 cm diameter) is the focus of most studies, the functional role of small woody debris (SWD, 0 cm to 10 cm diameter) tends to be greater in headwater streams (Gomi *et al*. 2001, Maxa 2009).

Using forest fuel models for intermountain forests (Anderson 1982), SWD loads in eastside forests typically range from about 5 to 10 tons/acre and increase post-harvest from slash and windthrow. Comparable levels would be expected in streams on an aerial basis. SWD is more abundant in headwater channels, because stream power is typically too low to transport it (Bilby and Ward 1989, Maxa 2009). In these cases, SWD can be a major step-forming element (Jackson and Sturm, 2002). In contrast, many LWD pieces are suspended above small streams, delaying their impact on the channel. In the long term, however, small wood appears to be less persistent than large wood because of more rapid decay and susceptibility to downstream transport during extreme events (Wallace *et al*. 2000, Scherer 2008).

In small forested basins, most sediment is supplied to channels via mass wasting and bank erosion. In harvested basins, ditches, roads, and skid track surfaces can also become major contributors (Hassan *et al*. 2005).

Sediment routing through a basin is complex. Changes in bedforms, large wood, and other channel features alter hydraulic resistance, shear stress, and in-channel sediment storage

(Buffington and Montgomery, 1999; Jackson *et al*. 2001; Kaufmann *et al*. 2009). Therefore, changes in sediment input may not be synchronized with changes in sediment export. In a classic example, Roberts and Church (1986) found that logging increased streambank erosion and landsliding in several watersheds in the Queen Charlotte Islands. However, the bed sediment residence time also increased by as much as ten-fold because large quantities of sediment were stored in in-channel sediment wedges, thereby delaying routing of this sediment downstream.

In a review paper, Dunne and others (2001) noted that one of the main problems with measuring and predicting the influence in forest management on sediment transport is that the monitoring period is typically too short to sample the variability of natural and disturbed hydrologic regimes. Episodic sediment transport is driven by episodic climatic events, and the likelihood of a large event is low during a study period of only a few years. Episodic mass wasting can deliver large quantities of sediment to stream channels in a short period of time and can mask treatment effects. Grant and Wolff (1991) documented 30 years of suspended and bed load sediment mobility from three small basins with different road building and forest harvest treatments. They estimated that approximately 85% of the total 30-year load was transported in a single event in which a series of debris flows scoured the channel to bedrock. Furthermore, the maximum post-harvest decrease in hillslope stability is thought to be delayed for several years as root strength declines (Sidle and Ochiai, 2006). For these reasons, the hydrologic processes most relevant to this study are expected to be those associated with increased soil moisture and runoff resulting from decreased forest interception and evapotranspiration. These include erosion of stream beds and banks and potential increases in mass soil movements, surface erosion, and soil creep. Blow down of buffers and road-related erosion may also deliver sediment to streams, though increased short-term recruitment of woody debris may result in much of this material being stored in basin over the course of the study.

Benthic Invertebrates and Other Aquatic Life in Headwater Streams

Benthic invertebrates are a critical component of the food web in all stream ecosystems (Hawkins *et al.* 1982; Wallace *et al.* 1997; Meyer *et al.* 2007) and are potentially sensitive to several environmental alterations that can be caused by forest practices (Hawkins *et al.* 1982, 1983, 1997, 2000, 2015; Davies and Nelson 1994; Richardson and Danehy 2007). Alteration of the riparian corridor can influence the amount of light, terrestrial-derived litter, water, sediments, and nutrients reaching the channel (Hawkins *et al.* 1983; Gregory *et al.* 1991; Davies and Nelson 1994; Kiffney *et al*. 2003). Excessive nutrients and fine sediments, altered thermal regimes, and altered hydrologic regimes are among the most pervasive sources of impairment to both aquatic life and water quality in the United States (Paulsen *et al.* 2008; Konrad *et al.* 2008; Carlisle *et al.* 2013).

Previous studies regarding the effects of forest practices on invertebrates and other aquatic species have produced mixed results (e.g., Newbold *et al.* 1980; Hawkins *et al.* 1982, 1983, 2000; Carlson *et al.* 1990; Kiffney *et al.* 2003; Quinn *et al*. 2004; Hernandez *et al.* 2005; Reid *et al.* 2010). Inconclusive results can easily arise due to the inherently high sampling variability associated with physically heterogeneous stream beds (Morin 1985; Norris and Georges 1993). Headwater streams can be especially difficult to sample because traditional sampling devices are too large and thus inefficient. However, choice of the specific response variable used as the ecological endpoint can also influence results. Many of the previous studies examining response of benthic invertebrates to forest practices used functional feeding groups or other coarse descriptors of invertebrate condition as response variables. But, functional feeding groups appear to be relatively insensitive to stream changes associated with forest practices and other human-caused disturbances. Functional feeding groups seldom discriminate between reference-quality and degraded streams and are thus not used as metrics in the indices that state and federal water quality agencies use for assessment purposes (e.g., Herbst and Silldorff 2006; Hering *et al.* 2006; Stoddard *et al*. 2008). Endpoints that assess biodiversity loss or changes in species composition tend to be more sensitive (Hawkins 2000, 2006). Moreover, analyses that test for the sensitivity of species to different types of stressors of concern (e.g., temperature, fine sediment, altered hydrology, altered food sources) can help identify which specific environmental alterations are most strongly affecting communities and are thus of greatest concern to managers (e.g., Huff *et al.* 2008; Relyea *et al.* 2012; Glendell *et al.* 2014; Murphey *et al.* 2015; Hubler *et al.* 2016; Turley *et al.* 2016).

Project Description

Purpose

The purpose of this project is to determine the extent to which the prescriptions found in the eastside Type N Riparian Prescriptions Rule Group are effectively achieving performance targets, particularly as they apply to sediment and stream temperature and their effects on aquatic life. As an effectiveness monitoring project, it is also expected to inform whether the current rule is effective in meeting these targets.

Objective

The objective is to inform Policy of the quantitative changes in FPHCP covered resources, water quality and aquatic life coincident with forest harvest activities in eastern Washington, and to determine if and how observed changes are related to activities associated with forest management.

Critical Questions

1. What is the magnitude of change in water temperature, canopy closure, and stream cover of Type Np channels in the first two years after harvest?
2. What is the magnitude of change in stream flow and suspended sediment export from the Type Np basin in the first two years after harvest?
3. What is the relationship between observed changes in resource condition and forest management activity?

Research Approach

This study will use a hierarchical design that incorporates a blocked Multiple Before-After/Control-Impact (MBACI) design with reaches nested within basins to quantify the magnitude of change that occurs as a result of harvest activity. The MBACI design, which is replicated in space and time, controls for natural variability throughout the pre- and post-treatment periods and allows us to estimate the likelihood that observed effects are related to anthropogenic activity (Underwood, 1994; Downes *et al*. 2002).

The study is designed with two-years of pre-treatment monitoring and at least two-years of post-treatment monitoring. Two-years is not enough time to capture the full range of effects, especially those that are likely to be episodic. Although the degree of inference will be limited by the relatively short pre and post-treatment phases, this has been shown to be adequate for quantifying the initial changes associated with harvest (e.g., McIntyre *et al.* 2017). Longer-term monitoring will be required to determine the overall trajectory of the response and to capture a broader range of climate conditions and greater potential for episodic changes with less frequent recurrence intervals.

The highest-level experimental unit is the Type Np basin. The Type Np basin is the drainage area above the uppermost point of fish habitat in a basin with perennial flow and it is an appropriate unit of study because the eastside Type N riparian prescriptions are designed to be applied over an entire basin with different requirements for different portions of the drainage network. By experimenting at the basin scale, we can examine reach-scale effects within the drainage basin, as well as cumulative exports to downstream fish-bearing waters. Type Np basins were used as the experimental units in two CMER Type N studies in western Washington (i.e., Type N Experimental Hard Rock and Soft Rock) and an eastside Type N basin-scale study should allow for direct comparison of study results.

The basin-scale treatment that will be applied represents the most common application of the Type Np rule for eastern Washington (WAC 222-30-022(2)). The rule allows for harvest to within 50 feet of the outer edge of the bankfull width of the Np stream and requires that equipment be limited from working in a 30-foot wide zone measured horizontally from the outer edge of bankfull width of a Np or Ns streams. If basal area requirements are met, it is permissible to harvest within the RMZ under a clearcut or partial cut strategy, though FPA surveys suggest that these RMZ harvest strategies are rarely used. As a result, the default application is a 50 foot buffer along 100% of the Type Np stream.

Under this treatment, the Np basin would be subject to harvest of as many trees as practical for efficient economic operations, and retaining trees as silviculturally necessary for successful regeneration, to within 50 feet from the bankfull edge of the stream on both sides. Such practices are representative of those commonly used in state and private timber management in the region, thus supporting a meaningful evaluation. In sites with seasonally dry Np reaches, some, but not all, of the RMZ associated with seasonally dry Np reaches will be harvested under the clearcut option or an alternative plan. Such practice follows the rules and practical and silvicultural considerations, and it is also representative of state and private timber management. When it occurs, it will allow us to evaluate the effectiveness of buffering or not buffering seasonally dry Np reaches.

Although harvest activities will be completed at the basin scale, monitoring will be conducted at the reach scale. Even under single a riparian prescription, implementation in the field will yield a range of shade loss across the landscape not only because of the practical and silvicultural considerations note above, but also because of the range of pre-harvest conditions (e.g., Groom *et al*. 2011, Ehinger 2015). We emphasize shade loss in the experimental layout because shade is a FPHCP resource that changes in direct response to forest practices, and change in shade is the primary driver of stream temperature change in response to forest practices. Because we will monitor at the reach-level, we will be able to consider reach-level covariates in our statistical analyses, evaluating the influence of several parameters on shade loss, not solely those expressed by basin-wide descriptors. A similar scope of analysis will be employed for the other response variables.

Shade is affected by buffer width, canopy density, and tree height, and can vary with species composition (Groom *et al*. 2011; Teply *et al*. 2014) and crown geometry (Seyednasrollah and Kumar, 2013). Even within a single prescription and forest type, differences in tree position, canopy, understory vegetation, and local topography will create differences in shade loss. Hemiview data collected in western Washington for the Type N Experimental Buffer Treatment

Project in Hard Rock Lithologies (Ehinger, 2015) showed that effective shade loss in a 50', no cut buffer averaged 14%, but at-a-station shade loss ranged from -0.1% to 52%.

Resilience of stands to management activities is heavily dependent on stocking (Teply *et al.* 2014). Where initial stocking is relatively high, for instance in moist, high elevation, mixed conifer stands, we expect a distribution shifted towards lower shade loss under management. Conversely, where initial stocking is relatively low, for instance in dry, low elevation ponderosa pine stands, we expect a distribution with greater riparian shade loss.

With only a single treatment, there will be no direct inference to alternative prescriptions. However, the results are expected to add to a growing body of literature from other locations and prescriptions and will fill a clear knowledge gap with respect to non-fish-bearing streams spanning a precipitation gradient in eastern Washington. From the information gained in this study, our understanding of process mechanisms, and the greater body of existing knowledge, we should be able to infer the likely direction and/or magnitude of change expected under other conditions, and to critically assess the results of this study in the context of similar paired-basin (e.g.: Mica Creek (Hubbart et al., 2007), Alsea (Bladon et al., 2016), Trask (Bywater-Reyes, 2017), and Hardrock studies) and reach-scale studies (e.g.: Idaho Stream Shade Rule study (Keefe et al., 2015) in the region.

Site Selection

The target population is forest land in eastern Washington that is likely to be harvested under WAC-222-30-022(2). This includes all state and private forest lands in eastern Washington that do not have their own habitat conservation plan. Approximately 44% of the target population area is in the Northern Rockies ecoregion, 22% is in the Eastern Cascades slopes and Foothills, 15% is in the North Cascades with the rest split among the remaining ecoregions. A review of Forest Practices Applications (FPA) from 1994 to October 2017 showed that most of the acreage harvested in eastern Washington over the period came from the Northern Rockies ecoregion (66%) followed by the Eastern Cascades Slopes and Foothills (14%, Figure 2).

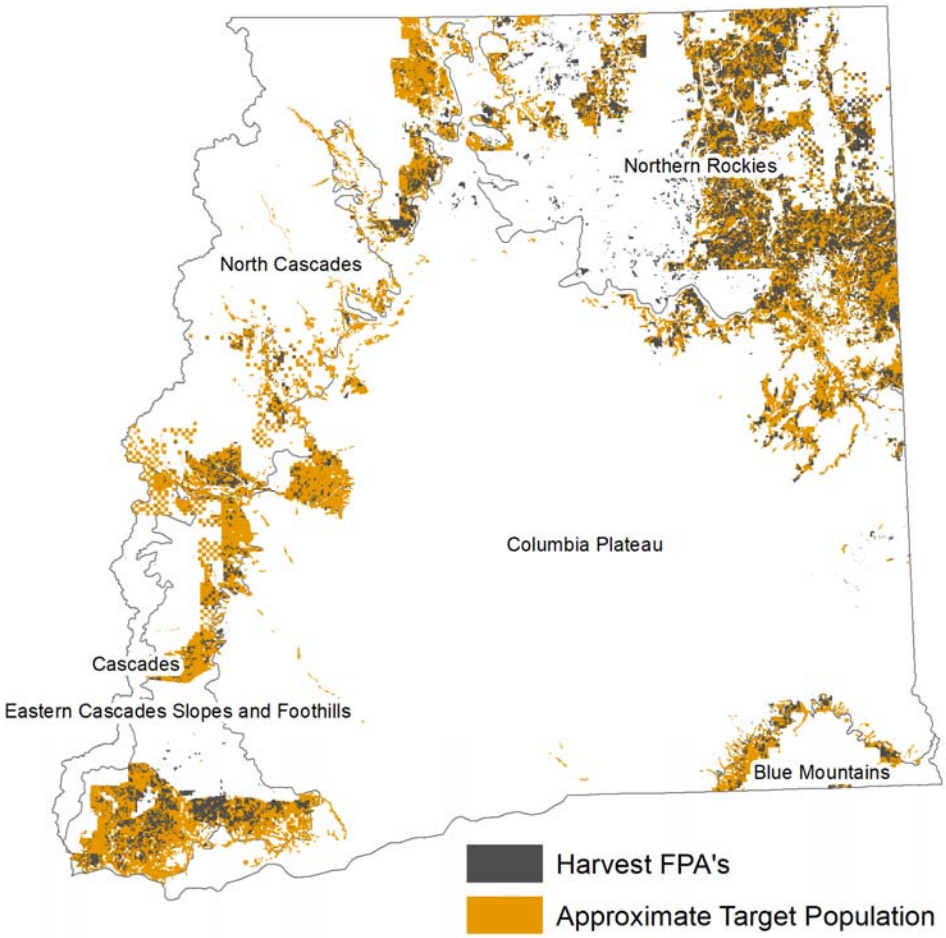

Figure 2: Approximate target population. The majority of Forest Practices Application (FPA's) of interest are in the Northern Rockies Ecoregion.

In the spring of 2017, land managers in the Northern Rockies ecoregion were contacted about the study. Using data regarding the extent of land managed for timber production by the state and large industrial landowners, approximately 77,000 acres of state and private land were office screened using aerial photographs and mapped Type N basins. The office screening identified 121 Type N basins that appeared to meet study criteria (i.e., entire Type N basin that appeared to be composed of mature harvestable timber). Site maps were sent to land managers with a question about whether the sites could be harvested in 2020 as part of this study. Private land managers identified 26 for possible inclusion.

Field reconnaissance of the 26 revealed three adjacent pairs (n=6) that met all study criteria. Approximately 1/3 of the remaining sites were rejected because they did not contain a Type Np stream. Of the remaining sites, three were rejected because they had profound natural anthropogenic or natural disturbance and it was assumed that a reference could not be found. The remainder were not included for logistical reasons (e.g., multiple small landowners in the basin) or because a suitable reference could not be found. While the rejection of sites can create a biased sample, the six sites selected for inclusion appear to be representative of the population of interest, with the exception sites with ongoing or major recent disturbance.

The three site pairs identified for inclusion of the study span a gradient of precipitation and channel wetness. The first pair of basins (Springdale) is approximately 27 miles northwest of Spokane, average 196 acres in size, are east facing and are dominated by second growth ponderosa pine. Both basins are isolated (or hanging) Np with no channel connection to downstream fish-bearing waters. They average about 2800 feet of channel that is mostly dry by late summer. The second pair of basins (BlueGrouse) is approximately 34 miles due north of Spokane, average 84 acres, are east facing and composed of second-growth mixed conifers. The northern treatment basin has about 2400 feet of channel, half of which is likely to be dry in late summer while the reference is flowing through most of its length. The third set of basins (Tripps) is approximately 24 miles northeast of Spokane (just east of Mount Spokane), averages 111 acres, are north facing and are dominated by Douglas fir and western redcedar. These sites each have over 4000' of channel that is largely perennial (Table 2; Figure 3).

A GIS exercise was conducted to evaluate how representative these sites were in terms of climate and hydrology compared to the population of interest. For climate (temperature and precipitation) we used the 1981-2010 Annual PRISM normals (PRISM Climate Group, 2012) and for hydrology we used the Hydrologic Landscape Characterization framework developed by Leibowitz and colleagues (2016) for the Pacific Northwest. The 2006 CMER lands layer was used to restrict the analysis to the domain of interest.

The exercise revealed that the three sites (Springdale, BlueGrouse, and Tripps) are very representative of annual minimum and maximum temperatures in the population of interest and span a gradient of precipitation with annual precipitation normals of 485, 685, and 1024mm/yr, respectively, which is very close to the 25th percentile, mean and 90th percentile of the annual precipitation in the target population (Figure 4).

Table 2: Basin characteristics for the three sites north of Spokane.

	Springdale		BlueGrouse		Tripps	
	North	South	North	South	West	East
Latitude	47.99706	47.99187	48.17171	48.16838	47.9168	47.91644
Longitude	-117.7131	-117.7178	-117.3847	-117.3829	-117.0565	-117.0481
Basin area (acres)	223	169	65	103	126	97
Elevation range (min-max, feet)	1884-2922	1880-2688	2857-4069	2832-4088	3032-3970	2866-3967
Approx. Channel length (ft)	3100	2500	2400	2100	6100	4200
Precipitation (mm)*	484	485	685	685	1022	1028
Temperature Min (deg. C)*	1.9	2	1.1	1.1	2.1	2.2
Temperature Max (deg. C)*	14.0	14.3	12.1	12.1	11.9	11.8

*30 year annual PRISM climate normals (1981-2010).

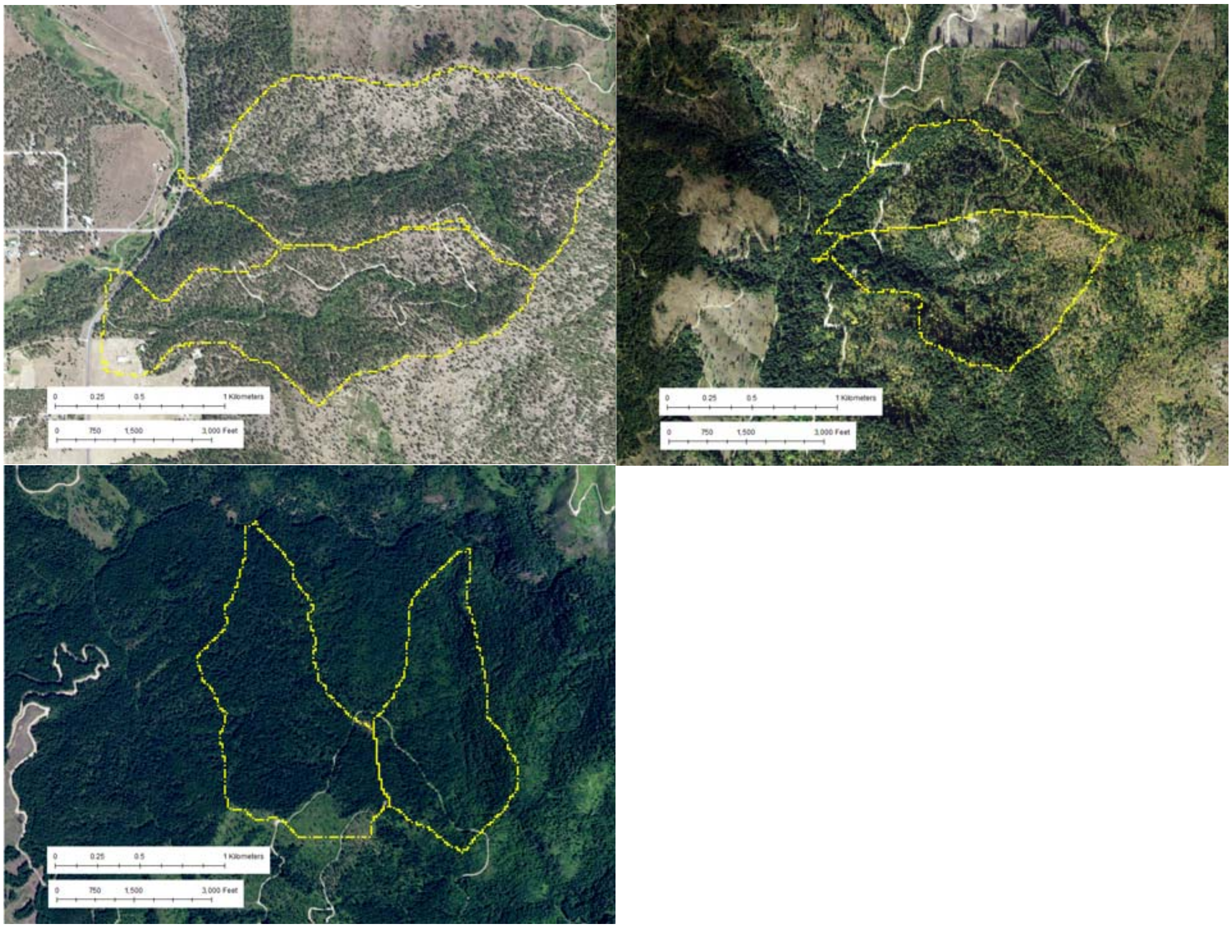

Figure 3: Basin pairs at Springdale (top left), Blue Grouse (top right), and Tripps (bottom).

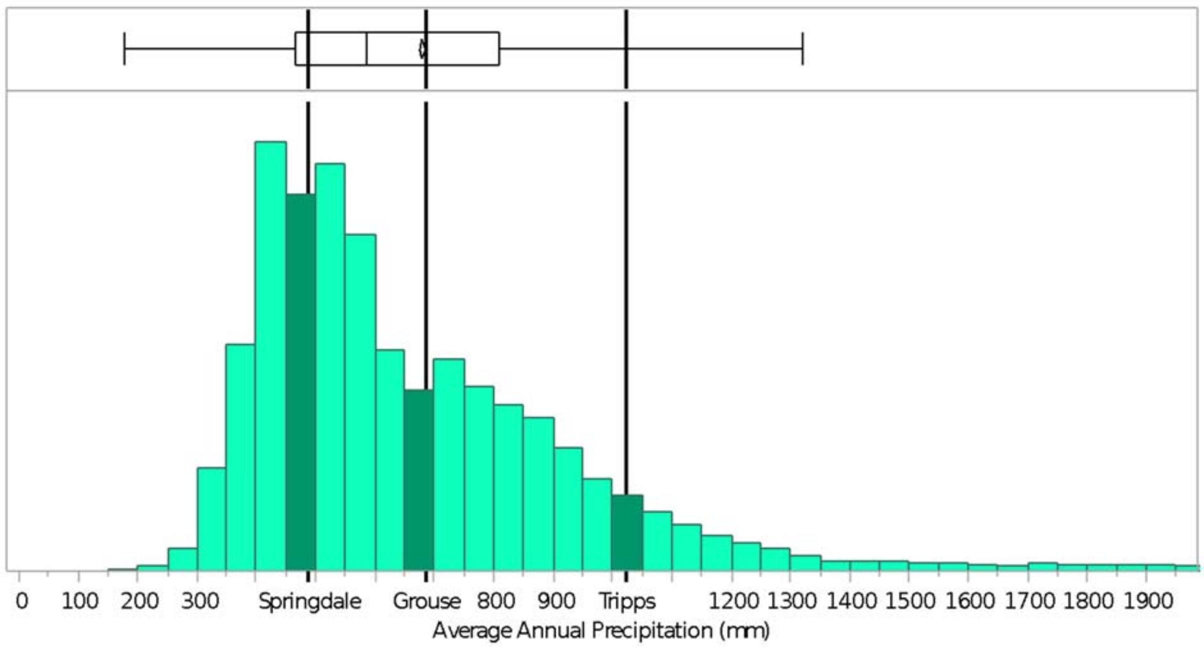

Figure 4: Annual precipitation normals for the Springdale, BlueGrouse, and Tripps sites compared with the population of interest.

The Hydrologic Landscape Characterization also includes a climate class and the result is very similar, with Springdale being called Dry, BlueGrouse Moist, and Tripps Wet; with Dry, Moist and Wet representing 36%, 32%, and 21% of the area of interest, respectively (Table 3). In terms of climate and terrain, the three sites are well-distributed. The sites are a little bit weighted towards having snowmelt dominated spring runoff and all the sites are classified as having low aquifer permeability. In eastern Washington, most of the high permeability basins are located in the Eastern Cascades Slopes and Foothills ecoregion.

CMER staff are currently working with Washington State Lands foresters to identify additional site pairs spanning a similar hydrologic gradient but located in the eastern Washington Cascades and Foothills. Study authors recommend proceeding with the existing three basin pairs and up to three additional basin pairs located in the eastern Cascades region if additional sites can be identified. If possible, additional sites should have stream orientations that differ from the existing three sites given the importance of basin orientation on solar exposure, evaporative demand, and prevailing wind stress.

Table 3: Distribution of target population land area and paired sites by the Hydrologic Landscape Characterization for the Pacific Northwest (Liebowitz et al., 2016).

Class	Categories	Proportion of population area	Sites
Climate	Very wet	3%	
	Wet	21%	Tripps
	Moist	32%	BlueGrouse
	Dry	36%	Springdale
	Semiarid	8%	
	Arid	-	
Seasonality	Fall or winter	62%	Springdale
	Spring	38%	BlueGrouse, Tripps
Aquifer Permeability	High	40%	
	Low	60%	Springdale, BG, Tripps
Terrain	Mountain	79%	BlueGrouse, Tripps
	Transitional	21%	Springdale
	Flat	-	
Soil Permeability	High	-	
	Low	100%	Springdale, BG, Tripps

Research Questions

1. **How does basin harvest affect the magnitude and timing of surface water discharge from the Type Np basin, and the amount and timing of dry reaches (if applicable)?**

Forest harvest typically increases water yield and can affect the magnitude and timing of stream discharge, but the hydrologic response to harvest is likely to differ between rain-dominated and snowmelt-dominated systems, among basins with differing biophysical characteristics, and among climatic events of differing characteristics. We will measure discharge at the basin outlet, and employ a MBACI analytical approach to quantify changes in yield, magnitude, frequency, and timing of streamflow. The spatial extent of dry and flowing reaches will be determined near the start of the late-spring drying period and end of the summer dry periods (McNamara *et al.*, 2005) by direct observations. We will compare and contrast the results of this study with the results of the two Type N studies in western Washington to better understand differences between rain and snow-melt dominated systems in the hydrologic response to forest practices. Understanding how harvest affects stream hydrology is critical to our understanding of forestry effects on sediment transport, water temperature, and nutrient export.

2. **How does harvest of Type Np stream basins affect the number, size, total volume, and distribution of woody debris within study reaches?**

The importance of LWD in forest stream ecosystems is widely known, and the contribution of SWD, particularly in headwater streams, is becoming increasingly recognized. We will quantify the short-term effects of specific forest harvest prescriptions on the frequency, size, volume, and distribution of in-channel LWD and on the volume and biomass of SWD in the study basins.

3. **Is there a relationship between the frequency and magnitude of sediment delivery pathways from forest units to streams and suspended sediment export (SSE), and how do pathways differ between treatment and reference basins before and after treatment?**

Sediment is a key component of channel forming processes, but excess sediment can impair water quality. As a result, the Forest Practices Rules were designed to minimize the delivery of management-induced coarse and fine sediment to streams. We will identify sediment delivery pathways and will explore whether sediment pathway frequency, magnitude, or type appear to change in response to treatment application. We will continuously measure turbidity and will sample surface water suspended sediment concentrations at the Type Np basin outlet. Using this data, we will estimate changes in SSE and relate them to any observed changes in source contribution.

4. **How does water temperature change within the Type Np basin, and can we relate temperature change to change in shade?**

Previous research indicates that substantial variability in stream temperature response may exist within a Type N stream network. We will examine the rate of reach scale surface water temperature change among a suite of discrete points within the Np stream network. We will use a range of metrics (including daily minimum, daily maximum, and diel fluctuation) to infer reach scale effects associated with different riparian management strategies and change in shade.

5. **How does harvest affect water temperature at the basin outlet and in downstream fish bearing waters?**

One objective of the Type N riparian prescriptions is the prevention of increased water temperature entering and flowing through downstream Type F waters. The analysis will evaluate changes in several metrics related to surface water temperature, including the 7-day average of the maximum daily water temperature (a metric used in the water quality standards) at the outlet of the Type Np basin. Temperature sensors will be installed in downstream fish-bearing waters where treatment effects can be isolated (e.g., there are no tributaries from other basins flowing into the reach). In these instances, shade measurements will also be extended downstream.

6. **How do canopy closure, effective shade, stream cover, hydrologic condition, periphyton (both as chlorophyll-a and ash-free dry mass), and detritus standing crops change in response to the different buffer treatments?**

Forest Practices Rules recognize the role that riparian forests play in providing shade, detritus, and woody debris to streams as well as the role of forests on the hydrologic cycle. We will determine how canopy closure, effective shade, stream cover, hydrologic condition, abundance of periphyton, and amount of streambed detritus differ among buffer treatments in the two years after harvest.

7. **How do benthic invertebrate assemblages and amphibians respond to the different buffer treatments?**

Benthic invertebrates and amphibians are sensitive to alterations in flow, temperature, nutrients, sediment, and food sources associated with forest practices. We will determine if benthic invertebrate assemblages differ among buffer treatments in terms of three types of measures: (1) estimates of abundance and food web structure that are relevant to higher trophic levels (e.g., amphibians, fish, and birds), (2) an index of biodiversity loss comparable to indices of taxonomic completeness that are used in Washington and elsewhere to determine if streams are meeting aquatic life use designations with respect to the Clean Water Act, and (3) indices of stressor-specific responses of the invertebrate community to alteration in temperature and fine sediment. We will also determine if detection frequencies of amphibians differ among treatments and reaches.

Analytical Framework

Analytical frameworks provide a methodological template for answering research questions and allow for systematic evaluation of data requirements. In this section, we provide brief explanation of the proposed sampling scheme, field measurements, and analytical techniques. This section is not intended to limit future analyses, but rather to ensure that data requirements and data use have been considered and included in the project budget. Additional analytical methods may be employed as needed.

Sampling Scheme and Field measurements

Within each Type N basin, the Np stream channel network will be delineated into variable length (<150 m) reaches based on changes in topographic shade or riparian condition, significant changes in channel morphology, and changes in flow characteristics (i.e., perennial, seasonal, and mixed as determined through reconnaissance) (Figure 4). Field measurements will occur at reach endpoints or be distributed within reaches depending on the measurement (e.g., temperature at endpoints, shade between endpoints). Discharge and suspended sediment concentration will be measured at flumes located at the basin outlet (Table 4).

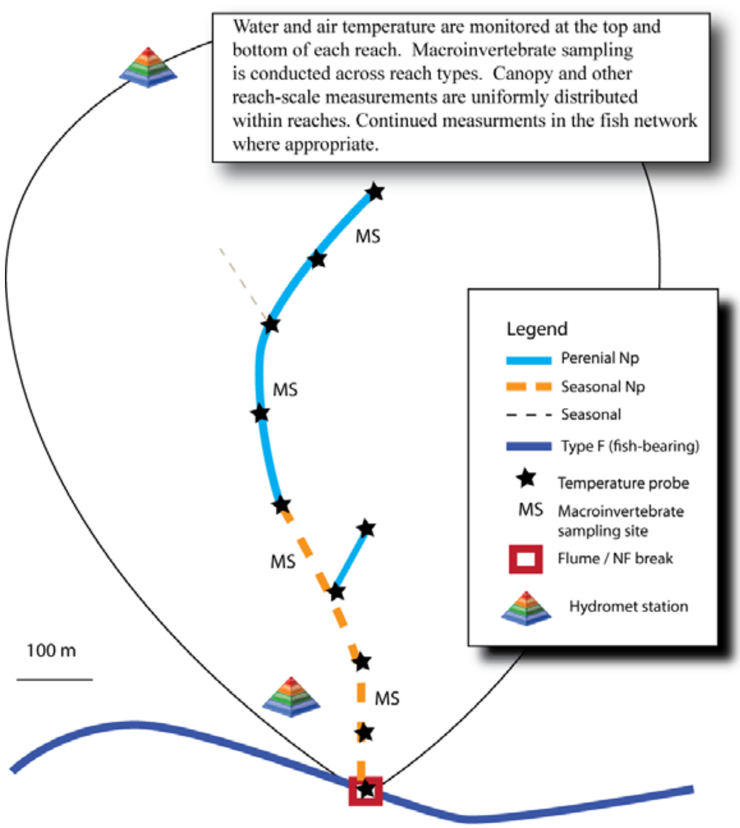

Figure 4: Conceptual diagram showing distributed variable length reaches with temperature monitoring at the upstream and downstream ends with a flume at the Np basin outlet.

Table 4. Variables measured in the study.

Parameter	*Methodology/instrument*	*Resolution*	*Frequency*
Measured at selected sites within each basin			
Hydrologic Maturity	Washington Forest Practices Board Manual	Basin	Pre- and Post-harvest
Air temperature/relative humidity	Vaisala HMP60 combination probe -40 to 60 °C; 0 to 100% RH	0.05 °C 0.02% RH	0.25 hour
Shortwave radiation (SW) and Longwave radiation (LW)	Hukseflux RA01 2-component radiometer SW: 0 to 1600 W m^{-2}; LW: -300 to 300 W m^{-2}	~0.015 W m^{-2}	0.25 hour
Wind speed and direction	Met One 034B 0 to 45 m s^{-1}	0.11 m s^{-1}	0.25 hour
Snow depth	Campbell Scientific SR50A 0.5 to 10 m	0.25 mm	0.25 hour
Precipitation (all weather)	OTT Pluvio^2 1500 mm capacity	0.1 mm	0.25 hour
Soil temperature	Campbell Scientific 107 -35 °C to 50 °C	0.2 °C	0.25 hour
Air temperature (points)	StowAway TidbiT -20°C to 50°C	0.2°C	0.25 hour
Surface water temperature (points)	StowAway TidbiT -5°C to 37°C	0.2°C	0.25 hour
Surface water temperature (longitudinal)	Longitudinal survey of stream temperature in Np network using TidbiT and GPS	0.2°C	Seasonally
Subsurface water temperature (points)	StowAway TidbiT -5°C to 37°C	0.2°C	0.25 hour
Effective shade	Hemispherical photos and solar path analysis, Solar Pathfinder, Eppley pyranometer @ 1 m and water surface	NA	Annually
View-to-sky	Hemispherical photos and view-to-sky analysis, Spherical Densiometer @ 1-m and water-surface	NA	Seasonally
Tree stocking	Vegetation transects within RMZs	NA	1-pre, 1-post
Sediment delivery pathway	Survey of all channels and roads	10 m	Seasonally
Lateral and longitudinal extent and distribution of wetted channel	Survey of entire channel system	10-m reach	Seasonally
Channel change	Monumented cross-sections surveyed with a total station from 10 locations per site.	1 cm	Annually
In-channel LWD and SWD	Samples collected from 3-5 reaches within each basin	100-m reach	Seasonally
Periphyton and detritus standing crops	Samples collected from 3-5 reaches within each basin	100-m reach	Seasonally
Benthic invertebrate densities and composition	Samples collected from 3-5 reaches within each basin	100-m reach	Seasonally
Measured at the outlet of each basin			
Water stage/discharge (point)	Flume with pressure transducer	NA	10-minute
Turbidity	DTS 12 Digital Turbidity Sensor with ISCO Model 3712C automatic pump sampler	0.01 NTU	10-minute

Suspended sediment concentration	ASTMD3977B	1 mg/L	Storm events @ turbidity thresholds

Hydrologic maturity

Upland cover will be measured pre- and post-harvest, evaluating hydrologic maturity within each basin, using procedures prescribed by the Washington Forest Practices Board Manual (2011). These methods distinguish forested and non-forested cover, then further classify forest cover based on conifer canopy cover. Pre- and post-harvest levels will be assessed using supervised aerial photo classification, validated in the field using spherical densiometers.

Hydrometeorological data collection

Hydrometeorological monitoring stations will be installed at each pair of study watersheds. One station will be located near the highest elevation and one will be located near the watershed outlet to monitor both precipitation and air temperature lapse rates and provide redundant measurements within the study watersheds. Stations will be located in small forest clearings where open locations are not available. The primary hydrometeorological station will record all-weather precipitation, snow depth, incoming shortwave and longwave radiation, air temperature, relative humidity, wind speed and direction, and soil temperature. A secondary meteorological station will be installed at a contrasting elevation and will record all-weather precipitation, snow depth, and air and soil temperature. Remote cameras will be installed at each weather station to record general patterns of snow distribution on contrasting aspects. Measurements will be recorded year-round at 15-minute intervals, and images will be recorded at least daily.

Data from the hydrometeorological stations will be used diagnostically to understand how timber harvest affects key processes and metrics including runoff ratios, flow-generation mechanisms (e.g. snow melt, rain-on-snow, rain), sediment transport, large wood recruitment from windthrow, and timing and amount of stream heating. This will be especially important to understand how changes in basin-level conditions (e.g. increased runoff generation) interact with reach-level changes (e.g. reduced shade) to produce temperature and other related changes.

Water and air temperature measurements

Automated temperature loggers will be used extensively to capture variations in both time and space that are expected to be relatively large given the small size, and therefore sensitivity of these systems to changes in the stream energy budget. Surface water temperature (Table 5) will be recorded year-round at 15-minute intervals using TidBit dataloggers (Onset Computer Co) at fixed stations at the top and bottom of each reach. Thermal reconnaissance of monitoring sites to identify obvious areas of groundwater upwelling and flow stagnation will be completed with the aid of a thermal imaging camera. Sampling site reconnaissance will be augmented with electrical conductivity surveys to identify anomalies that may be associated with localized groundwater inflows (Oxtobee and Novakowski, 2002) that may not be evident by thermal imaging (e.g. for cases where stream and groundwater temperatures are not distinct). Stream temperature probes will be placed in PVC pipe and submerged near the channel thalweg in an area without significant upwelling or stagnation. There is little precedent in the scientific literature for either process-based studies or standard methodologies for conducting research on such small streams, and hence it is anticipated that innovative techniques will need to be developed to answer the study questions related to flow and temperature dynamics.

Subsurface water temperature will be recorded year-round at 15-minute intervals TidBit data loggers (Onset Computer Co) at stations at the top and bottom of seasonal reaches. Where installation of piezometers in the streambed are not likely to alter subsurface flow (e.g. alluvium vs. fractured bedrock), stream temperature probes will be installed at depths of 0.15, 0.5, and 1 m below the streambed where possible.

Additional stream temperature probes will be placed in PVC pipe and submerged near the channel in areas suspected of significant groundwater discharge to the stream based on water temperature and electrical conductivity measurements recorded during the channel reconnaissance. Where substantial flow is gained, subsurface water temperatures will also be monitored.

Air temperature will also be recorded at 15-min intervals at each water temperature site. The data logger will be placed approximately one meter above the soil surface, adjacent to the stream bank, and sheltered from shortwave radiation by installing the sensor in a Gill radiation shield to minimize temperature measurement bias (Terando *et al*., 2017).

Riparian cover and effective shade

Riparian cover and effective shade will be measured using various methods intended to support comparison of study results to relevant studies elsewhere in the Pacific Northwest (e.g., Cupp and Lofgren 2014, Ehinger 2015, Gravelle and Link 2007, Groom *et al.* 2011, Keefe *et al*., 2015, Shumar and de Varona 2009, and Teply *et al.* 2015). We will estimate view-to-sky at 1 meter height and at the water surface using a spherical densiometer. A Solar Pathfinder will be used to measure effective shade at 1 meter height and at the water surface. Hemispheric canopy photos taken at 1 m above the water surface and at the water surface will be digitally processed estimate canopy and topographic density (CTD) and effective shade.

Percent canopy cover from low-growing trees, shrubs, and herbaceous vegetation below 1 meter will be measured along the streambank and over the stream using line-intercept methods adapted from Harris *et al.* (2005). Line intercepts will be conducted using ground-based LiDAR.

Riparian vegetation

Riparian stand data will be collected at a series of fixed-area strip plots within each Np reach. Plots will be distributed so that the riparian vegetation within each basin is systematically sampled with a minimum of 10 plots per basin and a maximum of 50 plots. Guidelines for transect spacing in Teply *et al.* (2013) will be followed to minimize autocorrelation among observations. Each plot will extend 7.6 m (25 ft) parallel to the channel azimuth and 15.24 – 22.86 m (50 -75 ft) out in a perpendicular direction on each side of the stream, corresponding to the width of the RMZ buffer. All standing trees ≥10 cm (4 in) diameter breast height (DBH) will be counted in each plot, along with their distance to the streambank, condition (live/dead), species, DBH, and live crown ratio.

Sediment delivery pathways

Measurements of sediment delivery are based on the methodology of (Litschert and MacDonald 2009). This methodology involves mapping and characterizing rills, gullies, and overland flow sediment deposits. In this methodology, rills are defined as erosional features having incised banks with no minimum depth and gullies are erosional features with incised banks greater than 30 cm deep or cross-sectional areas larger than 1000 cm^2; sediment deposits resulting from overland flow are called sediment plumes. To be recordable, a feature must have a minimum length dimension of 10 m so as to be locatable with a resource grade GPS and analyzed against attributes calculated from 10-m DEMs.

Field crews will traverse both sides of all stream channels within the basin to identify sediment pathways. These traverses will parallel the channel at a distance of approximately 10 ft. and along the outer edge of the RMZ. Features will be identified and mapped, the source described (e.g, skid trail, road, fire, unknown), condition (e.g., new exposed, partially vegetated) and dimensions recorded, including length from source to outlet, and it will be noted whether the feature is directly connected to the channel. Field crews will also look for and map any mass wasting features (e.g., slumps, earthflows, debris-slides) and record relative activity level and potential data of occurrence. Road drainage will be mapped and ditch condition, average road slope, and road crown and surfacing characteristics will be recorded for any section draining to a channel.

Pathways will be mapped in GIS and relevant topographic attributes (e.g., planform convergence) associated with each feature will be extracted using the highest resolution digital elevation models (DEM) available.

Stream network surveys

Seasonal GPS surveys and monumented time-series photography will be used to identify the spatial extent of flowing water, discrete changes in discharge, and areas of stream adjacent wetland that are in contact with the channel. Each reach will be given a hydrologic condition code and the change in net stream length for each code will be calculated for each season and year. During the summer sampling event, water temperature and electrical conductivity will be measured at 10 m increments throughout the wetted channel network. These data will be used to identify locations with discrete temperature and/or bulk biogeochemical changes that may result from groundwater and/or hyporheic flow inputs.

Cross-sectional surveys

Cross-sectional surveys will be conducted using a total station with angular accuracy of 1” or better and EDM accuracy of 1 mm + 2 ppm or better and a pole-mounted prism in order to accurately quantify channel aggradation and degradation. Instrument locations and backsight azimuths will be permanently monumented using rebar or magnetic ground stakes that are real time kinematic (RTK) surveyed for redundancy. Cross-sections will be established along a single foresight azimuth. Cross-section surveys will be conducted such that they accurately portray the elevations of channel features within the floodplain, and will record all breaks in slope, any bankfull channel markers, and edge of water surface.

Woody debris

Measurements of large woody debris loading are based on the methodology of Kaufmann *et al.* (1999), allowing quantitative estimates of the number, size, total volume, and distribution of large woody debris (LWD) within study reaches. LWD is woody material with small end diameter of at least 10 cm and length of at least 1.5 m. For each LWD piece, field surveyors will visually estimate length and both end diameters.

Field crews will tally all pieces of LWD that are at least partially within the bankfull channel margins, both vertically and horizontally, followed by those above the top-of-bank but within the channel horizontal margins. Tallies will assign LWD pieces to one of twelve diameter and length categories. The diameter classes are 0.1 m to < 0.3 m, 0.3 m to < 0.6 m, 0.6 m to < 0.8 m, and ≥ 0.8 m, based on the small end diameter. The length classes are 1.5 m to < 5.0 m, 5 m to < 15 m, and ≥ 15 m, based on the portion of the LWD piece that is ≥ 10 cm diameter and within the channel margins.

LWD pieces will be tallied according to 12 diameter and length size classes, with location identified as in or above the channel margins. Primary metrics are then calculated as the number of pieces and estimated volume of wood in the following primary size classes:

Length Diameter	S (1.5 m to < 5.0 m)	M (5 m to < 15 m)	L (≥ 15 m)
S (0.1 m to < 0.3 m)	SS	SM	SL
M (0.3 m to < 0.6 m)	MS	MM	ML
L (0.6 m to < 0.8 m)	LS	LM	LL
X (≥ 0.8 m)	XS	XM	XL

A nominal mean volume will be calculated for each piece of LWD according to its diameter-length class membership, as follows:

(π) [1.33 (Class minimum Diameter ÷ 2)^2] × [1.33 (Class minimum Length)]

Numbers and volumes of LWD in each diameter-length class will be regrouped and assigned to five cumulative size classes and cumulative metrics will be calculated as follows:

Class 1 – All size classes
Class 2 – All size classes, except SS
Class 3 – All size classes, except SS, MS, LS, SM
Class 4 – LM, XM, ML, LL, XL
Class 5 – XL

LWD cumulative metrics calculated for each size class:

PCS/100 m	LWD in within the channel margins (pieces/100m)
VOL/100 m	LWD volume within the channel margins (m^3/100m)
PCS/m^2	LWD within the channel margins (pieces/m^2)
VOL/ m^2	LWD volume within the channel margins (m^3/m^2)
PCS/100 m-all	LWD within and above the channel margins (pieces/100m)

VOL/100 m-all LWD volume within and above the channel margins (m^3/100m)

Small woody debris (SWD) consists of smaller (diameter < 0.1 m, length < 1.5 m) woody pieces that commonly contribute to stream cover, shading, and habitat complexity. SWD can respond to channel disturbance. SWD will be measured adapting Forest Inventory and Analysis protocols for line-intercept methods of monitoring downed fine woody debris (Woodall and Monleon 2008). Pilot surveys will be conducted to determine the number and spacing of transects required to meet acceptable sampling error rates. Transects will be located within the stream channel, aligned perpendicular to the thalweg. Along each transect, SWD will be tallied by one of three size classes (0 to 0.25 in, 0.25 in to 1 in, and 1 to 4 in). Volume of SWD would be calculated directly from transect tallies and biomass would be calculated using bulk density factors tied to these size classes in Woodall and Monleon (2008).

Macroinvertebrates, amphibians, benthic detritus, and periphyton

The aim of sampling macroinvertebrates, amphibians, benthic detritus, and periphyton is to provide a robust estimate of the abundance and composition of aquatic species and their resources. We considered the use of three types of samples that have traditionally been used to estimate the abundance and composition of invertebrates and other taxa within streams – (1) samples taken from benthic substrates (the primary habitat of the vast majority of stream dwelling invertebrates), (2) samples of emerging adult insects, and (3) samples of invertebrates drifting in the water column.

Benthic sampling is by far the most frequently used method for estimating abundance, production, and species composition of stream invertebrate assemblages (Merritt *et al.* 2008) and is the method used in all state-level biological monitoring programs (e.g., Carter and Resh 2001; Carter *et al.* 2006) and most previous studies of the effects of forest practices on stream invertebrates (e.g., Newbold *et al.* 1980; Hawkins *et al.* 1982, 1983, 2000; Campbell and Doeg 1989; Carlson *et al.* 1990; Kiffney *et al.* 2003; Quinn *et al*. 2004; Hernandez *et al.* 2005; Reid *et al.* 2010). Reach-level estimates must be derived from small samples taken at random locations within the reach of interest. A large enough number of these samples must be collected to overcome effects of within-reach, spatial heterogeneity in substrate size and current velocities on reach-wide estimates and produce estimates that are accurate and precise enough to detect ecologically meaningful effects of treatments. However, high variability among replicate samples and the expense of processing high numbers of replicates can compromise the sensitivity of assessments based on benthic samples.

Some previous studies have estimated emergence of adult insects to help assess the effects of forest and watershed disturbance or management on the productive capacity of streams (e.g. Banks *et al.* 2007, Mellon *et al.* 2008), but emergence samples are notoriously inefficient - e.g., based on an extensive review of the literature Davies (1984) reported that emergence traps captured 3-83% of benthic populations depending on species. Moreover, emergence traps sample only species with terrestrial life stages and are much more expensive to use given that many individual traps must be constructed or purchased, installed, and then visited frequently enough (e.g., every 3-4 days) to avoid bias associated with both high seasonal variation in emergence of different species and loss of trapped individuals following capture.

In contrast to benthic and emergence samples, drift samples can provide an estimate of those invertebrates most likely to be consumed by salmonids and other fish species that capture prey from the water column. Drift samples can also provide an estimate of potential invertebrate export to downstream reaches. However, robust estimates of drift require sampling over multiple 24-hour periods, which can be logistically difficult, if not impossible, when streams in several basins must be sampled. The cost of processing the number of samples needed to produce robust estimates of drift densities can also limit their utility.

The small size of headwater streams (often < 30 cm wetted width) restricts the type of traditional sampling gear we can use for invertebrates and amphibians. We will therefore use two sampling approaches to quantify aquatic life endpoints: (1) traditional benthic samples taken from a standard area at each of several randomly selected locations within study reaches and (2) environmental DNA (eDNA) samples filtered from a standard volume of stream water from each study reach (Yu *et al*. 2012; Deiner *et al.* 2017; Elbrecht *et al.* 2017). Traditional samples will allow us to estimate densities and biomass standing crops of the dominant species and identify their microhabitat requirements. eDNA samples represent well mixed DNA shed from organisms inhabiting all habitats within a reach. Because of high detection sensitivities, eDNA will allow us to more fully describe the species composition that occurs in each basin and reach than traditional sampling methods that typically suffer from extremely low detection efficiencies. Use of eDNA will also allow us to identify taxa to a much lower-level of taxonomic resolution than is possible from traditional samples, which are typically identified only to genus, family, or even coarser levels of resolution. eDNA samples also have a much higher chance of detecting specific species of interest that can be very difficult to detect with physical sampling (e.g., many amphibians). The chief disadvantage of eDNA is that eDNA samples cannot yet be used to estimate abundances.

Macroinvertebrates, amphibians, periphyton standing crops, and detritus standing crops will be collected seasonally at 3-5 reaches that represent the major reach types within each basin. Prior to sampling, we will walk the entire stream network in each basin and classify reaches based on their geomorphic and hydrologic properties. Because such small streams as the ones we will study have not been the primary focus of previous attempts to classify stream reaches and habitats (e.g., Hawkins *et al*. 1993, Bisson *et al*. 2017), we cannot a priori identify the specific classification system we will use, but it will incorporate information on whether reaches have perennial or seasonal flow, are geomorphically constrained or alluvial, have higher or lower channel gradients, and have relatively open or closed canopies. The classification should allow us to partition effects of natural environmental factors on aquatic life, but equally importantly it will provide a systematic means of ensuring that we sample all major habitat types within a basin.

At each study reach, we will follow the protocols outlined in Carim *et al.* (2015) to sample eDNA. We will use a peristaltic pump to pass 4-10 liters of water through a 1.5-µm pore size disposable filter. Filters will be carefully removed in the field using sterile techniques and preserved individually with a silica gel desiccant. . Preserved samples will be processed by the Utah State University Molecular Ecology Laboratory. All sampling equipment will be bleach-sterilized between each site to eliminate cross-site contamination. At every tenth site a negative field control (1 L sterilized DI water) will be collected after sampling to monitor for cross-contamination.

At each sampling site, we will also use a small diameter (15 cm diameter) sediment core to collect benthic invertebrates, inorganic substrates, and detritus from five locations within each reach equidistantly spaced along the thalweg of each reach. Cores will be inserted up to 15 cm into the stream bed (or until blocked by large substrates) and all material removed to sample jars. The five samples collected per reach will be pooled to form one sample representative of the reach. Samples will be preserved in 95% ethanol. Samples will later be processed in the lab to remove and identify invertebrates, characterize inorganic substrate size distributions, and quantify detritus standing crops. Periphyton standing crops will be estimated as both chlorophyll-a and ash-free dry matter. Chlorophyll-a standing crops will be measured in situ with a BenthoTorch® at 30 equidistantly spaced locations along the thalweg of each reach. Immediately after sampling, all sampling gear (nets, boots, etc.) will be disinfected to minimize the risk of spreading invasive species or pathogens (Parsons *et al*. 2012).

In the laboratory, we will follow standard protocols developed by the USU/BLM National Aquatic Monitoring Center (https://www.usu.edu/buglab/SampleProcessing/) to process benthic samples. Detritus will be elutriated from the inorganic sediments. This material will then be sub-sampled, if needed, to reduce the time needed to identify invertebrates. Subsampling will entail sequential removal of random portions of the sample until a total of 600 organisms are encountered. If less than 600 individuals occur in a sample, the entire sample will be processed. Individual macroinvertebrates will then be identified to lowest practical taxonomic level (typically genus). The remaining detritus will be dried, weighed, combusted, and then reweighed to determine the mass of detritus present in each sample. Invertebrate densities and detritus standing crops will be estimated from the area sampled and the proportion of each composite sample that was subsampled. Inorganic materials from each sample will be washed through nested sieves of progressively smaller mesh size to isolate different particle sizes. The volume of each size fraction will then be determined by displacement.

Discharge and suspended sediment export

A 12- to 24-inch (depending upon basin area and estimated average annual precipitation) Montana flume will be installed near the Type N/F junction, or in the reference basin at a location likely to have surface water discharge. In each flume, a pressure transducer will measure stage height. Flow will be calculated from stage height using the appropriate equation for the flume. Turbidity Threshold Sampling (Lewis and Eads, 2009) will be implemented using a Forest Technology Systems DTS-12 turbidity sensor, recording at 10-minute intervals, located as near the Type F/N junction as practicable. An ISCO Model 3712C automatic pump sampler, activated at a specific turbidity threshold value on both the rising and falling limbs of the turbidigraph, will collect discrete samples during high turbidity events and during base flow conditions. These samples will be analyzed for suspended sediment concentration (SSC) that will be used to develop a regression model to estimate SSC from the continuous turbidity record. The product of the estimated SSC and the associated flow, estimated from stage height at the flume, will be summed to calculate daily and monthly suspended sediment loads. Loads may also be evaluated over other time intervals to describe suspended sediment transport seasonally or during specific events.

Analytical Methods

As noted previously, the purpose of this section is not to lock the study into an *apriori* set of methods, but rather to describe how the data would be analyzed today to address the research questions below. Alternate methods may be used and the methods described here may be expanded upon.

Research Question 1: How does basin harvest affect the magnitude and timing of surface water discharge from the Type N basin?

For days with measurable surface water discharge, daily discharge will be calculated as the sum of the sub-daily measurements and will be normalized to contributing area. For periods without discharge, we will use a permutation or similar test to determine whether there is a change in the frequency no flow events.

For days with surface water discharge, the daily treatment effect (TE) can be estimated through a comparison of what would have been observed had the treatment not occurred (the "expected condition") and what was actually observed in the treatment basin during the post-treatment period.

$$TE = (y_t - \hat{y}_t) \tag{6}$$

where: y_t is the observed discharge on day *t* of the post-treatment period, and
$\hat{y}_t$ is the expected discharge on day *t* of the post-treatment period.
We will use the regression analysis approach advocated by Watson and others (2001) and Gomi and others (2006) to determine the expected condition. Daily data from the pre-treatment period will be fit to a generalized least-squares (GLS) regression model of the form:

$$y_t = \beta_0 + \beta_1 x_t + \beta_2 x_t^2 + \beta_3 \sin\frac{2\pi t}{365.25} + \beta_4 \cos\frac{2\pi t}{365.25} + \varepsilon_t \tag{7}$$

where: y_t is the temperature in the treatment site on day *t*,
x_t is the temperature in the reference site on day *t*,
$\beta_o, \beta_1, \beta_2, \beta_3, \beta_4$ are the estimated regression coefficients,
$\sin(2\pi t/365.25)$ and $\cos(2\pi t/365.25)$ are terms to account for seasonal variability, and
ε_t is an error term.

The error term (ε_t) can be modeled as an AR(*p*) process where the current error depends on previous errors:

$$\varepsilon_t = \phi_1 \varepsilon_{t-1} + \cdots + \phi_p \varepsilon_{t-p} + a_t \tag{8}$$

where: ε_{t-p} is an error term p days before,
ϕ_p is the autocorrelation coefficient at lag p, and
a_t is white noise centered at 0 and assumed to be independent of previous observations (i.e., $a_t \sim \mathrm{WN}(0, \sigma_a^2)$).

Alternatively, ε_t can be modeled as a MA(q) process where the random error at time *t* are affected by both a perturbation at time *t* combined with perturbations taken before time *t* (additional correlated noise terms). In this case:

$$\varepsilon_t = \theta_1 a_{t-1} + \cdots + \theta_q a_{t-q} + a_t \quad (9)$$

where: a_{t-q} is the noise term q days before, and

θ_q is the correlation coefficient at lag q.

As shown in Pinheiro and Bates (2000), the AR(p) model and the MA(q) models can becombined into an ARMA(p,q) model as:

$$\varepsilon_t = \sum_{i=1}^{p} \phi_i \varepsilon_{t-\mathrm{i}} + \sum_{j=1}^{q} \theta_j a_{t-j} + a_t \quad (10)$$

Since the standard calculation of r^2 is not appropriate to GLS, we will estimate a coefficient of determination (R^2) based on likelihood-ratios:

$$R^2{}_{LR} = 1 - \exp({}^{-2}/_{n} * (\log Lik(x) - \log Lik(0))) \quad (11)$$

where: log*Lik(x)*is the log-likelihood from the fitted model, and

log*Lik(0)* is the log-likelihood from the null model (i.e., intercept only).

We will use Eq. 7 and the pre-treatment discharge data to develop a prediction equation. We will then use that prediction equation to determine the expected discharge for each day in each treatment basin ($\hat{y}_t$).

We will compare several different measures of hydrologic change (both frequency and magnitude) using cumulative frequency and flow exceedance of both observed discharge and predicted discharge. If we detect changes in either flow frequency or magnitude, we will use the hydrometeorological station data to help identify potential causal mechanisms for the change including dominant flow-generating processes, and general role of snowpack in contributing to the surface flow regime.

The spatial extent of dry and flowing reaches will be determined near the start of the late-spring drying period and end of the summer dry periods by direct observations. We will use GPS and total station surveys to create a spatial coverage of hydrological condition over repeat surveys. These data will be incorporated into a single GIS line layer for each period, and by intersecting that lines we will determine where and by how much the hydrologic condition changed between surveys.

Research Questions 2 and 6: How does harvest of Type Np stream basins affect the number, size, total volume, and distribution of woody debris within study reaches; and how do canopy closure, shade, stream cover, hydrologic condition, algal and detritus standing crops, LWD loading, and SWD loading change in response to the different buffer treatments?

We will test for spatial autocorrelation among within-basin metrics using Moran's I and variograms and model the correlation or reduce it through averaging or use statistical methods to account for it. Data will be coded by treatment or reference and by period (e.g, pre-treatment vs post-treatment). We will test for a treatment effect using a generalized linear mixed model (GLMM) that incorporates covariates and random effects as appropriate, and treatment, period, and the treatment × period interaction as fixed effects. If the treatment × period interaction is significant, we will assume a treatment effect and will examine all treatment contrasts and report unadjusted p-values. GLMM's are chosen here because they allow for the specification of error distributions and random effects, and GLMM's were the underlying form of analysis in the Type N Experimental Hard Rock study to which this is a companion.

Research Question 3: Is there an immediate relationship between the frequency and magnitude of sediment delivery pathways from forest units to streams and suspended sediment export (SSE), and how do pathways differ between treatment and reference basins before and after treatment?

We will develop a series of exploratory multivariate linear regression models that attempt to relate sediment pathway characteristics (e.g., volume by type) to suspended sediment export over a range of time periods. The candidate models will consider lags, potential explanatory variables derived through expert knowledge (e.g., distance from source to outlet, stream power), and those derived through exploratory analyses using techniques like regression trees or principle components analysis.

Candidate models will then be evaluated using the information theoretic approach advocated by Anderson (2008). Under this approach, the strength of evidence for each of the competing models will be computed as the discrete probability of each model in the set given the data. For each model in the set, we will estimate model parameters using maximum likelihood or some derivation and will compute an Akaike Information Criteria (AICc) value for the model. AIC is an information theoretic approach for model selection and the model with the lowest AICc value is estimated to be the best in the sense of expected Kullback-Leibler (K-L) information loss (Anderson, 2008).[5]

Because AICc scores contain several unknown constants, we will rescale the AICc scores as the difference between the AICc score for the given model *i* and the model with the lowest AICc score:

$$\Delta_i = AICc_i - AICc_{min} \tag{12}$$

We will then calculate a Bayesian posterior model probability ($\boldsymbol{w_i}$) that the model (i) is the best K-L model using:

$$w_i = \frac{exp(-{}^{1}/{}_{2}\Delta_i)}{\sum_{r=i}^{R} exp(-{}^{1}/{}_{2}\Delta_r)} \tag{13}$$

where: $\boldsymbol{w_i}$ is the exponential difference ($\boldsymbol{\Delta_i}$) between model *i* divided by the sum of exponential differences across all models (Anderson, 2008). If there are more than one plausible models in the set, we will use model averaging across all models using the model probabilities (w_i) as weights and discuss the inference based on the combined set of parameters and their confidence intervals.

To evaluate whether sediment pathway volumes appear to change in response to treatments, we will examine the posterior density for the interaction between sediment volume and study period (e.g., before and after treatment) using hierarchical Bayesian inference solved through Markov chain Monte Carlo (MCMC). We will use a simple multi-level interaction model that incorporates year as a random effect (Kruschke 2015; McElreath 2016).

[5] In practice, it is important to use AICc which corrects for small sample bias.

Research Question 4: How does water temperature change within the Type N basin, and can we relate temperature change to change in shade or other factors?

Within each basin, the reach-scale change in water temperature (TE) will be determined using the methods described above. We will use graphical displays of daily TE and MMTE with associated 95% confidence intervals to illustrate longitudinal patterns in water temperature relative to canopy cover at the water surface, presence of surface water, summer longitudinal temperature profile and effective shade. Frequency histograms of pre- and post-treatment temperature will also be graphically displayed in a longitudinal format.

We will use a simple additive model to evaluate the relative contribution of different factors, including shade, to the reach-scale change in water temperature:

$$\Delta T_{out} = L + \Delta T_{in} + Shade + \Delta Shade + \Delta Hyp + \varepsilon \tag{14}$$

where: ΔT_{out} is TE for probe at the bottom of the reach,
L is length of the stream reach,
ΔT_{in} is TE for the probe at the top of the reach,
Shade is the initial shade provided to the stream by vegetation and topography,
ΔShade is the change in effective shade for the reach as a result of harvest, and
ΔHyp is the post-harvest change in hyporheic residence time.

In evaluating reach-scale change in water temperature, surface water changes will be evaluated along wet reaches and subsurface temperature changes will be evaluated along dry reaches. Where groundwater inputs are observed, a term may be added to the model to account for heat exchange. Reach-scale and basin-scale terms will also be considered, evaluated by their relative influence on temperature flux. We will consider nonlinear terms and will analyze the data using a range of models including Generalized Additive Models and GLMM. We will delete terms (and incorporate other terms as necessary) using information criteria to find the simplest and most parsimonious model.

Water temperature, stream reach length, and shade will all be directly measured, but the change in the hyporheic residence time lag (τ in Eq. 5) will be estimated using air and water temperature. The proposed method is similar to that used by Subehi *et al.* (2009) and involves using linear regression with harmonic variables to quantify the phase of air and water temperature change associated with the diel and annual cycles. Using trigonometric functions to model seasonal temperature cycles, we get:

$$T = A\sin[c(t + \phi)] \tag{15}$$

where: *t* is the time (e.g., day),
A is the amplitude of temperature fluctuation,
c is the cycle which equals 2π/L, where L is the period (e.g., L_{season}=365.25), and
φ is the phase shift.

The parameters in equation 15 are not linear, so we use a trigonometric identity to write:

$$A\sin[c(t + \varphi)] = U_1\cos(c \cdot t) + U_2\sin(c \cdot t) \tag{16}$$

where: U_1=Acos(ϕ), and
U_2=−Asin(ϕ).
Equation 16 is linear and, assuming that U_1 and U_2 are normally distributed random variables, we can write a model of sub-daily temperature measurements expressed as a function of time (decimal days) with daily and seasonal cycles using the following linear model:

$$T_t = \beta_1 \cos(2\pi t) + \beta_2 \sin(2\pi t) + \beta_3 \cos\left(\frac{2\pi}{365.25} t\right) + \beta_4 \sin\left(\frac{2\pi}{365.25} t\right) + \varepsilon \quad (17)$$

where: t is time (decimal day), and
β_1, β_2, β_3 and β_4 are estimated coefficients.
Using the same trigonometric identity shown in 20:

$$\varphi = \tan^{-1}(-\beta_2 / \beta_1) + \tan^{-1}(-\beta_4 / \beta_3) * 365.25 \quad (18)$$

where, ϕ is the daily phase. The lag (τ) between air and water response is estimated as the difference between the two phases ($\phi_{air} - \phi_{water}$) and ΔHyp is the difference in estimated lag for the pre-treatment and post-treatment periods (τ_{pre}-τ_{post}).

We will identify and examine reaches and periods of time where the change in water temperature appears to be insensitive to change in heat flux related to shade or hyporheic flow. We will examine the spatial relationship between these reaches and the spatial distribution of dry and wet reaches at each survey period and local hydrometeorological conditions. We will then perform post-hoc analyses to develop hypotheses about what factors may be driving water temperature response and identify reaches where temperature change appears to be driven by exogenous factors not considered in this study.

Research Question 5: How does harvest affect surface water temperature and 7-day average maximum surface water temperature at the basin outlet and in downstream fish-bearing waters?

Temperature probes located near the outlet of both treatment and reference basins will measure and record water temperature at 15-minute intervals throughout the period of study. The 15-minute data will be aggregated to daily time series metrics representing minimum, maximum, mean, and diel temperature range in order to reduce serial autocorrelation and issues associated with sub-daily lags.

The temperature TE for each day will be calculated using GLS fitting procedures described in the discharge section and the Hard Rock temperature chapter (McIntyre *et al.* 2017). Using these data, we will calculate a mean monthly treatment effect (MMTE) with 95% confidence intervals for each metric using a GLS model that accounts for temporal autocorrelation and different variance by month. Reference to reference comparisons in the Hard Rock study showed stationarity over many years and that the methods should reliably detect a mean monthly temperature changes as small as 0.5 – 1.0 °C with well-matched reference-treatment sites. The large number of comparisons (months) and the large number of locations will increase the likelihood of a Type II error so these data will not be analyzed in an ANOVA framework. Rather the focus will be on the pattern, magnitude, and variability in the monthly estimates.

In addition, we will calculate the maximum 7-day average TE (7DTE) for the July-August period. The 7DTE is used because it is comparable to the change in the maximum 7-day average daily maximum temperature as referenced in Washington State's water quality standards. For each treatment, we will estimate effect size with 95% high density intervals using hierarchical Bayesian inference solved through Markov chain Monte Carlo (MCMC). We will use a simple multi-level model that incorporates year as a random effect:

$$y_i = a_{j(i)} + \varepsilon_i \tag{19}$$

$$\varepsilon_i \sim Normal(0, \sigma^2) \tag{20}$$

$$a_{j(i)} \sim Normal(\mu, \tau^2) \tag{21}$$

where: y_i is the maximum July-August 7-day average TE in year i, and
μ is the expected value for treatment j.

Research Question 6: Changes in suspended sediment export

Turbidity and suspended sediment vary stochastically and as the result of a large number of geomorphic conditions, therefore change in suspended sediment concentration regime will be determined through a quantitative assessment of the historic return period for suspended sediment concentration in the pre- and post-treatment periods.

We will use an FTS system to conduct Turbidity Threshold Sampling (TTS; Lewis and Eads 2009) so that a water sample is collected by an ISCO pump sampler when stage height and turbidity exceeded specified thresholds for two consecutive measurements. This will ensure that water samples are collected across the range of turbidity values on both the rising and falling limbs. We will follow guidelines in Lewis and Eads (2009) to identify data that were influenced by progressive fouling (biofilm), debris fouling, direct sunlight on the sensor, non- or partial submergence of the sensor, burial of sensor or interference from the stream bottom, and air bubbles entrained in the water.

The measured SSC values and the corresponding turbidity data will be used to build a regression model to predict SSC for the entire data record.

$$SSC_i = \beta_0 + \beta_1 Turb_i + \beta_2 Turb_i^{\,2} + \varepsilon_i \tag{22}$$

where: *SSC* is the $\log_{10}$-transformed suspended sediment concentration for sampling event *i,*
Turb is the corresponding $\log_{10}$-transformed minimum turbidity value,
β_0, β_1, and β_2 are regression coefficients, and
ε_i is the error term.

SSE will be calculated as the product of the SSC and discharge for each 10-minute period. We will then use the methods similar to those described by Bywater-Reyes *et al*. (2017) for analysis of the Trask River Watershed data. We will develop annual rating curves between hourly suspended sediment export and hourly unit discharge such that:

$$Q_S = \alpha Q^{\beta} \tag{23}$$

or

$$\log(Q_S) = \beta \log(Q) + \log \alpha + \varepsilon \tag{24}$$

where: Q_s is suspended sediment export (kg ha^{-1} h^{-1}),
Q is unit water discharge (m^2 ha^{-1} h^{-1}),
α and β are the erosion terms that will used to test for treatment effects, and
ε is the error term.

We will then use mixed models to determine whether there is a period by treatment interaction in the rating curve parameters or whether other factors explain patterns in the relationship between SSC. We will also explore the methods outlined by Aich and colleagues (2014) to examine whether there are quantifiable changes in hysteresis patterns between event normalized discharge and SSC.

Research Question 7: How do benthic invertebrate assemblages and their food sources respond to the different buffer treatments?

The condition of macroinvertebrate assemblages in streams has been measured in several ways, which has confounded comparisons across different studies. Here we will use two types of measures: (1) measures directly related to food web dynamics (i.e., abundances of all and individual trophic classes of invertebrates) and (2) measures that assess shifts or losses in biodiversity. The first approach is useful in understanding if trophic-level attributes of invertebrate assemblages are affected by treatments. Such measures can inform us of the food available to higher trophic levels both on site (e.g., Murphy and Hall 1981; Murphy *et al.* 1981; Hawkins *et al*. 1983) and to downstream fish bearing reaches (e.g., Wipfli and Gregovich 2002; Wipfli and Baxter 2010). However, measures of overall abundance often are not sensitive to species replacements or changes in biodiversity. We will therefore also calculate both the change in compositional similarity and the proportion of taxa lost or gained that occurs between basins and reaches following treatment. Ideally, we would also calculate the indices of biological condition used by Washington Department of Ecology to assess the aquatic life status of invertebrate assemblages – e.g., indices of taxonomic completeness or multimetric indices (e.g., Hawkins 2006; Hawkins *et al.* 2000, 2010; Stoddard *et al.* 2008, Schoolmaster 2012), but these indices have not yet been developed for either eastern Washington streams or small headwater streams.

Sampling reaches within each basin will allow us to conduct two types of analyses. First, we can test for overall riparian buffer treatment effects on invertebrates (e.g., abundance, composition), detritus, and periphyton in two ways: (1) by determining if invertebrates and their food sources collected from a standard reach type differ with treatment and (2) by determining if the same attributes weighted by the proportion of different reach types within a basin differ with treatment. This approach will help assess the effects of within-basin environmental heterogeneity on treatment comparisons. Second, we will use Random Forest models (Cutler *et al*. 2007, Hawkins *et al.* 2010) to determine how reach-level biotic attributes vary in response to environmental differences at both the basin scale (buffer treatments, naturally occurring differences in basin geology, climate, and nutrients) and at the reach scale (shade, temperature, channel morphology and substrates, and flow). For invertebrate models, we will also include estimates of detrital and periphyton standing crops as predictor variables. These models will be used to identify the relative importance of different environmental drivers to detritus, periphyton, and different measures of invertebrate condition (see below). These models will also help identify interactions between naturally occurring and management-related environmental conditions that may produce contingency in biological responses to forest practices and confound inferences.

We will also use taxon- and stressor-specific 'tolerance' values to assess if the macroinvertebrate assemblages in treatment reaches and basins are responding to specific types of environmental alterations. For example, freshwater species vary markedly in their temperature preferences and tolerances, and thermal preferences for benthic macroinvertebrates are relatively well established (Brandt 2001, Huff *et al.* 2005, Carlisle *et al.* 2007). These data can be used to calculate an index of temperature-caused biological alteration. The difference between the mean temperature optima of species observed at a water body and the mean temperature optima of species expected under baseline (or control) conditions represents an assemblage-wide index of thermal-caused ecological change associated with forest practices. To create a temperature sensitive index of change (O/E_{temp}) in the ecological community, we weight taxon abundances at both the treatment (O) and control (E = expected) sites by each taxon's thermal optima, i.e., $O/E_{temp} = \sum(O_i * t_i)/\sum(E_i * t_i)$, where O_i = the abundance of taxon *i* in treatment reaches and E_i = abundance of taxon *i* in control reaches, and t_i = the temperature optimum of taxon *i*. This approach is being developed for causal analysis purposes in bioassessment (e.g., Yuan 2006) and should be generally suitable for use in detecting effects of thermal alteration. The same approach can be used to estimate indices for other stressors for which we have tolerance values – e.g., fine sediment (Reylea *et al.* 2012, Extence *et al.* 2013, Glendell *et al.* 2014, Murphy *et al.* 2015).

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