ENVIRONMENTAL FACTORS INFLUENCING PHOSPHORUS AVAILABILITY AND SPECIATION IN FOREST AND MEADOW SOILS OF THE LAKE TAHOE BASIN

A Thesis
Presented in Partial Fulfillment of the Requirements for the
Degree of Master of Science
with a
Major in Soil and Land Resources
in the
College of Graduate Studies
University of Idaho
by
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December 2019

Authorization to Submit Thesis

This thesis of Thomas J. Heron, submitted for the degree of Master of Science with a Major in Soil and Land Resources and titled “Environmental Factors Influencing Phosphorus Availability and Speciation in Forest and Meadow Soils of the Lake Tahoe Basin,” has been reviewed in final form. Permission, as indicated by the signatures and dates below is now granted to submit final copies for the College of Graduate Studies for approval.

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Abstract

Legacy fire suppression in the Lake Tahoe Basin has led to accumulation of forest litter, which may represent an important source of phosphorus contributing to eutrophication and reduced clarity in the Lake. The distribution and dynamics of soil phosphorus in the Basin's forests is not fully understood. In Chapter 2, the response of three labile soil phosphorus fractions to seasonality, ecosystem, and parent material is examined between two neighboring watersheds in a site-specific analysis. In Chapter 3, relationships of different environmental, mineral soil, and organic horizon properties with several pools of soil phosphorus are described around the Basin. Results showed that 1) parent material influences the magnitude of labile soil phosphorus concentrations, 2) ecosystem type is a major driver for source of labile phosphorus, and 3) organic forms comprise the majority of soil phosphorus readily lost to surface waters. Effective mitigation of phosphorus accumulation in watershed soils should use parent material and ecosystem type to prioritize forest management strategies.

ACKNOWLEDGEMENTS

Thank you to my advisor, Dr. Dan Strawn, for providing me with many opportunities to develop into a stronger soil scientist during the process of completing this thesis, as well as my committee members, Dr. Erin Brooks and Dr. Mariana Dobre. Project collaborators also included Dr. Caley Gasch (North Dakota State University) and Dr. Barbara Cade-Menun (Agriculture and Agri-food, Canada). Thank you to Alex Crump for your patience and guidance while I navigated my experience in the lab, as well as several undergraduate research assistants who provided much help to me along the way– Tyler Clemens, Jackson Cook, Kendra Gady, and Tiffany Perez. I would also like to thank department research specialists, Kendall Kahl and Anita Falen for assistance on specialized laboratory analyses. Additionally, thank you to Dr. Julia Piaskowski from the University of Idaho, College of Agricultural and Life Sciences Statistical Programs Unit, for consultation and insight. Finally, thank you to USDA-NIFA for funding the work in this thesis.

Dedication

I would like to dedicate this thesis to several people who provided me their love and support both before and during the process. Thank you to my parents Simonetta and Bill for instilling in me a value to bring a sense of pride to all activities in life. Thank you to my sister, Elizabeth, for your optimism and humor which has been especially appreciated the past couple years! Thank you to my friends, both those from Pennsylvania, and those I met at the University of Idaho, who have enriched my experience here. Finally, thank you Mackenzie, for your patience, encouragement, and the special memories that we made together while I finished this thesis.

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Al	aluminum
B1P	Bray-1 phosphorus
$\mathbf{B1P_{T}}$	total Bray-1 phosphorus
$\mathbf{B1P_{MR}}$	molybdate-reactive Bray-1 phosphorus
$\mathbf{B1P_{MU}}$	molybdate-unreactive Bray-1 phosphorus
Fe	iron
M3P	Mehlich-3 phosphorus
MBP	microbial biomass phosphorus
P	phosphorus
TOC	total organic carbon
TN	total nitrogen
WSP	water soluble phosphorus
$\mathbf{WSP_{T}}$	total water soluble phosphorus
$\mathbf{WSP_{MR}}$	inorganic water soluble phosphorus
$\mathbf{WSP_{MU}}$	organic water soluble phosphorus

Chapter 1: Introduction

1.1 Phosphorus History in the Lake Tahoe Basin

Phosphorus is an essential nutrient for all life, as it is key to the make-up of important biological compounds such as DNA and ATP (Correll, 1998). Despite the role of phosphorus in supporting life, the balance of ecosystems can be disturbed when an excess is added. This is particularly noticeable in aquatic ecosystems because increased phosphorus (P), the most common limiting nutrient, leads to algal blooms and eutrophication (Correll, 1998; Smith et al., 1999).

Since 1980, atmospheric deposition of nitrogen (N) in Lake Tahoe has caused a shift from N towards P limitation on the growth of phytoplankton (Elser et al., 2009; Goldman et al., 1993). Lake Tahoe still maintains an ultra-oligotrophic state (Hatch et al., 2001). However, due to nonpoint source nutrient loading, primary production in Lake Tahoe increases by approximately 5.66% each year (Goldman et al., 1993; Roberts & Reuter, 2010). Secchi depth (a method for measuring water clarity) has decreased from approximately 31 m in 1968 to 21 m at present day (Kerlin, 2017). This reduction in clarity is due to both sediment loading from erosion, and primary productivity as a response to dissolved and bioavailable nutrient loading (Jassby et al., 1999). Organic particles in the lake from primary productivity are responsible for 25% of water clarity attenuation in Lake Tahoe (Swift et al., 2006). Short-term changes in lake primary productivity are well-explained by dissolved inorganic and organic P loads from the Basin's streams (Hatch et al., 1999). A recent lake clarity model demonstrates that a return to the historical Secchi depth reading is possible within 20 years with load reduction rates of at least 2.75% per year from the current total maximum daily load (TMDL) (Sahoo et al., 2010).

1.2 Background

1.2.1 Nutrient Loading from Forests

Despite the long-held preconception of forests as nutrient sinks, they are becoming recognized as an important source of nonpoint nutrient loading, annually contributing 495,000 Mg of P to surface waters in the United States (Carpenter et al., 1998). This makes up about 30% of P from all nonpoint sources, and 25% from all nonpoint and point sources combined. Forest management strategies such as harvest can increase nutrient loading to watershed streams (Ice & Binkley, 2003), however increased P levels in streams have also been associated with undisturbed old-growth conifer forests (Binkley, 2001). Litter accumulation on forest floors may not always act as a nutrient sink; rather, mineralization of these nutrients can increase the potential for transport through both surface and subsurface hydrologic pathways (Miller et al., 2010). Considering the significant role that forests hold as nonpoint contributors of phosphorus to surface waters, it is important to investigate the processes that govern nutrient transport in forested ecosystems. The Lake Tahoe Hydrologic Model (Tetra Tech, 2007) divides forests into three land-use categories: unimpacted ("forested areas that have been minimally affected in the recent past"), burned, and harvested. Unimpacted forest land constitutes about 87% of total land-use distribution in the basin, while single and multi-family residential, non-residential, roads, and other vegetated areas cover the remaining basin surface (Table 3-7 from Tetra Tech, 2007). Considering that forests occupy most of the land area, their potential for releasing P to surface waters in the Lake Tahoe Basin must be investigated.

Several studies (Sharpley et al., 2001; Vadas et al., 2005) have provided evidence showing a positive relationship between labile soil P fractions and P concentrations in runoff. At Lake Tahoe, previous work has considered the potential of P input from forest litter accumulation and subsequent loss from interflow and runoff (Miller et al., 2005;

Loupe, 2005). Although previous research has emphasized overland flow from forests as a source of nutrient loading to the lake, estimates suggest that groundwater sources make up 15% by mass of total P loading (Roberts & Reuter, 2010). Furthermore, 61% (3700 kg) of the total dissolved P that is found in the basin's annual groundwater is believed to be derived from natural sources from unimpacted areas (U.S. Army Corps of Engineers, 2003). From the same Army Corps of Engineers report, about 65% of total groundwater P is believed to come from the region designated "Tahoe City/West Shore" which encompasses the western quadrant of the Tahoe Basin.

Shallow groundwater or baseflow is treated separately from groundwater in the Lake Tahoe TMDL Report (Roberts & Reuter, 2010). Using historical data from the Lake Tahoe Interagency Monitoring Program, the Lake Tahoe Watershed Model predicted baseflow to make up 75% (3.36 x 108 m^3) of the total combined volume of annual flow from both surface and subsurface runoff. In the TMDL report, the term baseflow is attributed to shallow groundwater that interacts with the basin's soils before flowing into surface water. Rowe and Allander (2000) measured a groundwater seepage, or baseflow, contribution of 38% to streamflow in the Upper Truckee River, which reinforces that subsurface runoff is a significant hydrologic pathway in some watersheds in the basin. The Lake Tahoe Watershed model assumes all nutrients in baseflow are in dissolved form. This model predicted an annual load of nearly 8,000 kg P from baseflow in non-urban areas, which makes up 94% of total P loading to surface waters from all baseflow in urban and non-urban areas (Roberts & Reuter, 2010). Because shallow groundwater is a substantial source of water to Lake Tahoe, the Basin's soils may hold a considerable role as either a sink or source for P being transported in groundwater that enters streams or flows directly into the lake. When accumulation of P surpasses the soil's capacity to sorb P, it is vulnerable to leaching and loading to surface waters, leading to eutrophication in the environment (Sharpley, 1995).

1.2.2 Soil Impact on Surface Waters

Phosphorus loss from soils through vertical leaching to groundwater and subsurface runoff is an important transport path to surface water (Sims et al., 1998). Heckrath et al. (1995) determined environmentally safe upper limits of agronomic soil test P in agricultural silt loam and silty clay loam soils, which they termed the "change point." Beyond this limit, soil P is vulnerable to increased losses through leaching and dissolved reactive P (DRP) in drainage water more closely follows soil available P trends. Maguire and Sims (2002) demonstrated that DRP in soil leachate commonly surpasses levels that lead to eutrophication of surface waters, and identified change point levels for additional soil available P test methods. Alternatively, Djodjic et al. (2004) found that soil water transport mechanisms (preferential versus matrix flow) are a more significant indicator of P leaching than measurements or indices of soil available P. These findings all suggest that P leaching from the soil profile is a significant pathway for transport to surface waters. Studying the nature of labile P at the soil-water interface in Tahoe Basin soils would provide appropriate context to determine if soil P concentrations are an important indicator of potential for P transport. Furthermore, most studies comparing soil available P and leachate P appear to have examined agricultural soils. In this study, forest soils of the Tahoe Basin will be evaluated for P forms that may be prone to leaching processes.

1.2.3 Soil Phosphorus Fractionation

Studies of phosphorus dynamics in soils commonly use a combination of different extraction methods to interpret the distribution of soil phosphorus pools. Several sequential extraction methods have been developed to determine the distribution of soil phosphorus in relevant fractions. Chang and Jackson (1958) applied their fractionation method to several soils. This sequential fractionation method measures inorganic phosphorus distribution across multiple mineral fractions representing differing levels of chemical weathering. These fractions include calcium phosphate, aluminum phosphate, iron phosphate, and occluded phosphate. Hedley et al. (1982) developed an alternative sequential fractionation method that allows for the separation of both inorganic and organic P from a range of extractable pools from labile to residual. In order of increasing stability, the full Hedley fractionation first estimates readily exchangeable resin inorganic P, followed by organic and inorganic labile P adsorbed to soil surface, microbial P, surface and internal aggregate chemisorbed Fe- and Al-P using a series of base extractants. Finally, acid extractants access apatite, occluded P, and the insoluble residual organic and inorganic P forms. Several studies have used the Hedley fractionation scheme as a basis for understanding P dynamics in various systems. Cross and Schlesinger (1995) conducted a large review of studies applying the Hedley fractionation to soils belonging to nine different soil orders. They found that the Hedley method can be used to relate the significance of biological versus geochemical cycling of phosphorus in a given soil. Feng et al. (2016) reported similar findings when conducting the Hedley fractionation method on soils along a climosequence. They found that with increasing degrees of weathering, soils had less phosphorus in the inorganic primary mineral fraction and more in the organic fractions. Condron and Newman (2011) identified specific elements in the protocol for Hedley sequential P fractionation which may lead to possible deviation between the operationally defined fraction and the fraction truly extracted in the experiment. For example, the distinction between organic and inorganic P in the Hedley method is improved with the addition of a base extraction following the acid extraction step.

Often, the only soil P fractions of interest for an experiment are those that make up the labile pools, which are the most dynamic and readily available for plant uptake or transport through the soils. Some common agronomic tests that measure soil P available for plant uptake are Mehlich, Bray, and Olsen extraction methods, while environmental tests for soil available P include water-soluble P (WSP), calcium chloride P, and iron strip P (Maguire & Sims 2002).

1.2.4 Modeling Phosphorus Transport

Effective management of phosphorus transport from nonpoint sources to surface waters involves the identification of sensitive areas in the landscape, where P is considered most vulnerable to losses from the soil (Sharpley et al., 1993). To accomplish this, simulation and landform assessment models are necessary that incorporate the two main factors controlling P movement, which are transport pathways (runoff and erosion) and sources. Several models have attempted to model P transport in various ways within specific geographic regions. Daly et al. (2002) used three environmental factors which included land use, soil type, and soil P data to model P discharge in rivers from 35 watersheds in Ireland. Depending on soil type, this model was able to explain 62-68% of variation in P loading to surface waters. The group attempted to apply the model across the entire nation, but this did not produce accurate results. However, the model was successful in identifying the combination of environmental factors yielding conditions most related to P losses. Kirsch et al. (2002) used the SWAT model to predict P transport under different management strategy scenarios. The transport and source inputs that this model used were climate, soils, internal drainage, wetlands/ponds, baseflow separation, groundwater, lakes, point sources, and land use. The model was successful in predicting locations of high loading or discharge rates, but not in predicting source areas in the landscape. Van der Perk et al. (2007) looked at the effects of soil type, soil chemical properties, and land use on the distribution of P in a watershed's soils, sediments, and surface waters. This group found that soil parent material and chemical properties were the best predictors of total and Olsen soil P. Schoumans and Groenendijk (2000) developed a chemically based model using sorption and desorption kinetics to predict annual P accumulation (fixation and immobilization) and losses (subsurface runoff and leaching). However, this model was limited to non-calcareous sandy soils in flat landscapes.

The Water Erosion Prediction Project (WEPP) is a modeling tool that can be used to predict sediment loading (Flanagan et al., 2010). This model has the potential to expand to include the ability to predict "hotspot" locations that might constitute nonpoint sources of P in a watershed. This would be accomplished by studying the relationships of certain readily available environmental data (such as topography and climate) with measured soil physicochemical properties that control soil P dynamics. The goal of Chapter 3 was to measure these soil parameters and identify key statistical relationships.

1.3 RESEARCH OBJECTIVES

The goal of this thesis is to explore the nature of labile P distribution and dynamics in soils of the Lake Tahoe Basin. Chapter 2 is a site-specific analysis of the influence of parent material, ecosystem type, and seasonality on soil labile P distribution in forested and meadow soils in Lake Tahoe Basin. Relationships between solid and solution phases from extractable pools are evaluated, and an explanation of a landscape process for labile P storage is provided. Chapter 3 explores relationships between a collection of environmental variables, soil properties, and phosphorus in the soil and litter layers in the Lake Tahoe Basin. The feasibility of these parameters for predicting soil P in the landscape is discussed, and relationships between various environmental factors and soil properties with forms of soil P at depth are described.

Chapter 2: Landscape Partitioning of Labile Soil Phosphorus in Response to Parent Material, Ecosystem, and Seasonality on the West Shore of Lake Tahoe, California

2.1 Overview

Phosphorus Dynamics in Riparian Meadow Ecosystems

Meadow ecosystems and their soils provide important functions for water quality and quantity, including reduction of sediment and nutrient loading to surface waters (US EPA, 2016). In the Lake Tahoe Basin landscape, meadows are transitional zones connecting terrestrial and aquatic systems and commonly exist adjacent to forests (Roby et al., 2015). Depending on spring snowmelt runoff rates, streams in the Basin have been observed to annually contribute up to 1,000 kg of soluble reactive (< 0.45-µm) phosphorus (P) to Lake Tahoe, which may represent an important short-term source of bioavailable P (Hatch et al., 1999). Forest floors with thick organic horizons in the Basin supply high levels of P after leaching with water (Miller et al., 2005; Miller et al., 2008). In some locations, hydrologic transport of P crosses meadows, which can act as sinks for the P, or as a source of P to surface runoff or lateral flow that eventually is released to streams and lakes.

Some wet meadows in the Lake Tahoe Basin are categorized as "stream environment zones," (SEZ) which is a designation used by the Lake Tahoe Basin Management Unit for an area of high value and management priority on the basis of ecosystem services that the land offers, including the filtering and storage of nutrients in runoff (Roby et al., 2015). Several groups have studied the capacity of similar riparian systems to perform these functions in other geographic locations. Casey et al. (2001) used phosphorus isotherms in soils including Cumulic Humaquepts (similar taxa are found in some Tahoe meadows) to demonstrate the importance of sorption capacity as a mechanism of nutrient attenuation. They determined that soil P concentration at time of sampling was 100 times less than that which would lead to solution P levels exceeding USEPA recommendations for lentic waters. Hoffman et al. (2006) found net loss of P via leaching from soils in riparian meadows during 2 of 3 sampling years. This group also found an inverse coupling of P and N dynamics, where P loss increased when N retention was elevated. Hoffman et al. (2009) explained that although sedimentation in riparian buffers is an important mechanism of P retention, these buffers may eventually become significant sources of DRP release to surface or groundwater.

Many studies of nutrient attenuation in riparian buffers involve those surrounded by agricultural or urban land use (including those referenced above), likely due to the commonly held perception of forests as nutrient sinks rather than sources. Therefore, the dynamics and vulnerability of phosphorus to be leached from both forest and meadow soils requires furthering studying to understand the relative potential impact on clarity of Lake Tahoe.

Seasonality and Labile Phosphorus

Seasonality has also been identified as an important signature on labile soil P patterns. Seasonality and vegetative cover were important drivers for differences in several pools of labile P observed by Zhao et al. (2009).This group's study found accumulation of total soil P during dry winter months in grasslands, but not in forest in a semiarid temperate region of China. They found labile P increases during the wet summer, and decreases in spring and autumn. This is dissimilar to the finding in soils of a tropical dry forest of Mexico studied by Campo et al. (1998), where labile P was found to accumulate instead during the dry season in water-soluble, bicarbonate-extractable, and microbial biomass pools. Fabre et al. (1996) found that seasonality led to differential trends in organic versus inorganic fractions of labile P in riparian forest soils of southwest France with high levels of spring precipitation and snowmelt. During the year of 1991, the group found labile organic P increased in the fall while labile inorganic P increased during late spring/early summer. Consequently, fractionation of P into organic and inorganic forms can influence seasonal mobility of P in the environment.

Mobility of Inorganic and Organic Forms of Phosphorus

Dominance of inorganic versus organic forms of labile P that are vulnerable to transport through watersheds varies depending on the environment. In podzol soils of a mixed coniferous forest in Finland, Backnäs et al. (2012) described the significance of profile depth, explaining that higher soluble organic P, although lesser than the inorganic fraction, was found in surface horizons due to presence of labile monoesters and diesters. Instead, Uselman et al. (2012) suggests that the amount of dissolved organic P in soil solution is largely dependent on the quality and ratio of above- and below-ground litter inputs. Multiple studies in different types of ecosystems have also shown the general prevalence of inorganic forms in soluble P; for example, Huang and Schoenau (1998) in boreal aspen soils; Magid and Nielsen (1992) in grassland and pasture soils. However, more recent research methods and findings point toward the importance of organic forms as a dominant source of P loading to surface waters. One study using sandy loam soils observed higher levels of labile organic P in solution after separate column leaching and isotherm experiments with DNA and inorganic potassium phosphate (Anderson & Magdoff, 2005). Another approach separated bulk soil extractions into colloidal and electrolytic fractions, and found most P was bound to colloids in organic forms (Missong et al., 2016). A study of soils formed in three different parent materials found an increase P mobility at sites that had lower total soil P status, and furthermore a tendency for organic forms to dominate mobilized dissolved P (Brödlin et al., 2019). The dynamics of biogeochemical cycling that affect P mobility are complex, and appear to be mostly influenced by environmental factors.

Research Objectives

The goal of this study is to investigate how phosphorus transforms between labile pools in soils of adjacent meadows and forests in response to seasonality from June through October. The pools of soil P to be studied are water-soluble, Bray-1, and microbial biomass because these are considered to be the seasonally dynamic and labile pools of soil P (Campo et al., 1998; Pistocchi et al., 2018). Different soil tests have demonstrated that they accurately explain P losses in leaching and runoff. Initially, certain extractants such as Bray-1 and Olsen were developed as soil fertility indices for plant-available P based on regional soil characteristics including soil pH (Jones, 1998). Bray-1 P is a more appropriate extraction for adsorbed labile P in acidic (pH <7) soils (Kovar & Pierzynski, 2009). Furthermore, water-soluble P and Bray-1 P extractions have been shown to better explain variation in runoff dissolved reactive phosphorus for a range of soil types (Pote et al., 1996; Vadas et al., 2005; Wang et al., 2010). The objectives of this study are to i) evaluate the partitioning of labile P in the landscape with regards to different factors including ecosystem type (meadow versus forest), seasonality, and parent material, and ii) investigate inorganic versus organic P fractionation in water-soluble and Bray-1 extractable P in soils of the Lake Tahoe Basin.

2.2 Methods

2.2.1 Study Sites and Sample Collection

Soils were sampled during 2018 around Page Meadow and Meeks Meadow, two forest-adjacent subalpine meadow systems located on the west shore of the Lake Tahoe Basin. Page Meadow is an alluvial floodplain surrounded by forested hillslopes of terminal moraines. Meeks Meadow is situated in an elongated glacial valley trough floodplain, confined on both sides by steep forested hillslopes of lateral moraines. Separate lobes of the Sierran Ice Cap extended over the present-day locations of both meadows, carving out their current floodplain topographic environments (Figure 2 of Ehlers & Gibbard, 2003). At Page Meadow and its surrounding forest, this resulted in the displacement and redistribution of basaltic and andesitic rocks from Miocene- through Pleistocene-age volcanic activity into younger glacial landforms (Kortemeier et al., 2018). Geologic substrate of the Meeks Meadow watershed is primarily granodiorite as glacial drift of till and outwash (Saucedo, 2005). Both meadows contain perennial grasses mixed with sedges, rushes, and forbs (USDA NRCS, 2007). The forest surrounding Page Meadow resembles a red fir forest association, including Jeffrey and lodgepole pines. Next to Meeks Meadow the forest consists of yellow pine association with Jeffrey pine, white fir, incense cedar, and sugar pine (TRPA, 2015; USDA NRCS, 2007). Climatic data from Tahoe City and Rubicon SNOTEL stations show approximate cumulative precipitation of 900 millimeters and mean annual temperature of 7.5 °C (Figure A.2; SNOTEL, 2018). Soils at nearby SNOTEL Stations fall under a xeric soil moisture and frigid soil temperature regimes (Table 2.2; Figure A.1).

Table 2.1: Profile descriptions summarized for a subset of meadow and forest soils in both the granitic and andesitic watersheds in this study.

Surface Soil Profiles from Granitic Watershed			**Surface Soil Profiles from Andesitic Watershed**		
Granitic Meadow Soil			**Andesitic Meadow Soil**		
Note: Closest to Meeks Creek			*Note: Floodplain position; closest to mouth of Page Meadow watershed drainage*		
Horizon	**Bottom Depth**	**Comments**	**Horizon**	**Bottom Depth**	**Comments**
Oi	3 cm	Grassy residues	Oi	1 cm	Grassy residue
A	15 cm	Dark from accumulation of soil organic matter (SOM)	A	3 cm	SOM accumulation
C	-	Sand bed	Bw	17+ cm	15-20% gravels
Granitic Meadow Soil			**Andesitic Meadow Soil**		
Notes: Toeslope-floodplain transitional zone; abundant redoximorphic features			*Notes: Floodplain position; near location of seasonal pond that dries up by late summer*		
Horizon	**Bottom Depth**	**Comments**	**Horizon**	**Bottom Depth**	**Comments**
Oi	4 cm	Grassy residues	Oi	2 cm	Grassy residue
A	8 cm	SOM accumulation	A	9 cm	SOM accumulation
Acg	12 cm	Many prominent depletions; common prominent concentrations; iron concretions observed	AB	16 cm	Lighter color; more aggregation
Ag	16+ cm	Increase to many, prominent iron concentrations; many prominent depletions	Bw	32+ cm	1-2% gravels; weak structure
Granitic Meadow Soil			**Andesitic Meadow Soil**		
Notes: In floodplain, about halfway between creek and footslope			*Notes: Floodplain position; opposite end of meadow from mouth of drainage*		
Horizon	**Bottom Depth**	**Comments**	**Horizon**	**Bottom Depth**	**Comments**
Oi	1 cm	Grassy residues	Oi	0.5 cm	Grassy residue
A1	4 cm	Many roots; granular	A1	5 cm	Few, faint iron concentrations; 1 SBK to GR
A2	10 cm	Less roots; more mineral soil material; subangular blocky (SBK); Slightly higher value soil color	A2	10+ cm	∼3% rock fragments
Bw	16+ cm	Slightly more developed SBK peds			
Granitic Forest Soil			**Andesitic Forest**		
Notes: Forest sampling sites less variable than meadow; fairly uniform profiles in top ∼15 cm			*Notes: Slightly elevated , local upland position relative to the meadow and located centrally in an area of forest*		
Horizon	**Bottom Depth**	**Comments**	**Horizon**	**Bottom Depth**	**Comments**
Oi	2 cm	Pine needles	Oe	2 cm	Unstratified mix of organic materials at varying extents of decomposition
Oe	4 cm	Increased decomposition	A	7 cm	Mineral soil, darker from SOM
A1	6 cm	Mineral soil, no gravels	Bw1	20+ cm	∼10% gravels; lighter color than AB
A2	10+ cm	∼5% gravels			
			Andesitic Forest		
			Notes: Toeslope position, close to forest-meadow boundary		
			Horizon	**Bottom Depth**	**Comments**
			Oe	1.5 cm	Unstratified mix of organic materials at varying extents of decomposition
			A	7 cm	Mineral soil, darker from SOM
			Bw	27 cm	Lighter color; loose structure

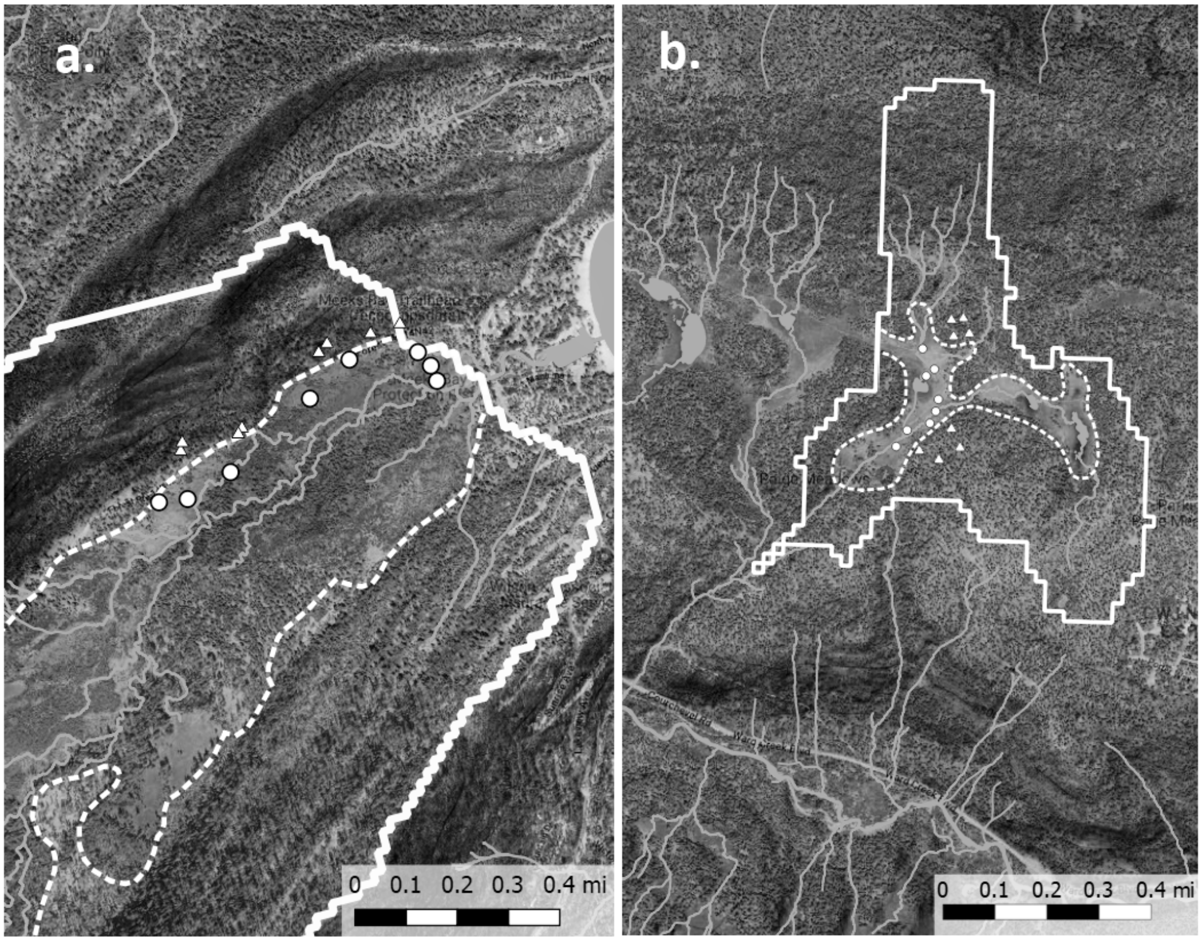


Figure 2.1: Granitic watershed at Meeks Meadow (a) and the andesitic watershed at Page Meadow (b). Solid lines are watershed boundaries. Dashed lines separate forest soils from meadow soils, based on map units from the SSURGO database. Circles are meadow sampling sites and triangles are forest sampling sites.

Soils were sampled in June, August, and October to capture the effect of seasonality from early summer through mid-fall. This spans the four months following summer solstice, which represents a hydrologic time period between the end of spring snowmelt and before significant snowfall when very little precipitation falls in the basin. The objective of sampling was to establish a meadow versus forest treatment effect on the study soils. Because the two watersheds in this study were formed in contrasting parent materials, this was accounted for as a second treatment. At each unique geology-ecosystem type combination, a total of eight representative sampling sites were identified. Spatial distribution

Table 2.2: Mean annual, mean summer, and mean winter volumetric soil moisture and temperature at Tahoe City and Rubicon SNOTEL Stations during Calendar Year 2018.

Percent (%)			Degrees Celsius (°C)		
	Tahoe City	**Rubicon**		**Tahoe City**	**Rubicon**
MASM	24.3	19.7	**MAST**	7.5	6.0
MSSM	15.7	15.1	**MSST**	12.8	10.6
MWSM	34.6	24.2	**MWST**	3.0	2.4

of sampling sites within each watershed are displayed in Figure 2.1.

In the field, the distinction of meadow versus forest soils was based on contrasting soil morphology related to the taxonomic classification in the SSURGO database. Profile descriptions for a subset of sampling sites in each watershed are listed in Table 2.1. Meadow soils in both watersheds are mapped as Inceptisols with aquic conditions for some time in normal years (USDA Natural Resources Conservation Service, 2007). Meadow soils in these watersheds had darker chroma of 1 compared to chroma between 2 and 3 in the forest soil. The taxonomy of these soils include subgroups Cumulic Humaquept at Page Meadow and Cumulic Humaquept and Aquic/Oxyaquic Dystroxerept at Meeks Meadow. The difference between these subgroups is a higher seasonal water table and an epipedon thick enough to qualify as either mollic or umbric in Cumulic Humaquepts. The Aquic and Oxyaquic Dystroxerepts at Meeks have slightly deeper water tables and dark ochric epipedons (approximately 15 centimeters) that verge on meeting the thickness requirement of a mollic or umbric epipedon. The forest soils surrounding Page Meadow are mapped as Humic Vitrixerands. The forest soils at Meeks are mapped as Humic Dystroxerepts.

2.2.2 SAMPLE HANDLING

During sampling, soils were stored in a cooler and kept on ice until returning to the main research station. Samples were stored undried in re-sealable plastic bags at 4 °C until analysis. Soils were passed through a 2-mm sieve. Some studies dry the soil prior to

P analysis, however it has been observed that drying and rewetting of soils upon analysis may lead to additional solubilization of P, particularly from the microbial biomass (Daly & Casey, 2005; Turner & Haygarth, 2001). Therefore, samples were not oven dried prior to analysis and P extractions were done within four weeks of collection.

2.2.3 Laboratory Analyses

Phosphorus Measurements

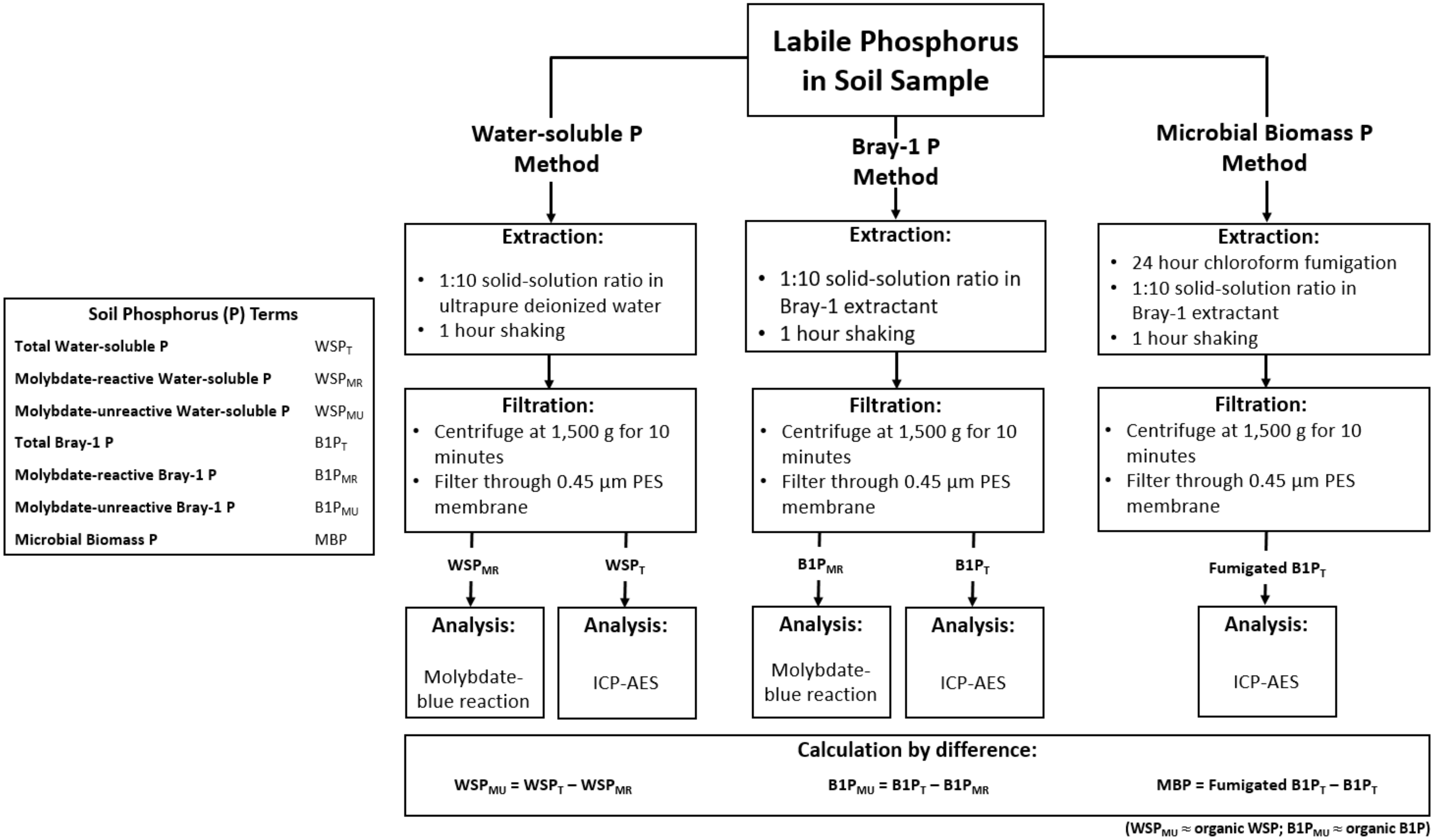
A subset of samples (n = 3) from granitic meadow, granitic forest, andesitic meadow, and andesitic forest sites were analyzed for soil total phosphorus (TP) using a multi-acid digestion at Bureau Veritas (Vancouver, BC), an ISO/IEC 17025 accredited laboratory. The sample was heated in HNO_3-$HClO_4$-HF, taken to dryness, the residue was dissolved in HCl, and then analyzed for TP on an ICP-MS.

Separate subsamples were taken to measure three pools of labile soil phosphorus, which included water-soluble, Bray-1, and microbial biomass (Figure 2.2). Equivalent soil dry-masses were measured on wet samples to account for mass of moisture in undried samples for each extraction. Soils were extracted for water-soluble P (WSP) in a 1:10 solid-solution ratio of 18-megaohm deionized water, shaken for 1 hour, centrifuged at 1,500 g for 10 minutes, and filtered through 0.45-µm diameter polyether sulfone (PES) membrane filters (Kovar & Pierzynski, 2009; Kuo, 1996). An aliquot was subsampled from the extract to estimate the inorganic fraction defined as the P in solution that is measured using the Murphy-Riley molybdate colorimetric test on a spectrophotometer (Kovar & Pierzynski, 2009; Murphy & Riley, 1962). Although this method is an estimation of inorganic orthophosphate in solution, it may also measure other molybdate-reactive forms, although they are typically much less than the orthophosphate in natural waters (Sharpley & Haygarth, 2000; Worsfold et al., 2016). Therefore, we hereafter refer to the collective fraction determined by the Murphy-Riley method as molybdate-reactive water-soluble P (WSP_{MR}). The total P in the extract (WSP_T) was analyzed on an ICP-

AES. The difference between the WSP_T and WSP_{MR} fractions is operationally defined as molybdate-unreactive WSP (WSP_{MU}), which may be associated with organic, non-hydrolyzable, and/or colloidal forms (Haygarth et al., 1997; Sharpley & Haygarth, 2000).

Soils were extracted for Bray-1 P (B1P) in a 1:10 solid-solution ratio of 0.025 M HCl and 0.03 M NH_4F solution, shaken for 1 hour, centrifuged at 1,500 g for 10 minutes, and filtered through 0.45-µm diameter PES membrane filters (Kovar & Pierzynski, 2009). Extraction time was increased from 5 minutes to 1 hour because extraction efficiency was observed to be stable between 1 and 3 hours of shaking time in these soils. An aliquot was subsampled from the extract to measure the molybdate-reactive fraction ($B1P_{MR}$) using a modified molybdate reagent for the Bray-1 chemical matrix and colorimetric analysis on a spectrophotometer (Beegle, 1998). $B1P_{MR}$ is an estimate of inorganic orthophosphate that was adsorbed to the soil, but the extract may also contain P that was hydrolyzed from polyphosphates and organic forms to some extent (Miller & Arai, 2017; Sharpley & Haygarth, 2000). The total P in the extract ($B1P_T$) was analyzed on an ICP-AES. The difference between $B1P_T$ and $B1P_{MR}$ is operationally defined as molybdate-unreactive B1P ($B1P_{MU}$), which includes organic phosphorus and other P forms that were not hydrolyzed by the Bray-1 extractant.

Microbial biomass P (MBP) was measured by treating a 1 g dry-mass equivalent subsample of each soil with 1 ml of chloroform, placed under vacuum with a beaker of approximately 30 ml of chloroform, allowed to evaporate for 24 hours, and extracted for Bray-1 P (Reddy et al., 2013; Voroney et al., 2008). Microbial biomass was calculated as the difference between chloroform fumigated and unfumigated samples. Bray-1 was observed to more accurately approximate microbial biomass P in acidic soils (Oberson et al., 1997; Wu et al., 2000). Furthermore, Wu et al. (2000) explains that using additions of inorganic P is not an accurate method to estimate a recovery coefficient for the sorption of P from lysed microbial biomass. Therefore, a recovery coefficient was not used to correct potential underestimations of microbial biomass P in the Lake Tahoe Basin soils.



Soil Phosphorus (P) Terms	
Total Water-soluble P	WSP_T
Molybdate-reactive Water-soluble P	WSP_{MR}
Molybdate-unreactive Water-soluble P	WSP_{MU}
Total Bray-1 P	$B1P_T$
Molybdate-reactive Bray-1 P	$B1P_{MR}$
Molybdate-unreactive Bray-1 P	$B1P_{MU}$
Microbial Biomass P	MBP

$WSP_{MU} = WSP_T - WSP_{MR}$

$B1P_{MU} = B1P_T - B1P_{MR}$

$MBP = \text{Fumigated } B1P_T - B1P_T$

($WSP_{MU} \approx$ organic WSP; $B1P_{MU} \approx$ organic B1P)

Figure 2.2: Fractionation scheme and terms for extractable pools of labile P in soil samples; adapted from Sharpley & Haygarth (2000)

Composite forest O-horizon samples from each watershed were ground and passed through a 2-mm sieve, extracted for water-soluble P in a 1:50 solid-solution ratio of 18-megaohm deionized water, shaken for 1 hour, centrifuged at 4,000 g for 15 minutes, and then filtered through a 0.45-µm PES membrane. An aliquot was subsampled from the filtered extract to estimate the inorganic fraction, defined as the P in solution that is measured using the molybdate-colorimetric test on a spectrophotometer. Total P in the filtered extract was analyzed on an ICP-AES.

In a subset of samples, total, inorganic, and organic fractions of soil TP was measured using the Saunders and Williams ignition method (Cade-Menun & Lavkulich, 1997). Duplicate 0.5 g subsamples of oven-dried soil were weighed and extracted in 1:60 solid-solution ratio of 1 N H_2SO_4, shaken overnight (approximately 16 hours), centrifuged at 1,500 g for 15 minutes, and supernatant was decanted to be analyzed on an ICP-AES. Prior to extraction, one of the duplicate 0.5 g subsamples was ignited by raising to 550 C over a 2 hour period, and maintained at this temperature for an additional 1 hour. Total phosphorus was equal to concentrations from the ignited sample, and organic phosphorus was estimated by the difference between ignited and unignited samples.

A subset of samples were also selected for P-31 NMR analysis to identify concentrations and speciation of organic P in the soils. Two October samples from each of the four site types were selected for analysis. Additionally, one granitic meadow site and one granitic forest site were also analyzed for June and August samples. Following standard extraction procedures for P-31 NMR, 2 g dry-mass equivalent undried soil subsamples were weighed, extracted in 25 ml of an alkaline solution of 0.5 M NaOH and 0.1 M EDTA, shaken for 4 hours, centrifuged at 1,500 g for 20 minutes, and supernatant was decanted and freeze-dried (Cade-Menun & Preston, 1996). A 1 ml aliquot of extractant was taken from each sample, diluted 1:10 with deionized water, and analyzed by ICP-AES for total P concentration. P-31 NMR spectroscopy was conducted University of Idaho's Department of Chemistry. Freeze-dried sample powder (<1 g) was dissolved in 0.9 ml of NaOH-

EDTA solution and 0.1 D_2O, and placed in a 5-mm NMR tube. The generated spectra were analyzed by Barbara Cade-Menun, Ph.D., Agriculture and Agri-food Canada, Swift Current, Saskatchewan.

Soil Characterization

All samples collected in October were measured for pH, approximate percent sand, total organic carbon, total nitrogen, and oxalate-extractable iron, aluminum, silicon, and phosphorus. A subset of 15 samples from June and the same 15 samples from August were also measured for pH. The pH of soils was measured with the 1:1 method using undried soils in 18-megaohm deionized water. Percent sand by mass was approximated by sieving the soils to measure particle size fraction greater than 63-µm . An elemental CNS analyzer was used to measure TOC and TN. Soils were extracted for poorly crystalline iron and aluminum oxides in a 1:50 solid-solution ratio of 0.2 M ammonium oxalate solution in darkness, shaken for 4 hours, allowed to settle overnight, centrifuged at 1,500 g for 30 minutes, and filtered through 0.22-µm diameter PES membrane filter (USDA, 2014). Extracts were analyzed on an ICP-AES for iron, aluminum, silicon, and phosphorus.

2.2.4 Statistical Analyses

Analysis of seasonal labile P changes was conducted using linear mixed-effects models for random effects of a repeated measures experimental design (‘nlme’ package version 3.1-137; R Core Team, 2018). Labile P status (WSP, B1P, and MBP) at each of the four sites (granitic meadow, granitic forest, andesitic meadow, and andesitic forest) during a given sampling month represented the fixed effect, and random effects originate from variability in P observations at individual sampling sites across sampling months. An autocorrelation structure of order 1 was included in the models to account for repeated measures. Pairwise comparisons of estimated marginal means of the mixed-effects models were done using Tukey’s Honest Significant Difference test for $p < 0.05$ (‘emmeans’ package version 1.3.5.1; Lenth, 2019). All other comparisons were tested for significance by fitting the data to analysis of variance (ANOVA) linear models using the ‘stats’ package in R (R Core Team, 2018). Tukey’s honest significance difference (HSD) test was used for assessing statistical differences ($p < 0.05$) between treatment means. Pearson’s correlation coefficients for $p < 0.05$ were used to evaluate the strength of relationships between several forms of soil P (‘stats’ package, R Core Team, 2018).

2.3 RESULTS

2.3.1 SOIL CHARACTERIZATION

The soil profile horizons are described in Table 2.1 in Methods section 2.2.1. At the granitic sites, O-horizons in the forest had a typical thickness of approximately 4 cm. Forest O-horizons at the andesitic sites ranged from 1.5 – 2 cm in thickness. Generally, the meadow A-horizons of both watersheds had darker moist colors, with chroma of 1, compared to typical forest A-horizon moist colors with chroma of 2 or 3. Mineral soil physicochemical properties are summarized in Table 2.3. Lower average pH was observed in meadow soils of both granitic and andesitic watersheds (5.38 and 5.35, respectively) compared to forest soils. The andesitic forest was slightly less acidic than meadows (average pH = 5.49, not statistically different from meadows), while average pH of the granitic forest soil was 5.84 and significantly higher than either meadow ($p <0.05$). Average sand content was similar in granitic meadow, granitic forest, and andesitic meadow (84, 87, and 88%, respectively), but it was significantly lower (75%, $p <0.05$) in the andesitic forest soils. In addition to having more clay and silt, the andesitic forest soils contain the highest level of poorly-crystalline iron (Fe) and aluminum (Al) oxide content measured by oxalate extraction, which is consistent with Andisol classification and mapping in this area by the USDA NRCS. Oxalate-extractable Fe and Al are not significantly different among the other three sites. Soil total organic carbon (TOC) displayed a significant difference between forests of the two watersheds, but not between the meadows. Granitic forest soil contained the lowest average TOC of all four sites, while andesitic forest soil had the highest overall. Average total nitrogen (TN) was observed to be approximately three times higher in meadow soils compared to forest soils of the same watershed. While this difference was notable in both watersheds, the difference was statistically significant only in the andesitic watershed (p = 0.054 for comparison of average TN between granitic forest and meadow soils).

Table 2.3: Physicochemical properties of 0-15 cm soil samples collected during October 2018. Arithmetic means displayed (n=8). Letters across a row indicate statistical differences in a soil property; same letters are not significantly different (p <0.05; TukeyHSD in R 'stats' package, version 3.5.2).

	Granitic Meadow			**Granitic Forest**			**Andesitic Meadow**			**Andesitic Forest**		
	Mean		SD	Mean		SD	Mean		SD	Mean		SD
pH	5.38	A	0.19	5.84	B	0.34	5.35	A	0.24	5.49	AB	0.30
Sand (%)	84	A	3.0	88	A	1.6	87	A	2.8	75	B	3.6
Oxalate-Al (%)	0.139	A	0.0848	0.242	A	0.0776	0.239	A	0.0343	1.65	B	0.403
Oxalate-Fe (%)	0.195	A	0.0786	0.244	A	0.0625	0.236	A	0.139	0.725	B	0.0762
Oxalate-Si (%)	0.0267	A	0.0157	0.0337	A	0.0134	0.0431	A	0.00684	0.431	B	0.151
TOC (%)	4.57	AB	2.57	2.77	A	0.986	5.19	AB	1.58	5.36	B	1.34
TN (%)	0.200	A	0.123	0.0606	A	0.0427	0.485	B	0.145	0.153	A	0.0338

Table 2.4: Average soil pH (n =3) for each site for samples collected in June, August, and October

	Granitic Meadow	**Granitic Forest**	**Andesitic Meadow**	**Andesitic Forest**
June	5.32	5.53	5.38	5.25
August	6.00	6.06	5.21	5.49
October	5.41	5.95	5.27	5.47

2.3.2 SOIL PHOSPHORUS

Organic Horizon Phosphorus

Loupe (2005) showed that Tahoe forest soil organic (O) horizons contribute soluble phosphorus to the environment through laboratory simulated precipitation and snowmelt leaching events. The O horizons in that study came from similar Jeffrey and Sugar Pine forests in a granitic watershed. In this study, the concentration of soluble P in composite O horizon samples overlying forest mineral soils was measured using a 1-hour shaking extraction with water, so results between the two studies are not directly comparable. However, both the Loupe (2005) leaching experiment and a shaking extraction of this study's granitic forest O horizon observed higher water-soluble phosphorus in minimally decomposed Oi horizons compared to the more decomposed Oe horizons. Both total and molybdate-unreactive water-soluble P in the andesitic forest Oe horizons were greater than P concentrations at the granitic forest study sites.

Table 2.5: Comparison of O-horizon soluble phosphorus content from several Tahoe forests. Loupe (2005) data comes from lab-simulated precipitation plus snowmelt leaching experiments. Granitic and andesitic forest columns list average water-soluble P via shaking extractions (experimental replicates of n = 5, RSD= 1.44 – 5.98%) of composite samples for individual Oi and Oe horizons at each forest in this study. Presence of Oi material was minimal in the Andesitic Forest.

	Loupe (2005)	**Granitic Forest**				**Andesitic Forest**			
	Total Phosphorus*	Molybdate-reactive phosphate		Total Phosphorus		Molybdate-reactive phosphate		Total Phosphorus	
	mg/kg	mg/kg	%RSD	mg/kg	%RSD	mg/kg	%RSD	mg/kg	%RSD
Oi	45.5	184	5.98	254	5.38	-	-	-	-
Oe	28.4	73.8	4.68	111	3.38	107	1.71	159	1.44

*Originally reported as inorganic P, but measured was by ICP; thus assigned as dissolved Total Phosphorus in this table

Total Soil Phosphorus

Average total soil P concentrations (TP; n = 3) in the andesitic watershed was significantly higher than the granitic watershed, but differences between forest and meadow soils within either watershed were not significant (Table 2.6, Figure 2.3).

Table 2.6: Total P (mg/kg) measurements in each of the four sites. Arithmetic means displayed. Letters within the last row indicate statistical differences in average TP across different site types. If any letters are shared, the averages are not significantly different ($p < 0.05$; Tukey HSD method in R 'stats' package, version 3.5.2).

	Granitic Meadow			**Granitic Forest**			**Andesitic Meadow**			**Andesitic Forest**		
	Total P		SD	Total P		SD	Total P		SD	Total P		SD
Site 1	470			440			930			1130		
Site 2	150			520			770			1010		
Site 3	520			300			860			1110		
Average	380	A	201	420	A	111	853	B	80.2	1083	B	64.3

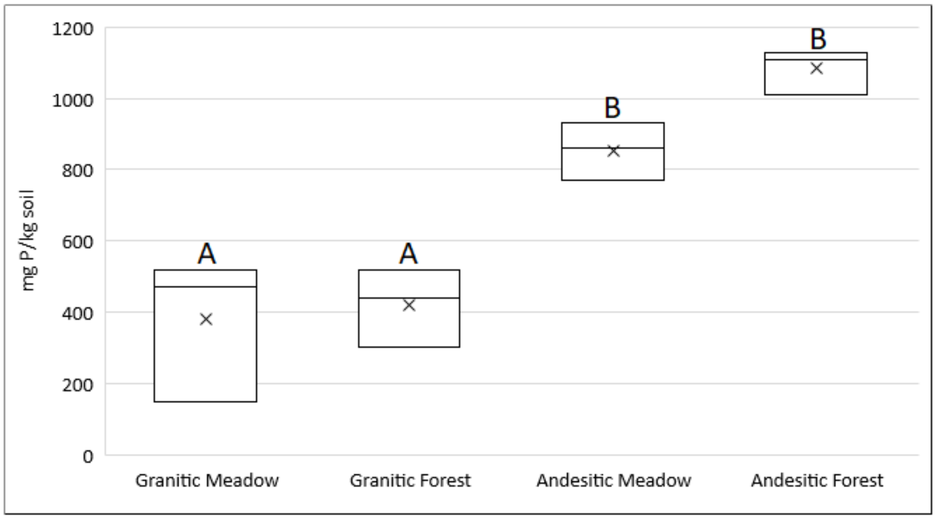


Figure 2.3: Total soil phosphorus at each site type. For each site, n = 3, top line is max, middle line is median, bottom line is minimum, and "X" is average. Non-matching letters reflect significant differences ($p < 0.05$) among site types.

Effects of Seasonality on Labile Phosphorus

WSP, B1P and MBP extractable P concentrations in soils from different seasons (Table 2.7) did not change through the seasons. The only significant differences in the phosphorus pools are in granitic forest soils and andesitic meadow soils. In granitic forest soils, WSP_{MR} reached a minimum ($p < 0.05$) in August while the organic fraction was at a significant maximum. In andesitic meadow soils, WSP_{MR} is higher in June than in August, and WSP_{MU} is higher in both June and August compared to October. WSP_{T} did not change significantly across sampling periods at any site. Seasonality did not exhibit significant effects on B1P at any site, regardless of ecosystem or parent material. Seasonality was an important factor controlling MBP in andesitic meadow soils only. In these soils, MBP was significantly different in all sampling time points, with a maximum in June and a minimum in August. Furthermore, differences in MBP between meadow and forest were most pronounced in June.

Table 2.7: Average (n = 8) seasonal observations of water-soluble, Bray-1, and microbial biomass P (expressed in mg/kg) in each of the four site types (MR = molybdate-reactive; MU = molybdate-unreactive). Estimated marginal (EM) means are displayed. Letters across a row indicate statistical differences in phosphorus during June, August, or October. If any letters are shared, the averages are not significantly different ($p < 0.05$; Tukey method in 'emmeans' R package, version 1.3.2).

A. Granitic Meadow

		June			**August**			**October**		
		Mean		SD	Mean		SD	Mean		SD
Water-soluble P	**Total**	0.777	A	0.339	0.64	A	0.528	0.396	A	0.352
	MR	0.272	A	0.356	0.04	A	0.103	0.244	A	0.259
	MU	0.490	A	0.414	0.57	A	0.433	0.201	A	0.117
Bray-1 P	**Total**	8.10	A	3.47	6.38	A	3.87	4.57	A	2.95
	MR	6.24	A	2.81	4.27	A	3.15	2.83	A	2.50
	MU	1.86	A	0.97	2.11	A	0.965	1.73	A	0.564
Microbial Biomass P	**Total**	18.6	A	12.9	18.8	A	17.7	13.1	A	7.42

B. Granitic Forest

		June			**August**			**October**		
		Mean		SD	Mean		SD	Mean		SD
Water-soluble P	**Total**	2.06	A	0.745	1.92	A	0.990	1.68	A	0.760
	MR	1.32	A	0.684	0.38	B	0.263	1.18	A	0.742
	MU	0.637	A	0.240	1.71	B	0.748	0.43	A	0.190
Bray-1 P	**Total**	104.0	A	45.4	103.1	A	40.5	112.8	A	68.3
	MR	86.8	A	36.0	85.7	A	35.1	97.0	A	63.3
	MU	19.5	A	9.84	17.5	A	6.00	15.8	A	13.2
Microbial Biomass P	**Total**	0.00	A	0.00	7.83	A	5.77	8.37	A	3.77

C. Andesitic Meadow

		June			**August**			**October**		
		Mean		SD	Mean		SD	Mean		SD
Water-soluble P	**Total**	1.73	A	3.15	1.59	A	0.678	1.35	A	0.643
	MR	0.69	A	1.08	0.16	B	0.120	0.35	AB	0.297
	MU	1.57	A	2.08	1.33	A	0.562	0.59	B	0.364
Bray-1 P	**Total**	9.10	A	2.52	11.5	A	4.58	10.3	A	4.38
	MR	5.84	A	1.60	6.91	A	2.83	4.99	A	1.90
	MU	2.53	A	1.39	4.64	A	2.57	5.35	A	2.98
Microbial Biomass P	**Total**	38.7	A	24.6	17.6	B	6.86	28.1	C	10.2

D. Andesitic Forest

		June			**August**			**October**		
		Mean		SD	Mean		SD	Mean		SD
Water-soluble P	**Total**	0.40	A	0.114	0.27	A	0.227	0.02	A	0.216
	MR	0.10	A	0.0344	0.00	A	0.000	0.10	A	0.047
	MU	0.14	A	0.0919	0.19	A	0.227	0.18	A	0.157
Bray-1 P	**Total**	6.73	A	2.91	3.81	A	1.89	6.50	A	3.92
	MR	3.50	A	1.96	1.64	A	1.38	2.87	A	2.60
	MU	3.22	A	1.08	2.17	A	0.553	3.62	A	1.43
Microbial Biomass P	**Total**	6.56	A	2.91	2.08	A	0.916	1.86	A	1.46

Table 2.8: Average soil labile P concentrations (mg/kg; n = 24) observed across all sampling months. Estimated marginal (EM) means are displayed. Capital letters within rows indicate statistical differences in P between the four site types. Lower case letters within columns represent statistical differences in total WSP, B1P, and MBP. If any letters are shared, the averages are not significantly different ($p < 0.05$; Tukey method in 'emmeans' R package, version 1.3.2).

		Granitic Meadow			**Granitic Forest**			**Andesitic Meadow**			**Andesitic Forest**		
		Mean		SD	Mean		SD	Mean		SD	Mean		SD
Water-soluble P	Total	0.604	ABa	0.419	1.89	Ca	0.828	1.56	BCa	1.90	0.230	Aa	0.187
	MR	0.188	A	0.290	0.961	B	0.713	0.399	A	0.663	0.0752	A	0.0581
	MU	0.420	AB	0.370	0.927	AC	0.726	1.16	C	1.28	0.170	B	0.163
Bray-1 P	Total	6.35	Aa	3.61	106	Bb	50.9	10.3	Ab	4.06	5.68	Ab	3.19
	MR	4.45	A	3.06	89.8	B	44.9	5.91	A	2.23	2.67	A	2.10
	MU	1.90	A	0.830	17.6	B	9.78	4.17	A	2.62	3.01	A	1.21
Microbial Biomass P	Total	16.8	Ab	13.0	5.36	Ba	5.41	28.1	Cc	17.1	3.50	Bb	2.89

Effects of Parent Material on Labile Phosphorus

Major significant differences were not observed in water soluble phosphorus (WSP) or Bray-1 P (B1P) between meadows of the two watersheds (Table 2.8). Of these two pools of soil P, only $\mathrm{WSP}_{\mathrm{MU}}$ was significantly higher in andesitic meadows than in granitic meadows.

Labile P in the forested soils of the two watersheds demonstrated significant differences in response to the geology (Table 2.8). Average $\mathrm{WSP}_{\mathrm{T}}$, $\mathrm{WSP}_{\mathrm{MR}}$, and $\mathrm{WSP}_{\mathrm{MU}}$ were significantly different between the two watersheds, with $\mathrm{WSP}_{\mathrm{T}}$ in the granitic forest soils approximately eight times higher than that of the andesitic forest (Figure 2.4, Table 2.8). Differences in $\mathrm{B1P}_{\mathrm{T}}$, $\mathrm{B1P}_{\mathrm{MR}}$, and $\mathrm{B1P}_{\mathrm{MU}}$ were also significant, and forest soils in the granitic watershed were approximately nineteen times higher in average $\mathrm{B1P}_{\mathrm{T}}$ than the andesitic watershed (Figure 2.4, Table 2.8).

Soil MBP was significantly different in meadows, but not between forests of the two watersheds (Figure 2.4, Table 2.8). MBP in meadows was higher in the andesitic site compared to the granitic site.

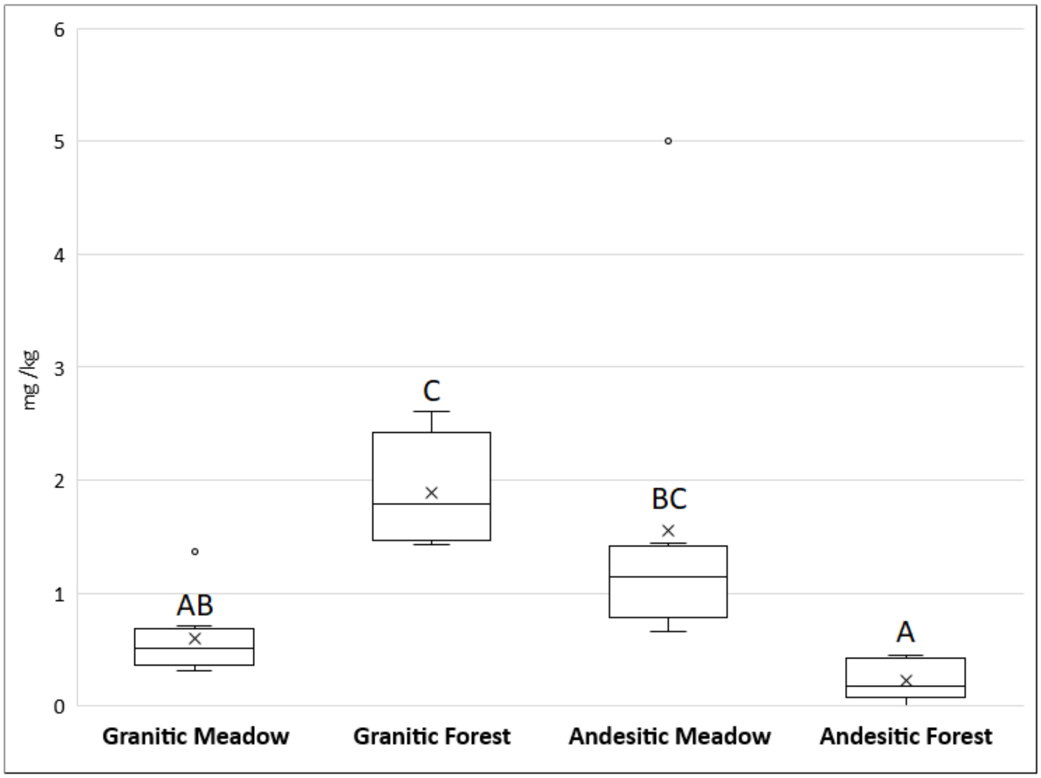


Figure 2.4: Total water-soluble phosphorus across each site type. Capital letters (from Table 2.8) indicate statistical differences between sites; same letters are not different. Boxes are the interquartile range (IQR; middle 50% of data), whiskers are upper and lower 25% quartiles, middle lines are medians, "X" markers indicate averages, and outliers are those points falling beyond 1.5 times the upper/lower IQR limits.

Effects of Ecosystem Type on Labile Phosphorus

Ecosystem type (meadow versus forest) showed significant impact on MBP status in both watersheds (Figure 2.5). Meadow soils contained significantly more MBP than neighboring forest soils in each respective watershed. The response of WSP to ecosystem type was not the same between the two watersheds (Table 2.8, Figure 2.4). Significantly elevated $\mathrm{WSP_{MU}}$ in meadow soils of the andesitic watershed contributed to higher $\mathrm{WSP_T}$ in andesitic meadows over forests, whereas the presence of significantly higher $\mathrm{WSP_{MR}}$ in granitic forest soils led to higher average $\mathrm{WSP_T}$ compared to granitic meadow soils. Considering the differences between meadows and forests in the two watersheds, ecosystem type alone does not provide the best explanation of WSP variability. $\mathrm{B1P_T}$, $\mathrm{B1P_{MR}}$, and $\mathrm{B1P_{MU}}$ were only different in meadow and forest samples from the granitic site (Table 2.8).

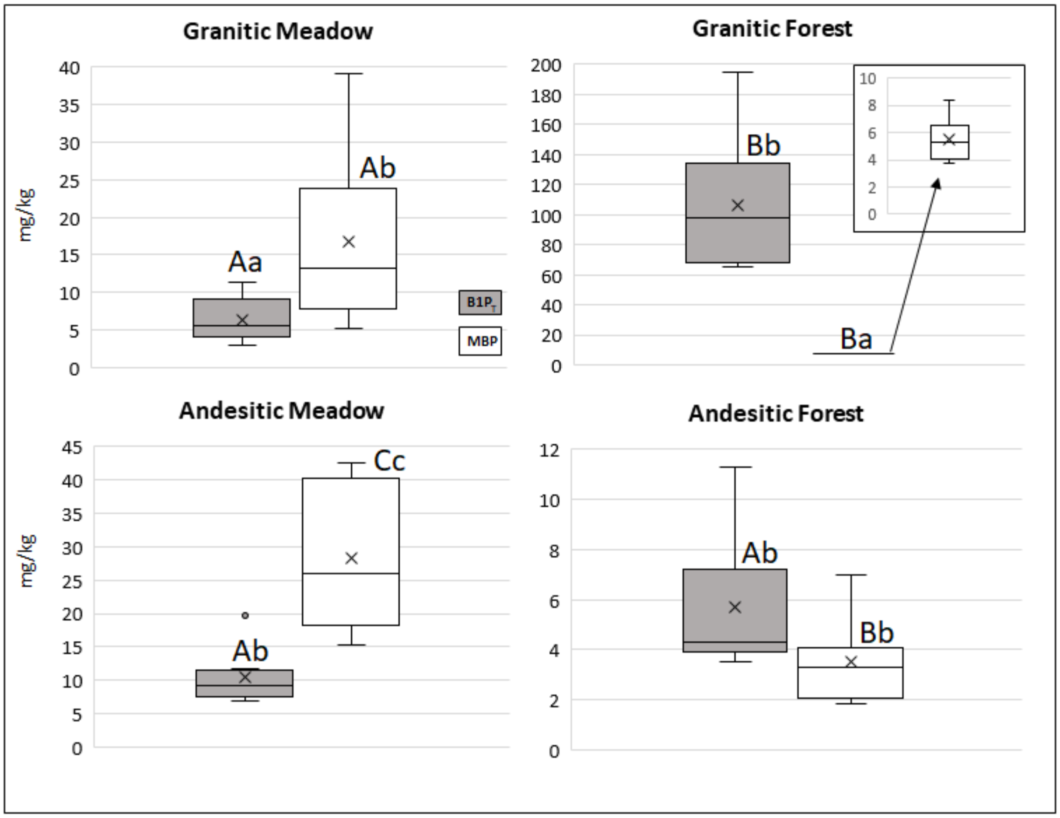


Figure 2.5: Total Bray-1 and microbial biomass phosphorus across each site type. Capital letters indicate statistical differences between sites, for a given pool of labile P; same letters are not different. Lower case letters indicate differences between B1P and MBP, within sites **only**.

In the granitic watershed, forest soils contained significantly more $B1P_T$ than meadows (Figure 2.5). MBP was significantly higher than B1P in meadows at both watersheds. In the forests, B1P was higher than MBP, but the difference was only significant in the granitic watershed.

Relationships between Labile Phosphorus Pools

Relationships between total and molybdate-reactive fractions across all three labile P pools (total only for microbial biomass) were assessed for any significant correlations. WSP_{MR} was correlated with $B1P_T$ and $B1P_{MR}$ (Figure 2.6). WSP_{MR} was more strongly correlated with B1P than WSP_T. Pearson's r values ranged from 0.65 (p-value <0.001) between WSP_{MR} and $B1P_T$, and 0.63 between WSP_{MR} and $B1P_{MR}$. The correlations between the molybdate-unreactive fractions of WSP and each of the B1P fractions were not significant. MBP was not correlated to WSP or B1P.

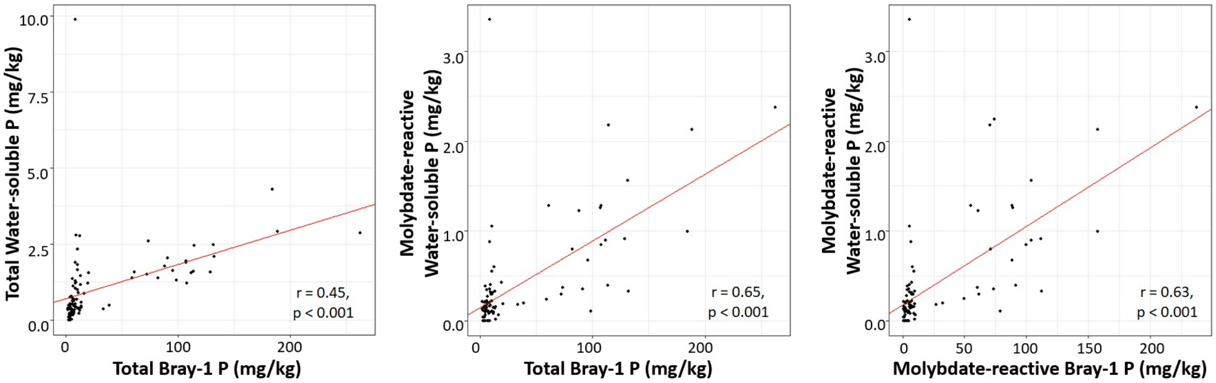


Figure 2.6: Correlation of total and molybdate-reactive water-soluble P with total and molybdate-reactive Bray-1 P among all sites and seasons.

Molybdate-reactive forms dominated B1P in all soils except those in the andesitic forest (Table 2.9). $B1P_{MR}$ comprised only 43% of the total pool in those soils. Despite the presence of predominantly molybdate-reactive P in the Bray-1 pool, molybdate-unreactive forms tended to comprise the majority of the water-soluble pool. Molybdate-unreactive forms dominate WSP in all soils, except in granitic forest soils, where average molybdate-reactive and -unreactive forms both evenly make up 50% of the pool.

Table 2.9: Relative percentages of molybdate-reactive and -unreactive P making up WSP and B1P pools. Percent fractionation into molybdate-reactive and -unreactive forms was determined using the arithmetic mean of each site, and omitting observations where zero phosphorus was detectable (three WSP measurements of andesitic forest soils in August).

	Water-soluble P		**Bray-1 P**	
	MR	MU	MR	MU
Granitic Meadow	30%	70%	65%	35%
Granitic Forest	50%	50%	83%	17%
Andesitic Meadow	25%	75%	59%	41%
Andesitic Forest	30%	70%	43%	57%

Table 2.10: Fractionation of TP with Saunders and Williams method (Cade-Menun & Lavkulich, 1997). TP is soil P in 1 N H_2SO_4 after ignition, inorganic P is soil P in 1 N H_2SO_4, and organic P is the calculation by difference of total and inorganic fractions.

Sites		Total Phosphorus	Inorganic Phosphorus		Organic Phosphorus	
		mg/kg	mg/kg	%	mg/kg	%
Granitic	Site 1	273	67.5	25	206	75
Meadow	Site 2	218	59.0	27	159	73
Granitic	Site 1	550 (12)	510 (3.8)	93	39.6	7.0
Forest	Site 2	371 (1.1)	282 (0.10)	76	89.6	24
Andesitic	Site 1	1205	368	31	838	69
Meadow	Site 2	560	116	21	444	79
Andesitic	Site 1	574	393 (1.9)	69	181	31
Forest	Site 2	988 (0.0)	799	81	189	19

Percent relative standard deviations (% RSD) for replicate analysis (n = 2) of select samples are displayed in parentheses next to concentrations; RSD was less than 2% in replicate analysis for all samples except those from granitic forest site 1.

Forms of Soil Organic Phosphorus in the Granitic Watershed

Determination of total inorganic and organic P in soil using the Saunders and Williams method in 1 N H_2SO_4 with ignited and unignited subsamples (Cade-Menun & Lavkulich, 1997) was conducted for two October samples from each site type (eight samples total; Table 2.10. Across both meadows, organic P made up 69 – 79% of the total extractable P with this method. In the forest soils, organic P was lower, ranging from 7 – 31% of the total extractable P. To gain further understanding into the speciation of P in our soils, one granitic meadow (site 1) and one granitic forest soil (site 2) from June, August, and October extracted with NaOH-EDTA and analyzed by P-31 NMR spectroscopy.

The NaOH-EDTA extraction efficiency ranged from 36-62% of total soil phosphorus extracted by 1 N H_2SO_4 (Table 2.11). The P not extracted by NaOH-EDTA is considered to be bound in minerals, and not readily available to the soil solution or biological cycling (Cade-Menun et al., 2015). Extractable P in 1 N H_2SO_4 (via ICP-AES) and NaOH-EDTA (via P-31 NMR spectroscopy) were significantly correlated, for both inorganic and organic fractions (Figure 2.7), suggesting that recovery rates by NaOH-EDTA were proportionate to total P by 1 N H_2SO_4. The average of June, August, and October 1 N H_2SO_4 extractable P was 75% organic in one granitic meadow soil (Table 2.11). In a granitic forest soil, the H_2SO_4 extractable P was on average 76% inorganic. Organic forms of phosphorus comprised 78% of the NaOH-EDTA extractable P in the soil from the granitic meadow averaged across June, August, and October, as determined by P-31 NMR spectroscopy (Table 2.12). In the granitic forest soil, approximately 74% of NaOH-EDTA extractable P existed as inorganic orthophosphate.

Table 2.11: Total P in NaOH-EDTA and 1 N H_2SO_4 extractions to supplement P-31 NMR analysis of organic species in a granitic meadow soil and a granitic forest soil in June, August, and October.

	NaOH-EDTA	**Total P**	**Total P recovery by NaOH-EDTA**	**Total Inorganic P**		**Total Organic P**	
	(mg/kg)	(mg/kg)	(%)	(mg/kg)	(%)	(mg/kg)	(%)
Granitic Meadow							
June	183	362	51%	85.7	24%	276	76%
August	117	294	40%	77.0	26%	217	74%
October	98.6	273	36%	67.5	25%	206	75%
Granitic Forest							
June	257	416	62%	346	83%	70.2	17%
August	276	508	54%	347	68%	161	32%
October	205	371	55%	282	76%	89.6	24%

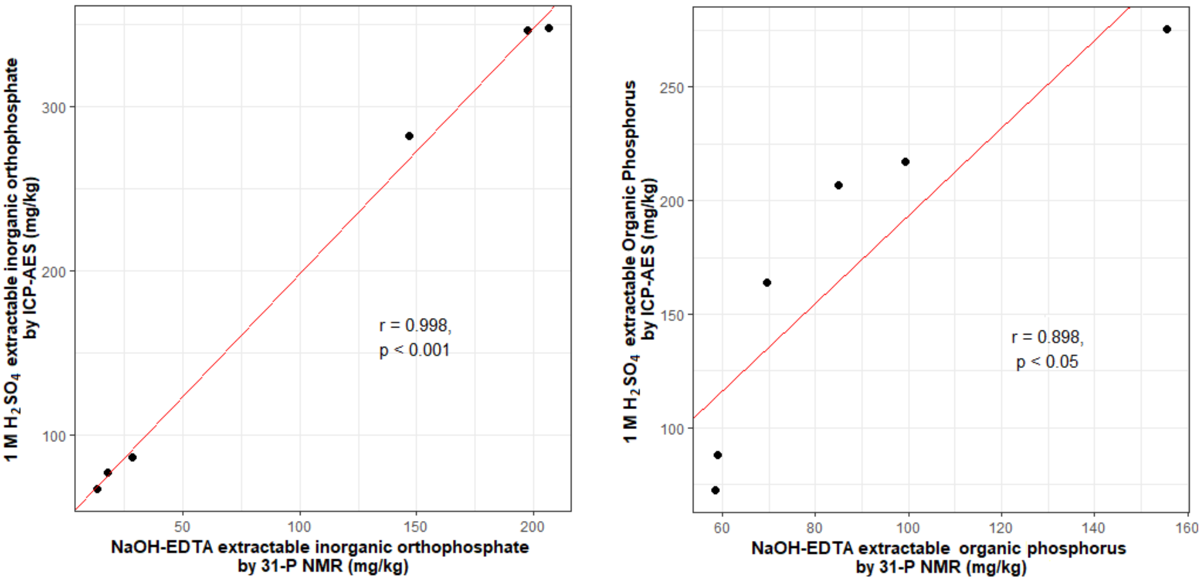


Figure 2.7: Correlations of both inorganic (left) and organic (right) 1 N H_2SO_4 extractable phosphorus and NaOH-EDTA extractable phosphorus, determined by ICP-AES and NMR spectroscopy, respectively.

In the granitic meadow soil, the largest group of organic phosphorus compounds is orthophosphate monoesters (averaging 50% of all NaOH-EDTA extractable P across June, August, and October; Table 2.12). Orthophosphate diesters made up the next largest group (25%). Monoester phosphates are phosphates bonded to one ester compound, while diester phosphates are those bonded to two ester groups (Condron et al., 2005). The average of June, August, and October phosphonates was 2.37% of NaOH-EDTA phosphorus in the meadow soil. The amounts of organic P species were the same across June, August, and October (14%, 6%, and 2% for monoesters, diesters, and phosphonates, respectively) in the granitic forest soil.

Relationships between the main groups of P-NMR species with both WSP_{MU} and MBP were assessed for significant correlations (Table 2.13). Phosphonates exhibited the strongest correlation with WSP_{MU} (Pearson's $r = 0.8$, $p = 0.054$). Phosphonates were not correlated to MBP. Phosphate diesters were significantly correlated to MBP ($r = 0.86$, $p < 0.05$), but not WSP_{MU}.

Table 2.12: Concentrations of inorganic and organic phosphorus species as determined by P-31 NMR spectroscopy in a meadow and forest soil from each sampling period collected from the granitic watershed.

	Inorganic Phosphorus				**Organic Phosphorus**						
	Polyphosphates		**Orthophosphate**		**Phosphate Monoesters**		**Phosphate Diesters**		**Phosphonates**		**Total NaOH-EDTA extractable P**
	mg/kg	%	mg/kg	%	mg/kg	%	mg/kg	%	mg/kg	%	mg/kg
Granitic Meadow											
June	5.12	2.80	29.3	16.0	98.2	53.7	45.2	24.7	5.12	2.8	183
August	13.1	11.2	17.2	14.7	54.0	46.0	31.3	26.7	1.64	1.4	117
October	6.41	6.50	14.1	14.3	50.4	51.1	24.8	25.2	2.86	2.9	98.6
Average	8.22	6.83	20.2	15.0	67.5	50.3	33.8	25.5	3.21	2.37	133
Granitic Forest											
June	8.73	3.40	198	77.0	35.4	13.8	11.3	4.4	3.59	1.4	257
August	7.74	2.80	207	74.8	40.1	14.5	14.1	5.1	7.74	2.8	276
October	6.97	3.40	146	71.2	30.8	15.0	18.5	9	2.87	1.4	205
Average	7.81	3.20	176	74.3	35.4	14.4	14.6	6.17	4.73	1.87	246

Table 2.13: Pearson's r correlations and corresponding p-values listed for the relationship between P-31 NMR identified organic species and both WSP$_{\mathrm{MU}}$ and MBP.

	Molybdate-unreactive Water-soluble P (WSP$_{\mathrm{MU}}$)		**Microbial Biomass P (MBP)**	
	Pearson's r	p-value	Pearson's r	p-value
Pyrophosphates	-0.01	0.99	-0.79	0.062
Polyphosphates	-0.36	0.48	-0.63	0.18
Phosphonates	0.80	0.054	-0.59	0.22
Total Monoesters	-0.48	0.33	0.78	0.069
Total Diesters	-0.72	0.11	0.86	0.030

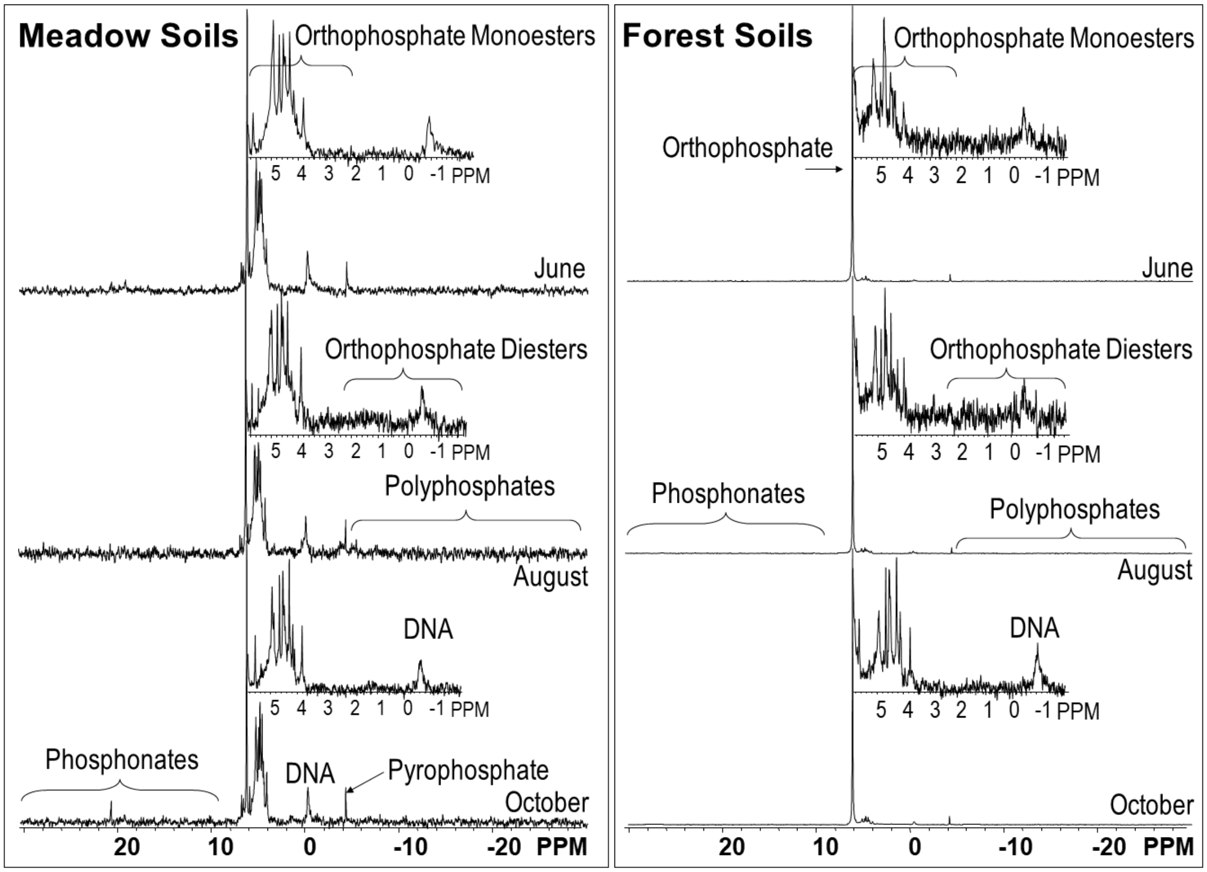


Figure 2.8: P-31 NMR spectra for granitic meadow soil and granitic forest soil in June, August, and October. Interpretation of peak speciation was done by Dr. Barbara Cade-Menun of Agriculture and Agri-food Canada.

2.4 Discussion

2.4.1 Parent Material and Soil Phosphorus

Total phosphorus in unfertilized soils is typically directly linked to phosphorus content of the parent material (Gardner, 1990; Hahm et al., 2014; Porder & Ramachandran, 2013). Although phosphorus concentration for a given rock type is variable to some degree, it can be an important factor in explaining soil P differences in the landscape, especially for soils sharing similar climates. Soil TP concentrations in our study are similar to values reported for some early pedogenic-stage soils derived from similar parent rock materials (Porder & Ramachandran, 2013). TP in both forest and meadow soils was over two times greater in the andesitic watershed soils than in the granitic watershed soils (Figure 2.3). Average TP of the Andisols in our study are similar to average TP across thirteen different natural, uncultivated andisols from a review of three studies (Yang & Post, 2011). In the same review, average TP across nine different Inceptisols in four separate studies was 490 mg/kg, similar to Inceptisols in the granitic meadow and forest. Inceptisols in the andesitic meadow were considerably higher in TP than average values reported by Yang and Post (2011), possibly due to the fact that these Inceptisols may be only slightly weathered from more highly P-concentrated andesitic glacial deposits.

Despite having lower average TP than the andesitic forest soils, the percent WSP, B1P, and MBP in the granitic forest soils is greater (Table 2.14). The expression of WSP, B1P and MBP both per unit soil mass and total soil P allows for comparisons in labile fractions to be made between the different soils (Table 2.8). Buffering capacity is the degree to which the soil can adsorb or release P from exchange sites to prevent changes in the solution concentration (Holford, 1997). Soils with larger TP reserves are considered to have more labile P and a greater buffering capacity to replenish the loss of P from solution that occurs by plant uptake or leaching (Daly et al., 2015). Although a stronger

P buffering capacity might be expected in the andesitic forest where the highest TP was observed, the presence of greater amounts of amorphous Fe- and Al-hydroxide (Table 2.3) create strong adsorption sites for the P (Khare et al., 2005), explaining why these soils had lower labile P than the granitic forest soils.

A similar phenomenon of increasing soil P availability with decreasing TP content was also described by Brödlin et al. (2019) in soils forming in similar parent materials to our study (a high TP volcanic rock and a low TP sandy glacial till). In both Brödlin et al. (2019) and our study, the low TP forest soils had more labile P than that of the high TP forest soils; they also had more sand and less Fe- and Al-hydroxides than the high TP forest soil. In our study, higher labile P in soils with lower overall TP supports the suggestion that reduced sorption capacity leads to soil P export out of the watershed, thus depleting the total P in the soils.

Table 2.14: Average (n = 3) TP, WSP, B1P, and MBP in each site normalized both per unit soil mass (mg/kg) and percent of TP.

	Total Phosphorus	**Water-soluble Phosphorus**		**Bray-1 Phosphorus**		**Microbial Biomass Phosphorus**	
	mg/kg	mg/kg	% of Total	mg/kg	% of Total	mg/kg	% of Total
Granitic Meadow	380	0.305	0.114	3.22	0.945	16.4	4.37
Granitic Forest	420	1.86	0.460	101	23.7	8.44	2.04
Andesitic Meadow	853	1.36	0.167	7.83	0.938	28.8	3.47
Andesitic Forest	1083	0.243	0.0225	7.09	0.671	1.69	0.161

Although B1P was different between forest soils of the two parent materials (Table 2.14), it was similar across both meadows. Certain ecosystem properties, such as floodplain topographic positions with high organic carbon and high seasonal water tables as seen in the granitic and andesitic meadow soils of this study, may mask the influence of mineralogy on labile P (Johnston et al., 1995; Sah et al., 1989). Sah et al. (1989) found that above 8 g/kg (0.8%), organic carbon plays a role in controlling the availability of phosphorus in soil by reaching sufficient levels to complement the reduction of ferric iron. Average TOC in the meadows from this study ranged from 4.57% in the granitic watershed to 5.19% in the andesitic watershed. During periods of flooding, the precipitation of ferrous iron-phosphate minerals such as vivianite may occur (Zhang et al., 2003; Heiberg et al., 2010; Rothe et al., 2016), which then have the potential to become less labile upon drainage through occlusion by oxidized iron-hydroxide minerals (Sah & Mikkelsen, 1986b). Long-term labile P may decrease in some flooded-drained soils as compared to corresponding unflooded soils, even after soils remain drained over four months (Sah & Mikkelsen, 1986a). Therefore, the seasonal flooding, high organic matter, and reducible-iron content (Table 2.3) in some Lake Tahoe Basin meadows may be more influential on P mobility than properties associated with different underlying parent materials.

2.4.2 ECOSYSTEM TYPE AND SOIL PHOSPHORUS

Phosphorus Forms in a Meadow Soil and Forest Soil

In one meadow soil and one forest soil from the granitic watershed, fractionation by LOI and 1 N H_2SO_4 revealed that total soil P averaged across June, August, and October was overwhelmingly 75% organic in the meadow, while the forest soil average across these months was only 24% organic. A comparison of P-31 NMR speciation in the same soils verified the predominance of organic P forms in the meadow and inorganic orthophosphate in the forest. Chiu et al. (2005) observed a similar fractionation of inorganic and organic P forms between subalpine grassland and forest soils using P-31 NMR spectroscopy.

In both the granitic meadow and forest, monoesters followed by diesters were the most common forms of soil organic P. These forms of soil organic P tend to dominate in both soil leachate and water extractions (Toor et al., 2003; Young et al., 2013). Although substantially higher concentrations of total soil organic P were observed in the meadow soil, the ratio of monoesters to diesters was approximately 2:1 in both the forest and meadow. According to Cheesman et al. (2010), preservation of this ratio from forested hillslopes to the floodplain meadows may be evidence of P transport across landscape positions. In conjunction with the increased organic P in meadows, average soil TOC was approximately 1.8% higher in the granitic meadow than the forest, but this difference was not statistically significant. These meadow characteristics along with the fact that sedimentation rates sometimes exceed mineralization rates in floodplain soils (Noe et al., 2013) raises this question: does the prevalence of organic P in meadow soils result in part from the loading and accumulation of soluble organic P over time?

Dominant Pools of Labile Soil Phosphorus Across Ecosystems

In both watersheds, we observed contrasting responses of MBP and B1P to ecosystem type (meadow versus forest). MBP was more predominant than B1P in the meadows, while the opposite was true in the forests where B1P was greater than MBP (Figure 2.5. This pattern was statistically significant across all sites except andesitic forests. Although not statistically different in the andesitic forest soil, average B1P was still notably greater than MBP in these sites ($p = 0.06$, for concentrations normalized per unit soil mass). The muted response of B1P in the andesitic forest soils may be partly due to the high P-retention capacity given their andic properties (McDaniel et al., 2005).

Availability of soil P is one factor that may govern P immobilization into microbial biomass (Olander & Vitousek, 2004; Pistocchi et al., 2018; Spohn & Widdig, 2017). Pistocchi et al. (2018) observed that during incubation of a soil with low available P, net mineralization to inorganic forms was minimal (3-4%), while a soil with higher available P had 40% net mineralization. Similar behavior occurred in the granitic watershed, where lower B1P was complemented by elevated MBP in the meadow. In the granitic forest soils, geochemical processes (adsorption and precipitation) may take precedence over immobilization by microbes when P sources saturate the soil and exceed biological demands (Olander & Vitousek, 2004). Average B1P in the andesitic meadow was comparable to that in the granitic meadow, which may explain why immobilized MBP was greater than B1P in both meadows. However, B1P in andesitic forest soil was also similar to that in both meadows, but yielded significantly lower MBP than either meadow. Therefore, limited labile P in the Andisols of the andesitic forest did not prompt elevated MBP. MBP may be inhibited by aluminum toxicity which is commonly observed in Andisols, and reduces microbial enzyme production including phosphatase, thereby limiting the capacity to contribute to cycling of soil nutrients (Kunito et al., 2016).

A difference in soil total nitrogen (TN) was also observed in response to ecosystem type in watersheds of this study (Table 2.3). Although only significant at the andesitic site, meadow soil TN was over three times higher than the adjacent forest soil in both watersheds. One mechanism of interaction between soil nitrogen and microbes is that mineralization of soil organic P for microbial uptake requires large amounts of nitrogen for the production of phosphatase enzyme (Marklein & Houlton, 2012; Vitousek et al., 2010). Thus, another explanation for the presence of significantly more MBP in both meadows than in either forest is sufficient N availability for phosphatase generation. Silvan et al. (2003) also observed higher rates of P immobilization in response to higher simultaneous additions of N and P. Therefore, if solution P from the forests in our study is transported to the meadows where soil TN is higher, the mobilized solution P may potentially undergo net immobilization to microbial biomass upon interception by the meadow soil.

2.4.3 Cycling of Labile Soil Phosphorus Pools

The fraction of soil P that is readily mobilized to surface and subsurface hydrologic pathways is accurately estimated by water extraction (McDowell & Sharpley, 2001). Therefore, the composition of soil WSP and its relationship to B1P and MBP were evaluated to understand how these labile pools may increase the magnitude of P export to surface waters in the Basin. Average WSP in soils from this study was mostly occupied by P that is not reactive with molybdate blue chemistry, which is assumed to be organic P (Worsfold et al., 2016); the granitic forest soil is an exception where molybdate-reactive and -unreactive P were each 50% of the total pool (Figure 2.9). This supports conclusions by other groups that some organic P species are an important component of soluble P, representing a fraction of soil P with a higher risk of loss to surface waters (Anderson &

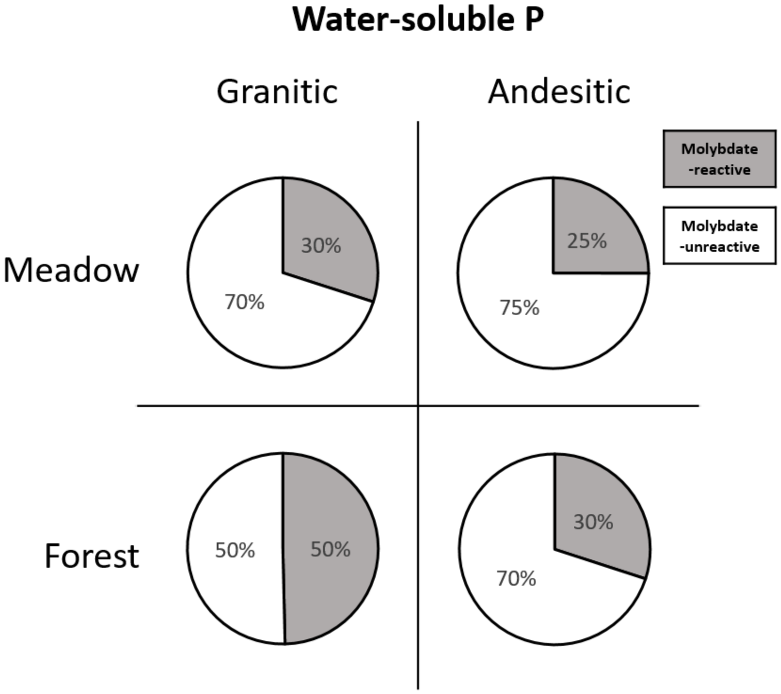


Figure 2.9: Average percentages of molybdate-reactive and -unreactive water-soluble P from each of the four sites.

Magdoff, 2005; Dodd & Sharpley, 2015; Missong et al., 2016). Therefore, $\mathrm{WSP}_{\mathrm{MU}}$ from both forest and meadow soils is potentially a mobile phosphorus source transported to surface waters in the basin.

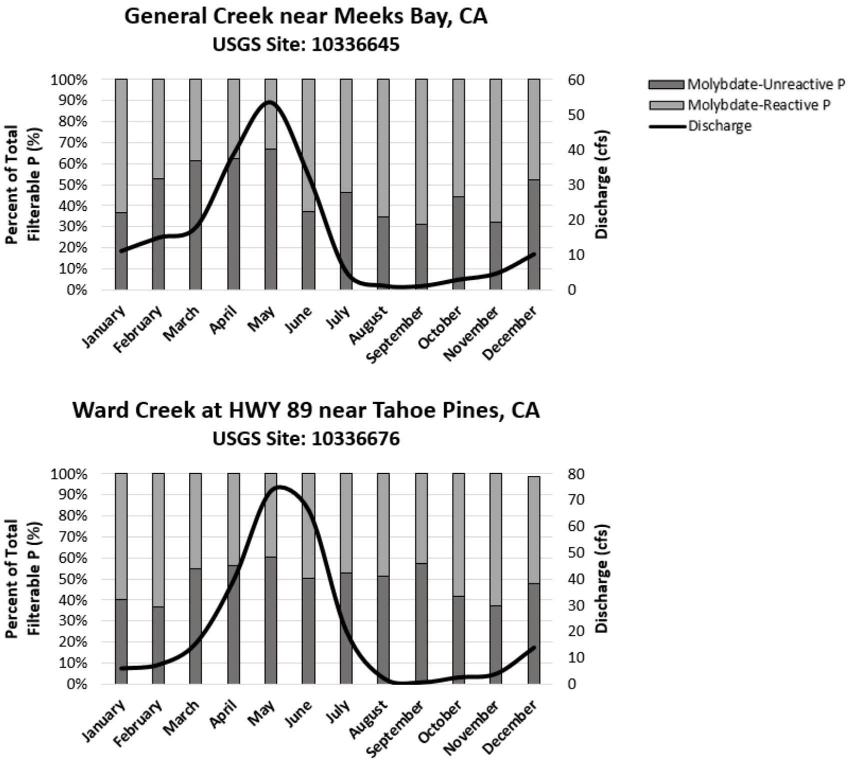
$\mathrm{WSP}_{\mathrm{MR}}$ was positively correlated with $\mathrm{B1P}_{\mathrm{MR}}$ (Figure 2.6). Strong linear relationships typically exist between tests that approximate adsorbed labile P and soil solution P (McDowell et al., 2001; Paulter & Sims, 2000). The equilibrium reactions or buffering between adsorbed P and soil solution (Herlihy & McCarthy, 2006; Weihrauch & Opp, 2018) is likely reflected by the positive relationship of $\mathrm{WSP}_{\mathrm{MR}}$ with $\mathrm{B1P}_{\mathrm{MR}}$ (Figure 2.6). While unintentional hydrolysis by the Bray extractant potentially led to increased molybdate-reactive forms (Miller & Arai, 2017), molybdate-unreactive forms still constituted 17 – 57% of average $\mathrm{B1P}_{\mathrm{MU}}$ (Table 2.9). Possibly, competition for exchange sites by some molybdate-unreactive forms such as organic inositol phosphates (stable monoester phosphate compounds) may contribute marginal rates of additional inorganic orthophosphate desorption to solution (Berg & Joern, 2006). This may explain why a slightly higher positive correlation of WSP_{MR} with $B1P_T$ was observed when compared to WSP_{MR} with $B1P_{MR}$.

Unlike the relationship of WSP_{MR} and B1P, WSP_{MU} was not correlated to either B1P or MBP. Of all measured forms of P, WSP_{MU} only exhibited a significantly positive correlation with soil phosphonates in the granitic meadow and forest soils from June, August, and October. Although soil phosphonates are commonly associated with microbes (Yu et al., 2013), phosphonates are also produced by numerous other soil organisms (Condron et al., 2005). Because phosphonates were not correlated with MBP in our soils, the source of phosphonates may not be microbial.

Water-soluble P from leaf litter is an important source of labile P return to the soil (Uselman et al., 2012). Water-soluble P from Oi and Oe horizons in our forest sites ranged from approximately 100-250 mg/kg (Table 2.5). These concentrations are two orders of magnitude above soil WSP_T, and in the andesitic forest, water-soluble P from the Oe horizon was almost 700 times higher than WSP_T. Considering the enriched status of P in water extractions of forest O horizons from both watersheds, transfer from these litter layers represents a plausible contributor of labile P to the forest's mineral soil.

2.5 Conceptual Model

In a Lake Tahoe watershed, Ohara et al. (2011) recorded more than 90–95% of field-observed hillslope drainage occurring as subsurface lateral flow through soil for rain only, snowmelt only, and rain-on-snow events. The gradual and persistent leaching of soil P to laterally flowing groundwater is an important phosphorus transport path to surface waters (Sims et al., 1998). Consequently, mobilization of labile P resulting from interaction of lateral flow with the basin's soils may be an important mechanism of P loading to surface waters, especially during periods of continuous snowmelt. Moreover, when streams near our research sites experience peak discharge, molybdate-unreactive fractions make up 61 – 67% of filterable (<0.45-µm m) P (Figure 2.10; U.S. Geological Survey, 2016). Therefore, molybdate-reactive/-unreactive fractions in these nearby streams during snowmelt more closely reflect the fractionation of soils (50 – 75% $\mathrm{WSP_{MU}}$; Figure 2.9) than O-horizons (28 – 36% $\mathrm{WSP_{MU}}$ based on our exploratory composite sampling; Table 2.5). One possibility is that inorganic or molybdate-reactive forms of P in nutrient laden runoff and interflow from forest O-horizons is attenuated by forest and meadow soils in some watersheds leading to net exports of organic forms. An alternative explanation is that lateral flow through soils is a larger source to streams in the basin than surface runoff from O-horizons. The extractable labile pools evaluated in this study offer insight into what forms of soil P are interacting with shallow groundwater or snowmelt moving through the watersheds' soils in contrasting areas of the Lake Tahoe Basin landscape. A conceptual model of labile P distribution and potential watershed transport based on observations from this study is shown in Figure 2.11.



General Creek near Meeks Bay, CA

USGS Site: 10336645

Ward Creek at HWY 89 near Tahoe Pines, CA

USGS Site: 10336676

Figure 2.10: Monthly average discharge and percent molybdate-reactive/-unreactive fractions of total filterable P for General Creek (years 2008-2018) and Ward Creek (2008-2014). Our andesitic sites drain to Ward Creek, while General Creek is a separate neighboring drainage, on the north side of the ridge at our granitic sites.

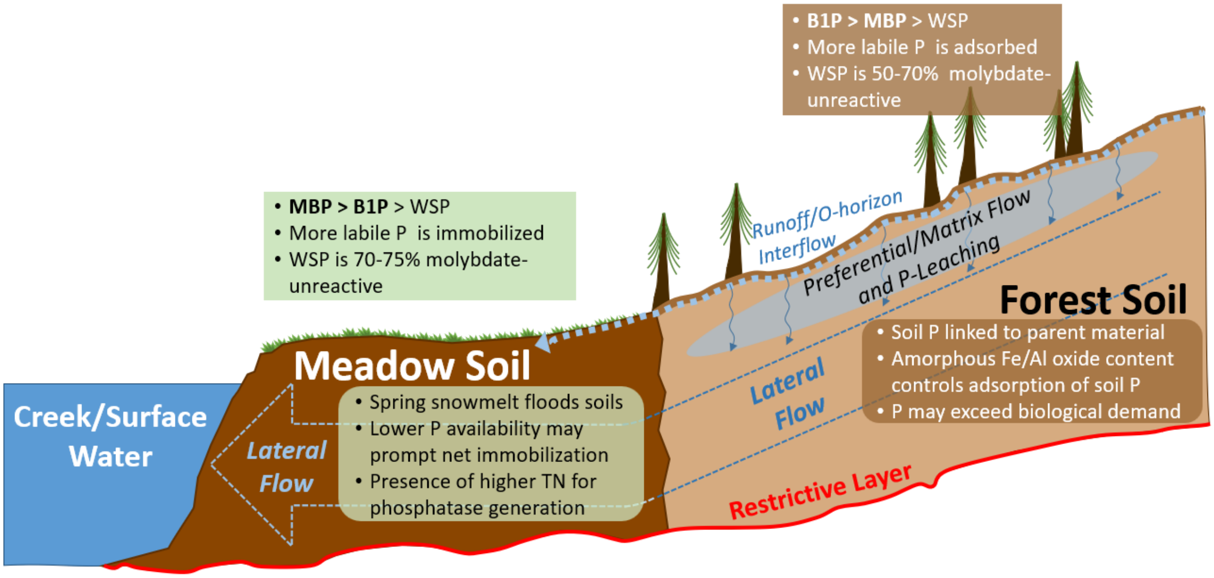


Figure 2.11: Partitioning of labile P forms in the landscape, and potential pathways of mobilized P transport. Most water-soluble P is in molybdate-unreactive forms, which may be partly associated with organic forms of P.

Brödlin et al. (2019) characterized soluble organic P in leachate as "a potential leak in the P cycle [contributing] to the progressive P depletion of soils." Because organic P is more mobile than inorganic orthophosphate in soil, and it accounted for the majority of WSP, it may constitute the fraction of TP in soils that is most vulnerable to transport, such as lateral flow during spring snowmelt that saturates the soil. Therefore, un-mineralized soil organic P, possibly derived from the P-rich forest O horizons, may be prone to transport to the floodplain meadows. A potentially beneficial focus of future research may examine the P-31 NMR speciation of forest O horizons and compare composition of P forms to that in both forest and meadow soils. Also, expanding P-31 NMR analysis to additional sites may allow for further investigation into relationships between WSP_{MU} and various organic P species across both watersheds, which may offer further understanding towards P loading to the basin's surface waters.

2.6 CONCLUSION

Results from this study provide insights into processes regulating the loss of labile soil P to surface and groundwater. When moving from forest to meadow, P storage shifts from sorption on minerals to immobilization in microbial biomass. Under certain conditions, these labile P sinks are each vulnerable to losses by unique processes in the environment. B1P from forested hillslopes may be gradually depleted as losses from the soil solution are continuously replenished, while MBP in meadows may be lost to subsurface lateral flow draining to surface waters upon seasonal drying and rewetting patterns that promote mineralization (Turner & Haygarth, 2001).

Chapter 3: Environmental Factors Influencing the Fractionation of Soil Phosphorus in Forests of the Lake Tahoe Basin, California and Nevada

3.1 Overview

Phosphorus (P) loss from the leaching of temperate forest soils is an important source of nutrient loading to surface waters, especially during periods of high-flow hydrologic events (Bol et al., 2016). Effective strategies for reducing P loading to surface waters involves the identification of transport paths as well as source areas in the landscape to support targeted management activities (Easton et al., 2007; Reid et al., 2018). Johnson et al. (2010; 2011) reported the presence of hotspots in Lake Tahoe Basin soils where nutrients, including labile phosphorus, are accumulating at higher concentrations than surrounding soils. These hotspots may be occurring from infiltration and sorption of P leached from organic horizons into the soil. This establishes a need for identifying site characteristics associated with soils in the landscape having higher than expected concentrations of labile P.

To ascertain the environmental factors and soil physicochemical properties related to the distribution of labile and other P forms in the Basin, literature explaining these relationships was reviewed. Important environmental factors include climate, geology and parent material, topography, and soil type. Climate has been observed to impart a significant influence on the extent of soil weathering which relates to properties that regulate the quantity and forms P retention in soil (Feng et al., 2016; Zhang et al., 2005). Additionally, labile soil P stocks can be sensitive to contrasting seasonal wet and dry periods (Campo et al., 1998; DeLonge et al., 2013), which is reflective of the xeric soil moisture regime of the Tahoe Basin (Figure A.1). Phosphorus composition of rocks and soil parent material mineralogy is another important factor controlling P input into watersheds, particularly by the erosion and release of nutrient-laden sediments (Ankers et al., 2003; Dillon & Kirchner, 1975). Dillon and Kirchner (1975) reported differences in P composition between igneous rocks of various origin, with higher levels of P associated with volcanic rock compared to plutonic rock. Jurassic through Cretaceous plutonic granitic rocks as well as volcanic andesite and basalt, formed during Oligocene through Pleistocene periods, are the most common rock types in the Basin (US Army Corps of Engineers, 2003). Consequently, the two main soil types are low nutrient granitic soils (the dominant type covering most of the basin) and volcanic soils with higher nutrient concentrations located in the northwestern region of the basin (Roberts & Reuter, 2010). Site characteristics of slope and topographic wetness index (TWI) were suggested by Moore et al. (1993) to offer insight into finer-scale hillslope processes controlling landscape distribution of labile soil P. Day et al. (1987) explained the influence of topography on landscape distribution of soil P forms, observing lower landscape positions with more intensely weathered soil particles were associated with higher levels of both total and labile soil P. Roberts et al. (1985) found increasing organic P and labile P, and decreasing inorganic P from upper to lower slope positions. In Chapter 2, we also observed a similar trend with a higher organic fraction of soil P in floodplain meadows compared to forested hillslopes. Soil physicochemical properties such as texture, mineralogy, and organic matter content influence P sorption capacity, which controls the potential for P losses from the profile (Daly et al., 2001; Djodjic et al., 2004; van Es et al., 2004). Djodjic et al. (2004) reported higher rates of P leaching as a result of bypass flow in a soil that had not been fertilized in forty years compared to soils that received higher P applications, but also had high sorption capacity.

Important physicochemical soil properties influencing P dynamics were presented by Tiessen et al. (1984), and include pH, total organic carbon (TOC), total nitrogen (TN), texture, and extractable iron (Fe) and aluminum (Al). Retention of P varies widely depending on the range of soil pH (Barrow, 1984; Devau et al., 2009). In acidic conditions, adsorption of soil P tends to occur with iron and aluminum oxides, while at intermediate pH levels, deprotonation of clay phyllosilicates allows for replacement by divalent and trivalent cations that can adsorb P (Devau et al., 2009). Agbenin (2003) also demonstrated the importance of extractable Fe and Al oxide content as a primary factor in controlling the sorption capacity and availability of soil P. Soil organic C and P mineralization have been shown to share a positive relationship because of observed increases in phosphatase activity with increases in organic carbon (Satti et al., 2007). However, Feng et al. (2016) studied soils along a climosequence, and observed decreases in soil P relative to TOC and TN as aridity decreased and weathering intensity increased. This group attributed this change to a shift from abiotic to biotic control on P cycling at sites with increasing wetness.

Soil environmental properties are commonly tested for their separate or combined abilities to predict status and potential mobility of P in soils across landscapes. Regression analysis by Roger et al. (2014) found that different terrain attributes including elevation, slope, and TWI were not strong predictors when used for several different extractable pools of soil P across a total of 245 sites including cropland, grassland, and mountain pasture. At 504 sampling sites in 16 different agricultural watersheds in Manitoba, Wilson et al. (2016) found TWI to be an important predictor of labile soil P, with higher concentrations in low-lying positions. Achat et al. (2016) compared the performance of various physicochemical soil properties to explain P availability, and found that iron and aluminum oxides as well as organic carbon content of soils were important predictors for inorganic P dynamics. The ability of different site characteristics and soil properties to influence the landscape distribution of soil P forms, or to predict potential source areas of labile soil P is highly variable and dependent on local environmental conditions. The objective of this project was i) to investigate the relative influence of various environmental factors and soil properties on the distribution of several soil P forms in forested watersheds of the Lake Tahoe Basin in California and Nevada, ii) evaluate the efficacy of these variables for prediction of labile P, and iii) examine the strength of relationships between these variables with soil depth.

3.2 METHODS

3.2.1 SAMPLING AND SPATIAL DATA

Soil samples from 53 sites during summer 2016 and another 38 sites the following summer 2017 (for a total of 91 sites) were collected from around the Lake Tahoe Basin forests of California and Nevada, using a composite sampling technique. Composite 0 – 5 cm soil samples were collected randomly within an approximate 8-meter radius around a central point. At a subset of 25 sites from 2016, samples were also collected at depth from a single, central point within the composite sampling radius in three layers (0 – 5 cm, 5 – 20 cm, and 20 – 40 cm). All surface composite and depth samples were air-dried, sieved ($<$ 2.00-mm), and stored in re-sealable plastic bags. At each of the original 53 locations from 2016, forest organic horizons were also described and sampled from a uniform area of 1 square foot. A measurement of litter depth and mass was recorded, and litter samples were subsequently dried and stored in re-sealable plastic bags.

Figure 3.1 depicts the spatial distribution of sampling points around the Basin and the heterogeneity of geologic and climatic environmental conditions represented with sampling. Total annual precipitation ranges from 615 – 1577 mm across all sampling sites in the Basin. Sampling distribution adequately captures the precipitation imbalance between the western and eastern parts of the basin. We divided our sampling into “wet” (above 1000 mm annual precipitation) and “dry” (below 1000 mm) treatments, which effectively represents the rain shadow phenomenon in the Basin. Sampling also resulted in fairly even distribution across north-facing ($n = 33$) and south-facing ($n = 32$) topographic aspect. North-facing hillslopes were designated as those having an orientation above 292.5 and below 67.5 degrees, and south-facing were those between 112.5 and 247.5 degrees. Aspect and other terrain attributes including elevation, slope, and topographic wetness index were derived from a 30-meter DEM (U.S. Geological Survey, 1996).

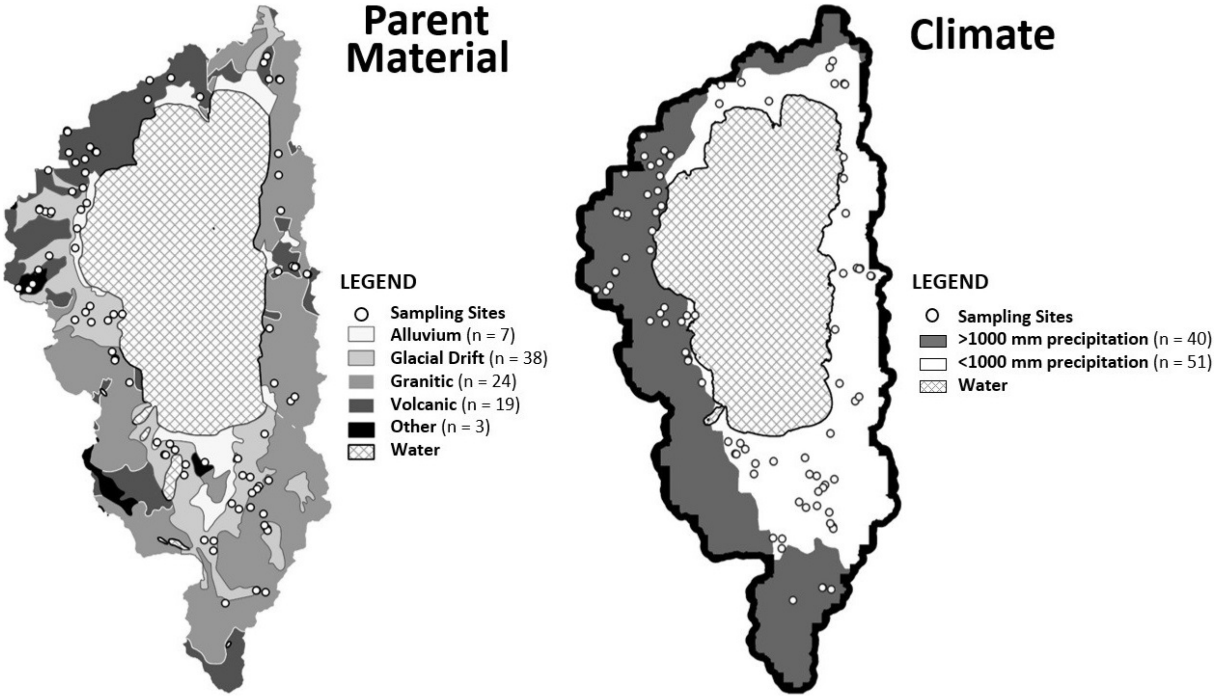


Figure 3.1: Spatial distribution of sampling sites among parent material types and wet/dry zones (above/below 1000 mm precipitation).

3.2.2 LABORATORY ANALYSES

Soil phosphorus analysis for total P (TP) and Mehlich-3 P (M3P) in all surface composite and profile depth samples was provided by A & L Great Lakes Laboratory (Fort Wayne, Indiana, United States).

All samples were measured for pH, percent sand, total organic carbon, total nitrogen, and oxalate-extractable iron, aluminum, and phosphorus. The pH of soils was measured with the 1:1 method using previously dried soils in ultrapure deionized water. Percent sand by mass was approximated by sieving the soils to measure particle size fraction greater than 63-µm. An elemental CNS analyzer was used to measure total organic carbon (TOC) and total nitrogen (TN). Soils were extracted for poorly crystalline iron and aluminum oxides in a 1:50 solid-solution ratio of 0.2 M ammonium oxalate solution in darkness, shaken for 4 hours, allowed to settle overnight, centrifuged at 1,500 g for 30 minutes, and filtered through 0.22-µm diameter PES membrane filter (USDA, 2014). The oxalate extracts were analyzed on an ICP-AES for iron, aluminum, silicon, and phosphorus.

Organic (O) horizon samples were ground and passed through a 2-mm sieve, extracted for water-soluble P in a 1:50 solid-solution ratio in 18 megaohm deionized water, shaken for 1 hour, centrifuged 4,000 g for 15 minutes, and filtered through 0.45-µm m diameter PES membrane filter. An aliquot of filtered extract for each litter sample was subsampled and analyzed for molybdate-reactive WSP on a spectrophotometer (Murphy & Riley, 1962), and total O-horizon WSP was analyzed on an ICP-AES.

3.2.3 Statistical Analyses

Box-whisker plots are used to present the spread of our results, by separating data into quartiles. The top and bottom whiskers represent the upper 25% and lower 25% of observations, respectively, the box confines the middle 50% or interquartile range (IQR) of observations, the middle line is the median, and the “X” is the average. Observations are designated as outliers if they fall beyond 1.5 times IQR above the upper interquartile limit, or beyond 1.5 times IQR below the lower interquartile limit.

Significance in response to the different treatments was tested by fitting the data to analysis of variance (ANOVA) linear models using the R ‘stats’ package in R (R Core Team, 2018). Tukey’s honest significance difference (HSD) test was used for assessing statistical differences ($p < 0.05$) between treatment means. Pearson’s correlation coefficients were used to evaluate the strength of relationships between extractable pools of soil P and all other continuous variables, with the ‘Hmisc’ package version 4.2-0 in R (Harrell, 2019) at three significance levels ($p < 0.05$; < 0.01; < 0.001).

Forward stepwise multiple regression analysis was done using ‘leaps’ package version 3.0 in R (Lumley, 2017) to select models using environmental variables and soil physicochemical properties as predictors of M3P. This package conducts a parsimonious model selection process to identify regression equations yielding the lowest Akaike information criterion (AIC) and highest adjusted-R^2 parameters using the least number of independent terms for prediction of a dependent variable. Stepwise addition of variables stops when successive variables no longer increase significance. Selected predictors were then fitted to an OLS linear model using the 'stats' package in R (R Core Team, 2018). The variance inflation factor (VIF) from the 'car' package in R (Fox & Weisberg, 2019) was used to assess the extent of multicollinearity among independent variables.

3.3 Results and Discussion

3.3.1 Organic Horizons and Surface Mineral Soils

Extractable pools of soil P, environmental variables, and soil physicochemical properties for all Basin sampling sites (n = 91) are described in Table 3.1. Soil TP in the basin ranged from 110 – 2,320 mg/kg, with several outliers reaching up to 4,520 mg/kg (Figure 3.2). Soil oxalate P ranged from 50.8 – 1,488 mg/kg, with outliers reaching up to 2,159 mg/kg. Mehlich-3 P in the Basin's soils ranged from 11 – 205 mg/kg. Average soil pH in the basin is 5.92, but range from an acidic pH of 4.00 to a more circumneutral pH of 7.72. Most soils in the basin (68 of 91 sites) contained 80 – 90% sand. Average TOC was 4.84% and average TN was 0.175%. Average oxalate Al was 0.561% (max of 2.56%) and average oxalate Fe was 0.475% (max of 1.80%). Average elevation of sampling sites was 2058 m, approximately 161 m above surface elevation of Lake Tahoe. Slope steepness of sampling sites ranged from approximately 1 – 26%. Topographic wetness index (TWI) ranged from 4 – 16. Total annual precipitation ranged from 616 mm to 1577 mm.

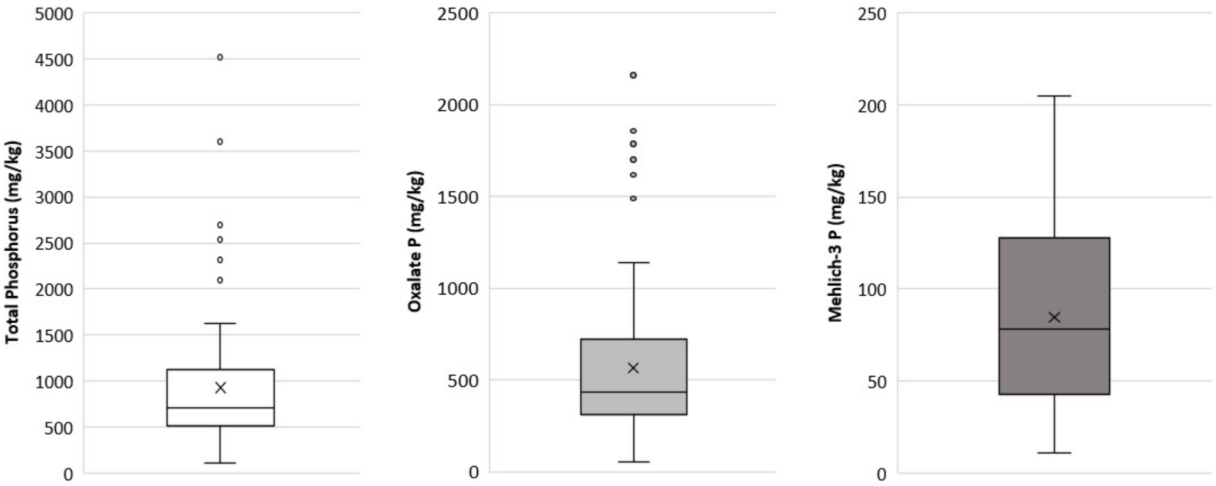


Figure 3.2: Box-whisker plots summarizing spread of observations for total, oxalate, and Mehlich-3 P at all sampling sites in the Basin (n = 91).

Table 3.1: Summary descriptive statistics (mean, median, min, max, std. deviation, std. error) for extractable soil phosphorus pools, environmental variables, and soil physicochemical properties across the Basin.

	Mean	**Median**	**Min**	**Max**	**Standard Deviation**	**Standard Error**
Extractable Soil Phosphorus						
Total P (mg/kg)	929	710	110	4520	718	75.3
Mehlich-3 P (mg/kg)	84.6	78	11	205	48.8	5.11
Mehlich-3/Total P (%)	13.7	11	1	62	11.5	1.21
Oxalate P (mg/kg)	566	430	50	2160	433	45.3
Environmental Variables						
Slope (%)	9.54	8.74	0.799	25.7	6.08	0.637
Elevation (m)	2058	2013	1900	2411	134	14
Precipitation (mm)	969	968	616	1577	215	22.6
Topographic Wetness Index	6.57	6	4	16	1.95	0.204
Soil Physicochemical Properties						
pH	5.92	5.9	4	7.72	0.841	0.0882
TN (%)	0.175	0.124	0	0.96	0.152	0.0159
TOC (%)	4.84	3.68	0.575	17.3	3.33	0.349
Sand (%)	84.5	85.9	63.7ˆ	95.3	6.41	0.672
Oxalate Al (%)	0.561	0.342	0.07	2.56	0.521	0.0547
Oxalate Fe	0.475	0.425	0.051	1.8	0.3	0.0315

Sample size, n = 91 for all variables

ˆnext lowest measurement for Sand was 72%; 68 of 91 samples falling between 80-90%

Table 3.2: Summary descriptive statistics (mean, median, min, max, std. deviation, std. error) for organic (O) horizon properties across the Basin, including depth, Mass, and water-soluble P (per unit litter mass and per unit forest land area).

Organic Horizon Property	Mean	Median	Min	Max	Standard Deviation	Standard Error
Depth (cm)	3.2	2.0	0.0	12	3.3	0.4
Mass (g)	438	385	0.25	1231	296	40.6
Molybdate-reactive WSP (mg/kg)	74.1	71.2	17.5	166	33.3	4.71
Total WSP (mg/kg)	119	115	33.5	229	45.7	6.46
Molybdate-reactive WSP (mg/m^2)	336	247	14.7	1781	317	44.9
Total WSP (mg/m^2)	533	437	28.2	2557	473	66.9

Sample size, n: 69 for depth, 53 for mass, 50 for all litter P measurements

Organic (O) horizon depth, mass, and water-soluble P (WSP) content is shared in Table 3.2, and spread of WSP data is represented in Figure 3.3. Average organic (O) horizon thickness was 3.2 cm, with a max of 12 cm (n = 69). Mass of litter from O-horizons ranged from 0.250 – 1,231 g, with an average of 438 g (n = 53). Total water-soluble P from the forest O-horizons ranged from 28.2 – 1,435 mg/m^2, with a few outliers reaching up to 2,557 mg/m^2 (n = 50). On average, molybdate-reactive forms accounted for 61% of the total water-soluble P from litter, but ranged from 52 – 70%. Our characterization of Tahoe forest O-horizons shows the potential for readily soluble P forms to accumulate in these layers. This supports previous work in the Basin which demonstrated the capacity of these litter layers to leach P, resulting in enriched runoff loading directly to surface waters (Miller et al., 2005) or development of hotspots along preferential flowpaths in the mineral soil (Johnson et al., 2011).

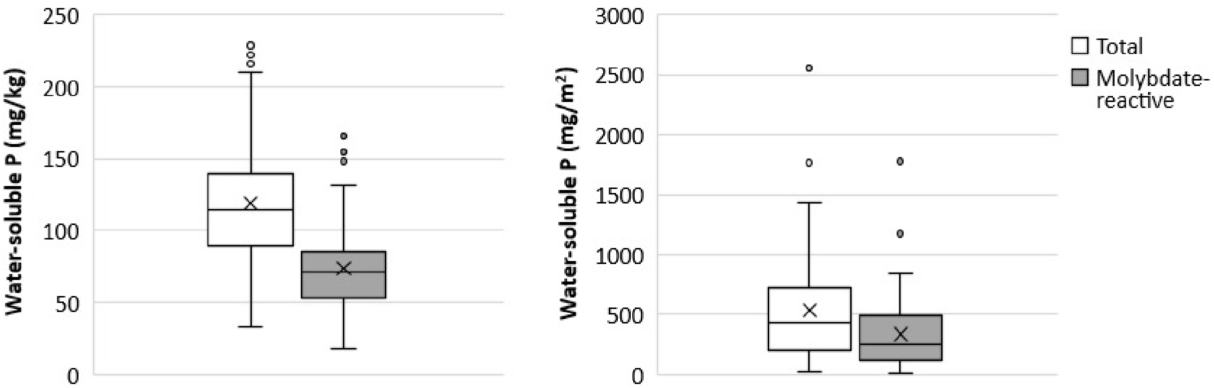


Figure 3.3: Box-whisker plots summarizing spread of observations (n = 50) for total and molybdate-reactive water-soluble P in O-horizons, expressed in per unit litter mass (left) and per unit forest land area (right).

Relationships between Soil Phosphorus and Discrete Variables

The results show two statistically different ($p < 0.05$) tiers of TP content in soils derived from the four parent material classes in the basin (Figure 3.4). Soils from glacial drift and granitic parent materials had significantly lower TP than those from volcanic and alluvial parent materials. The lower TP parent materials appear to have higher average M3P concentrations than the high TP parent materials, but these differences are not statistically significant on a soil mass basis. When normalized per unit TP, average percent M3P is higher ($p < 0.05$) in the low TP parent material soils than high TP parent materials (Figure 3.5). This difference is most significant between granitic soils and volcanic soils.

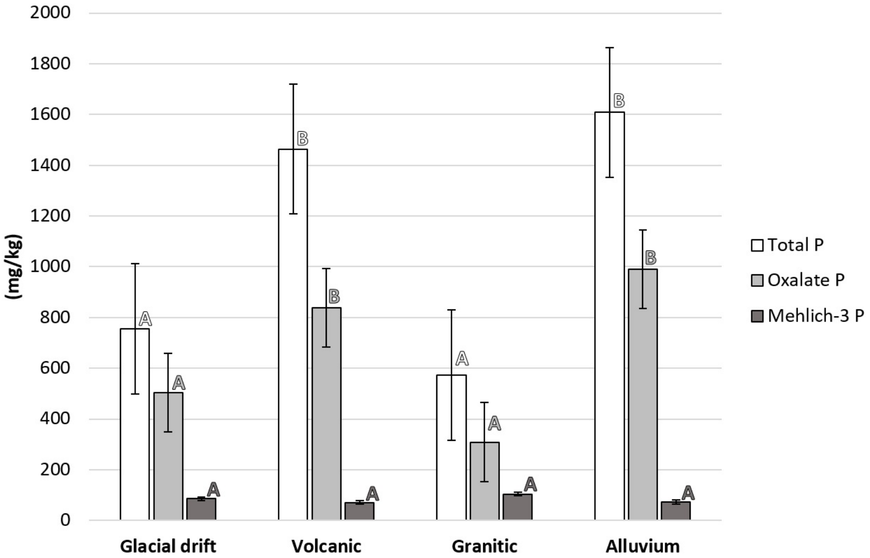


Figure 3.4: Average composition of extractable soil phosphorus (Total, Oxalate, and Mehlich-3) with standard error bars in each parent material. Comparisons of each soil pool across sites are made (letters with the same color can be compared). Different letters indicate statistically significant differences ($p < 0.05$).

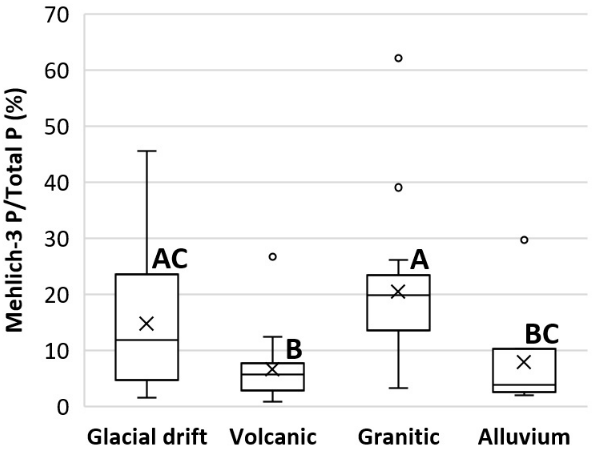


Figure 3.5: Box-whisker plots of Mehlich-3 P expressed per unit TP from each parent material. Letters indicate statistically significant differences ($p < 0.05$).

In a study by Brödlin et al. (2019), low TP soils also contained larger fractions as labile P than TP-rich soils, which they attributed to reduced iron and aluminum oxides as well as higher sand content in the low TP soils that suggests a reduced sorption capacity. Associated physicochemical properties of our study soils agree with the Brödlin et al. (2019) explanation (Figure 3.6); the granitic and glacial drift soils have a lower capacity to sorb P than volcanic and alluvial soils. Additionally, parent materials in the basin with higher permeability including granitic glacial till, outwash deposits, and granitic rocks of the Carson Range (Tumbusch et al., 2007), may be more prone to weathering and depletion of P content prior to pedogenesis (Porder & Hilley, 2011), explaining lower TP status in soils from those parent materials.

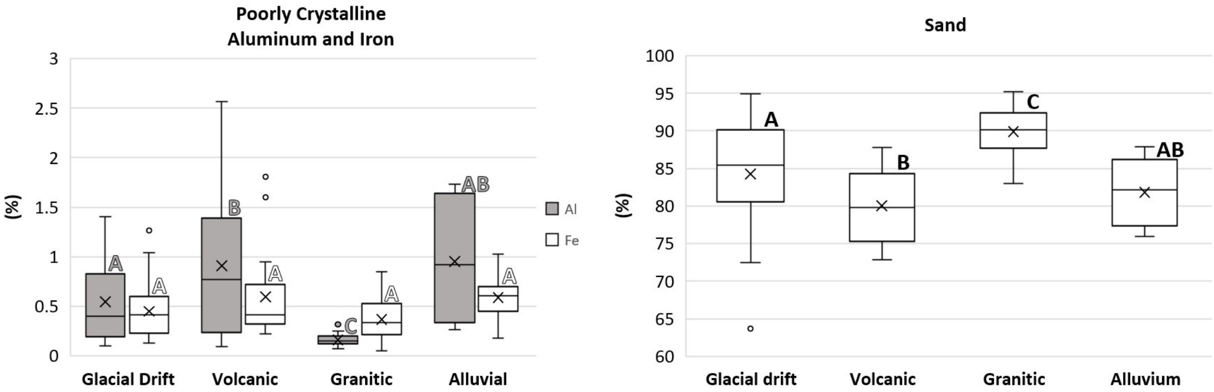


Figure 3.6: Oxalate Al and Fe (left), and sand content (right) in response to parent material treatment. Letters indicate statistically significant differences ($p < 0.05$).

Precipitation classes (above/below 1000 mm precipitation) demonstrated significant differences in both TP and M3P (Figure 3.7). Wet sites had higher average TP than dry sites. A similar pattern between TP and M3P in geology also occurred with precipitation classes, where low TP dry sites had higher average M3P and high TP wet sites had low M3P. These results are contrary to several studies of labile P along climosequences. According to Brye et al. (2004), Arkansas grassland soils receiving 158 mm of additional precipitation during months with low potential evapotranspiration had no significant difference in M3P compared to the soils that received less precipitation. A greater disparity in average precipitation between wet and dry sites in our study (345 mm) may justify the significance in our results. However, multiple studies have found increasing contributions to soil labile P from the orthophosphate diester fraction of organic P in soils receiving higher rates of annual precipitation (e.g., undisturbed New Zealand tussock grasslands (Tate & Newman, 1982); North American Great Plains (Sumann et al., 1998).

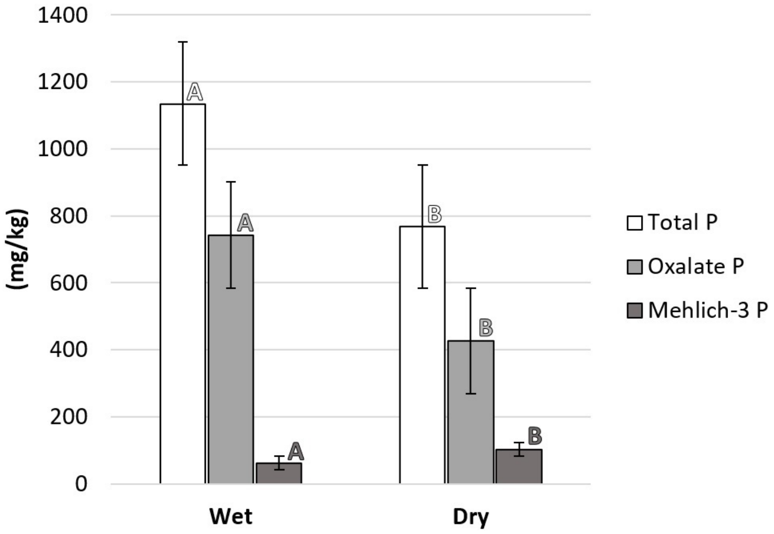


Figure 3.7: Average composition of extractable soil phosphorus (Total, Oxalate, and Mehlich-3) from each climate. Letters indicate statistically significant differences ($p < 0.05$).

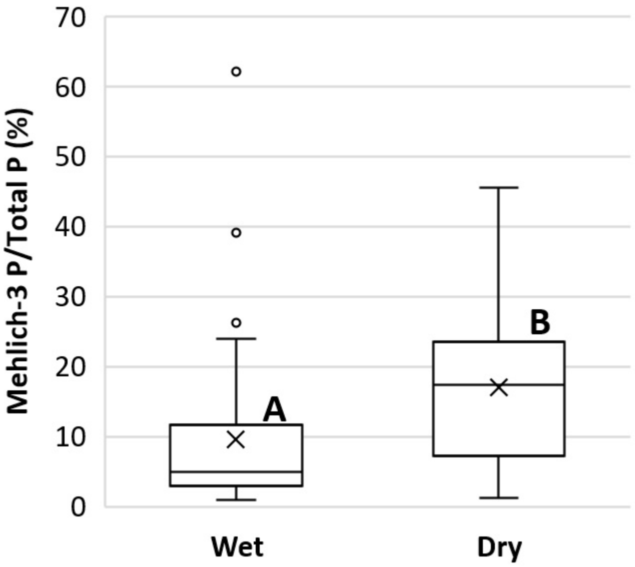


Figure 3.8: Boxplots of Mehlich-3 P expressed per unit TP from each climate. Letters indicate statistically significant differences ($p < 0.05$).

A significant response to topographic aspect (north versus south) did not occur in any pool of extractable soil P (Figure 3.9). Despite differences in rates of evapotranspiration and leaching processes, Måren et al. (2015) also observed no significant differences in P with respect to slope aspect in high-altitude, trans-Himalayan semiarid valley soils. Other groups observed higher soil labile P on south-facing hillslopes (Sidari et al., 2008; Yimer et al., 2006). Sidari et al. (2008) attributed the presence of greater plant available nutrients including P on south-facing slopes to simultaneously higher levels of hydrolyzing enzymes. Although not statistically significant, average TP and oxalate P in the Tahoe Basin soils were slightly higher on south-facing slopes, but M3P was not notably different. Because only oxalate P and TP (more recalcitrant P) expressed a response to aspect, it is more likely that contrasting rates of parent material P depletion by weathering (Klemmedson & Wienhold, 1992) rather than enzymatic activity contributed to differences in soil P with regard to topographic aspect throughout the basin.

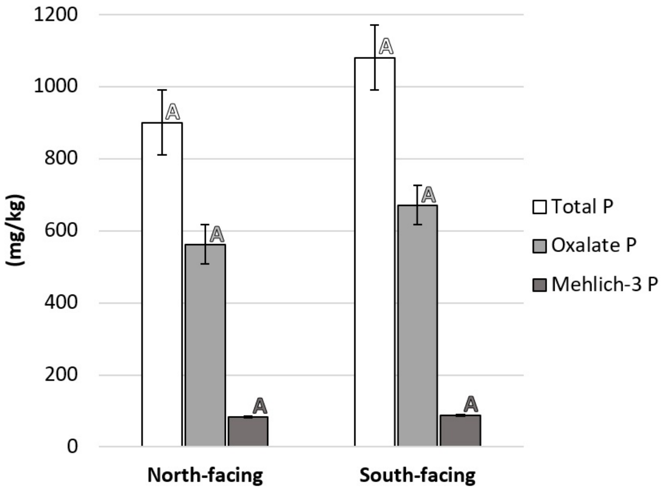


Figure 3.9: Average composition of extractable soil P (Total, Oxalate, and Mehlich-3) from north-facing (n = 33) and south-facing (n = 32) hillslope aspect. Letters indicate statistically significant differences ($p < 0.05$).

Relationships between Soil Phosphorus and other Continuous Variables

Pearson's correlations and p-values among the extractable soil P pools and with environmental variables, soil physicochemical properties, and O-horizon properties are summarized in Table 3.3. Oxalate P shares a strong positive correlation with TP. Similarly, oxalate Al and Fe share significant positive relationships with both oxalate P and TP. Alternatively, oxalate P, Al, and Fe were all significantly and negatively correlated with M3P/TP. Therefore, as poorly crystalline Al and Fe oxides increase, oxalate and total P both increase while the fraction of M3P decreases. Because oxalate P is representative of

P tightly held by poorly crystalline Al and Fe oxides in soil (Wolf & Baker, 1990), this fraction of mineralogy is an important factor restricting the mobility of P in basin soils.

Both TOC and TN had significant positive correlations with TP and oxalate P, and negative correlations with M3P and M3P/TP. The strong positive correlation of TN with TOC suggests that most soil N is likely associated with soil organic matter (Figure 3.10). The positive correlation of TOC with oxalate P and oxalate Al suggests the possibility of P retention by some Al-humus complexes in these soils (Kang et al., 2009; Ohno et al., 2007).

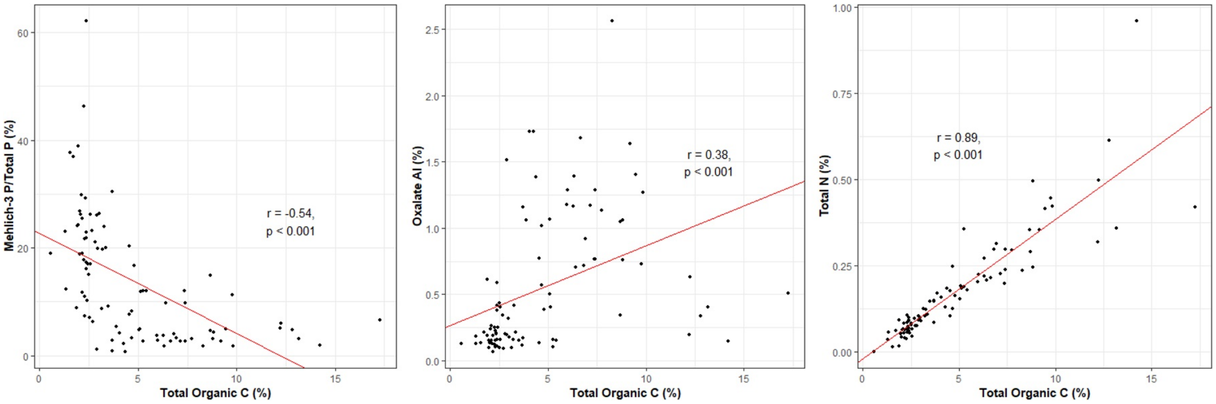


Figure 3.10: Scatterplots depicting significant ($p < 0.001$) correlations of TOC with M3P/TP, oxalate Al, and total N.

Attributes related to terrain (slope, elevation, TWI) shared little to no relationship with pools of extractable soil P in the surface composite soils. Percent slope shared a weak, but significant positive relationship with TP and oxalate P. Litter depth, mass, and WSP exhibited weak, but significant positive relationships with TP and negative relationships with M3P/TP. Litter WSP correlation with TP and M3P/TP was only significant when litter WSP was expressed per unit forest land area, and not per unit litter mass. This implies a tendency for thicker and heavier forest floor litter layers to accumulate in the basin above soils having larger TP reserves but smaller fractions as M3P. This may be due to immobilization and "intrasystem" cycling of P by microbes within the O-horizons (Gosz et al., 1976; Yanai, 1992). Johnson et al. (2011) also found a weak, but significant correlation between Lake Tahoe Basin forest O-horizons and labile P in the underlying mineral soil. The group was not able to make any conclusive determinations of soil P hotspots in the Basin based on this relationship.

A stepwise multiple regression analysis selected linear models with the lowest AIC and highest adjusted-R^2 for the prediction of M3P in surface composite soil samples (Table 3.4). Conducting the model selection process on the entire surface composite dataset yielded an adjusted-R^2 of 0.67. Using subsets of surface composite soils from low TP soils (formed on granitic and glacial drift) and high TP soils (formed on volcanic and alluvium) improved the adjusted-R^2 for each selected linear model to 0.79 and 0.84, respectively, for each TP grouping. VIF is only high (> 10) in some interaction terms that share variables with other terms in the equation, suggesting that multicollinearity is not an issue in the selected models (Allison, 2012). A distribution of observed M3P versus residuals using the models specific to low and high TP parent materials shows that the model's performance is best at intermediate concentrations of M3P (approximately 75 – 150 mg/kg; Figure 3.11). The absolute value of percent residuals was approximately 26% on average, and outlier residuals only occurred in the low TP parent material model (Figure 3.12). The highest rate of prediction error occurred mostly with observations below 50 mg/kg of M3P.

The model for all soils (low and high TP) relies primarily on soil physicochemical properties as input parameters to predict M3P (especially sand content and oxalate Fe, Al, and P). The models using subsets of the data continued to primarily select terms that included oxalate extractable soil elements, but also showed increased dependency on environmental factors. The model for high TP soils includes four predictors that incorporate the use of TWI individually, squared, and interacting with oxalate P and Al. This implies that TWI alone may not exhibit a significant relationship with M3P (see correlation of these two variables in Table 3.3), but that the relationship between TWI and M3P changes at varying concentrations of oxalate Al and P. In low TP soils, precipitation is an important environmental variable, along with oxalate P, for predicting M3P. Regression analysis demonstrates the ability to effectively predict labile P in the soil using the set of environmental variables and soil properties included in this study.

Table 3.3: Pearson's r correlations of extractable soil phosphorus pools with environmental variables, soil physicochemical properties, and O-horizon properties.

	Total P	**Mehlich-3 P**	**Mehlich-3 P/Total P**	**Oxalate P**
Extractable Soil Phosphorus				
Mehlich-3 P	-0.09	-	-	-
Mehlich-3 P/Total P	-0.52***	0.64***	-	-
Oxalate P	0.88***	-0.05	-0.49***	-
Environmental Variables				
Slope	0.28**	-0.02	-0.18	0.25*
Elevation	0.10	-0.11	-0.20	0.00
Precipitation	0.26*	-0.41***	-0.39***	0.39***
Topographic Wetness Index	-0.03	-0.14	-0.13	-0.05
Soil Physicochemical Properties				
pH	0.23*	-0.09	-0.06	0.23*
TN	0.28**	-0.45***	-0.55***	0.28**
TOC	0.31**	-0.41***	-0.54***	0.33**
Sand Content	-0.44***	0.50***	0.64***	-0.53***
Oxalate Al	0.76***	-0.35***	-0.59***	0.85***
Oxalate Fe	0.37***	-0.26*	-0.42***	0.42***
Oxalate Si	0.67***	-0.42***	-0.56***	0.74***
O-horizon Properties				
Litter Depth	0.33**	-0.09	-0.24*	0.22
Litter Weight	0.27	-0.26	-0.39**	0.23
Litter Total WSP (mg/kg)	0.19	-0.07	0.06	0.17
Litter Molybdate-Reactive WSP (mg/kg)	0.16	-0.07	0.11	0.14
Litter Molybdate-Unreactive WSP (mg/kg)	0.23	-0.07	-0.06	0.20
Litter Total WSP (mg/m^2)	0.32*	-0.15	-0.29*	0.26
Litter Molybdate-Reactive WSP (mg/m^2)	0.31*	-0.14	-0.27	0.25
Litter Molybdate-Unreactive WSP (mg/m^2)	0.33*	-0.16	-0.33*	0.27

*significant at $p < 0.05$; **significant at $p < 0.01$; ***significant at $p < 0.001$

Table 3.4: Results of stepwise multiple regression analysis to select linear models for the prediction of Mehlich-3 P at all sites, for just high TP sites (those formed on volcanic or alluvial geology), and low TP soils (those formed on granitic or glacial till geology).

Selected Predictors	Regression Coefficients	p-value	VIFˆ	adjusted-R^2
All Sites				
(Intercept)	50.7	0.232	-	0.67
Precipitation*pH	-7.19E-03	3.27E-03**	1.48	
TP*TN	-0.176	5.43E-07***	6.36	
Oxalate P*Sand	46.3	4.55E-03**	312	
Oxalate P*Oxalate Fe	-423	0.0112*	4.62	
TN*Oxalate Al	141	0.0342*	11.1	
Sand*Oxalate Al	-1.23	5.31E-06***	11.6	
Oxalate P	-1.73E+03	0.177	344	
Sand^2	6.22E-03	0.259	3.83	
High TP Soils formed over Volcanic and Alluvial Geology				
(Intercept)	382	1.04E-05***	-	0.84
Precipitation*Sand	1.34E-03	3.81E-04***	1.62	
Precipitation*Oxalate Al	-0.104	1.12E-03**	33.44	
TWI*Oxalate P	133	3.32E-04***	10.1	
TWI*Oxalate Al	7.66	0.0403*	24.8	
TN*Oxalate Al	-80.7	0.0659	4.28	
TWI	-108	8.02E-05***	84.6	
TWI^2	6.44	1.03E-03**	82.4	
Oxalate Fe^2	-13.1	0.0237*	1.51	
Low TP Soils formed over Granitic and Glacial Drift Geology				
(Intercept)	97.1	3.44E-05***	-	0.79
Oxalate P	-4.24E+03	1.07E-03**	156	
Elevation*TP	-2.80E-05	2.54E-02*	11.4	
Precipitation*Oxalate P	-0.86	1.40E-01	57.8	
Precipitation*pH	-7.83E-03	2.60E-02*	2.92	
Precipitation*TN	-9.30E-02	5.99E-04***	2.08	
Oxalate P*Sand	93.7	1.87E-10***	82.2	
pH*Oxalate Fe	-5.58	3.48E-02*	1.75	
Sand*Oxalate Al	-0.376	8.39E-02	4.07	

*significant at $p < 0.05$; **significant at $p < 0.01$; ***significant at $p < 0.001$

ˆVariance Inflation Factor (VIF)

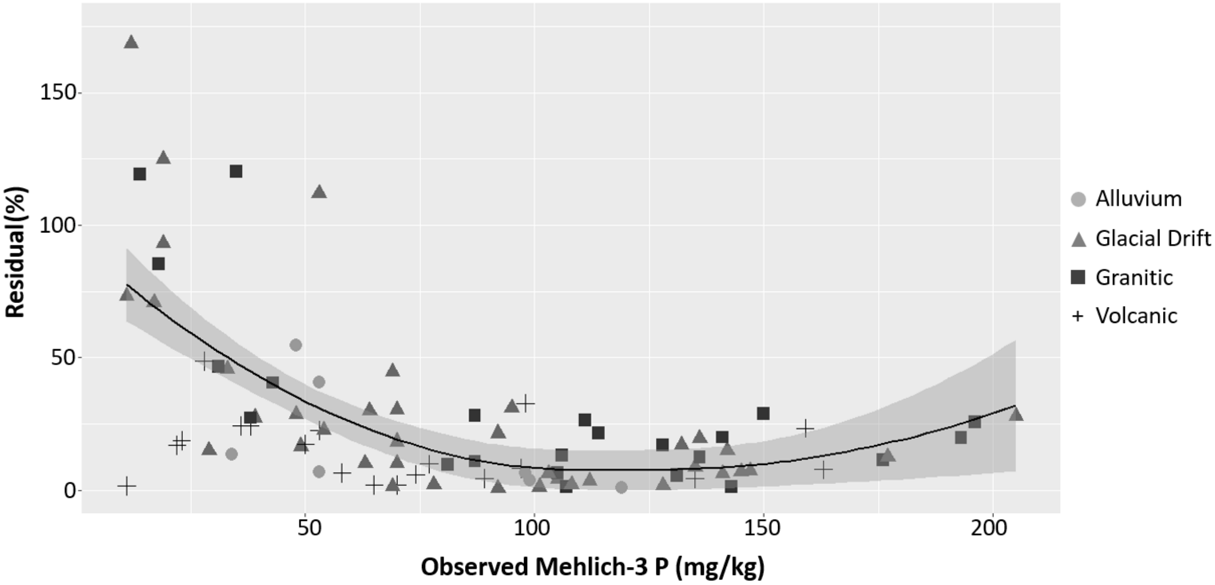


Figure 3.11: Distribution of observed Mehlich-3 phosphorus versus percent residuals of predicted Mehlich-3 phosphorus for each site, using parent material specific models. The LOESS fit shows the trend in model performance over a range of true concentrations, with a 95% confidence interval (generated with 'ggplot2' in R; Wickham, 2016).

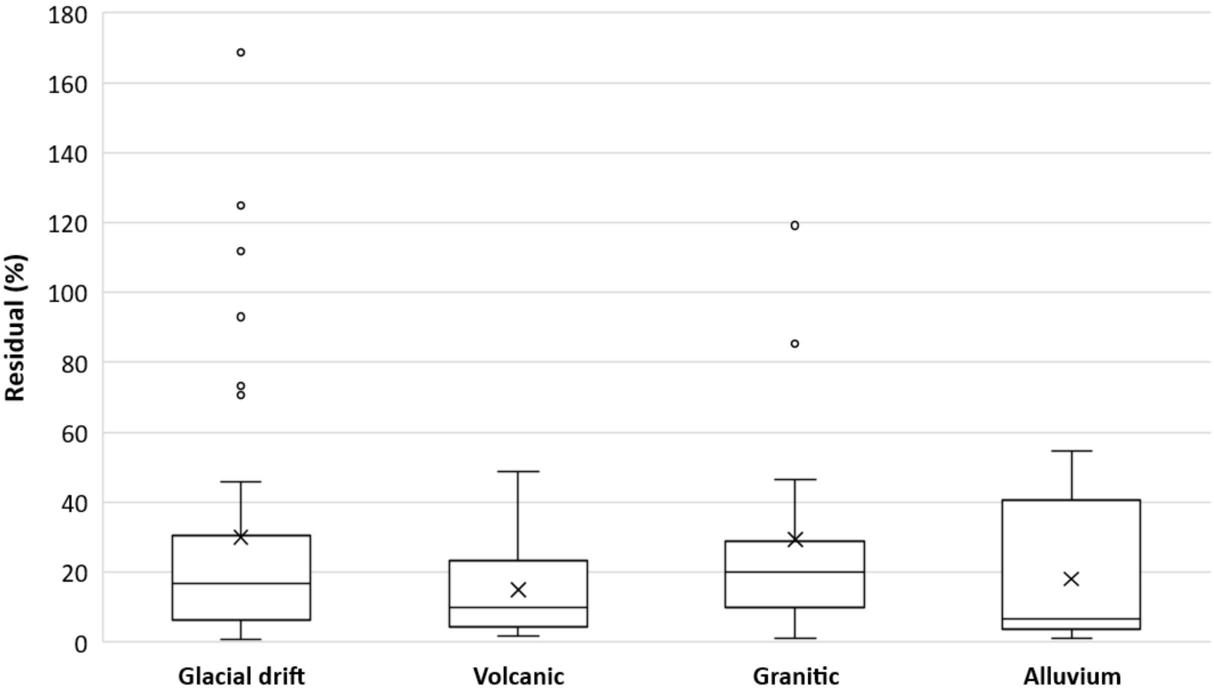


Figure 3.12: Box-whisker distribution of residuals (by parent material category) from low/high TP parent material models.

3.3.2 Soil Profile Phosphorus

Summary statistics for profile soil properties and P content (0 – 5, 5 – 20, and 20 – 40 cm) are presented in Table 3.5. To gain insight into potential environmental factors influencing the distribution of phosphorus in the soil profile, differences across three soil depths at 0 – 5 cm, 5 – 20 cm, and 20 – 40 cm were assessed. The only categorical sampling treatment to yield a significant difference in the distribution of soil profile P was climate (Figure 3.13). Climate influenced only the M3P extractable pool. At wet sampling sites, average M3P decreased steadily with depth, but the difference was only significant between the 0 – 5 cm and 20 – 40 cm layers. Webb et al. (1986) observed that higher retention of soil P with increasing depth along a climosequence occurred in because of increased amorphous Al and Fe caused by higher weathering rates in wetter soils. Similarly, we observed higher overall oxalate Al as well as increasing oxalate Fe with depth at wetter sites (Figure 3.14).

Oxalate extractable Al was the most important property for explaining the distribution of subsurface forms including TP, M3P/TP, and Oxalate P (but not M3P per unit soil mass) at both 5 – 20 cm and 20 – 40 cm (Table 3.6). TOC and TN were only correlated significantly to P at 5 – 20 cm soil depth. TOC only shared a weak significant negative correlation with M3P/TP, while TN was significantly positively correlated with TP and negatively correlated with M3P/TP. Despite the presence of some correlation between O-horizon properties with the surface composite soil P data, no correlations existed with soil P data at depth. Strong significant negative correlations existed between precipitation and both M3P and M3P/TP at the 20 – 40 cm depth only.

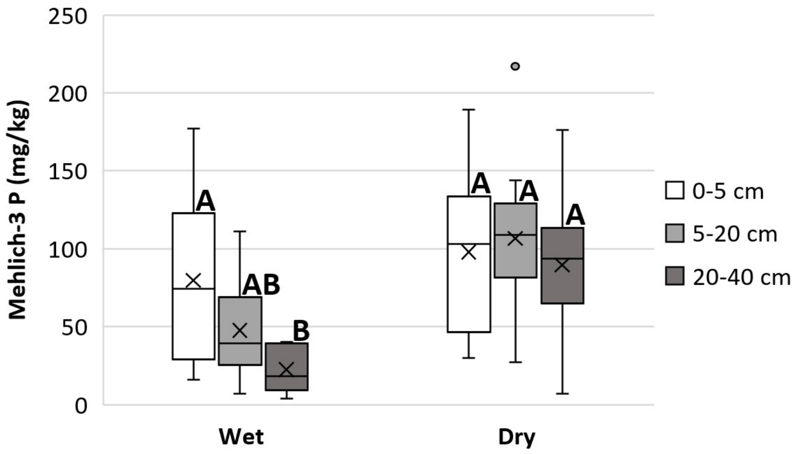


Figure 3.13: Distribution of Mehlich-3 phosphorus in soil profiles at wet (n = 12) and dry (n = 13) sampling sites.

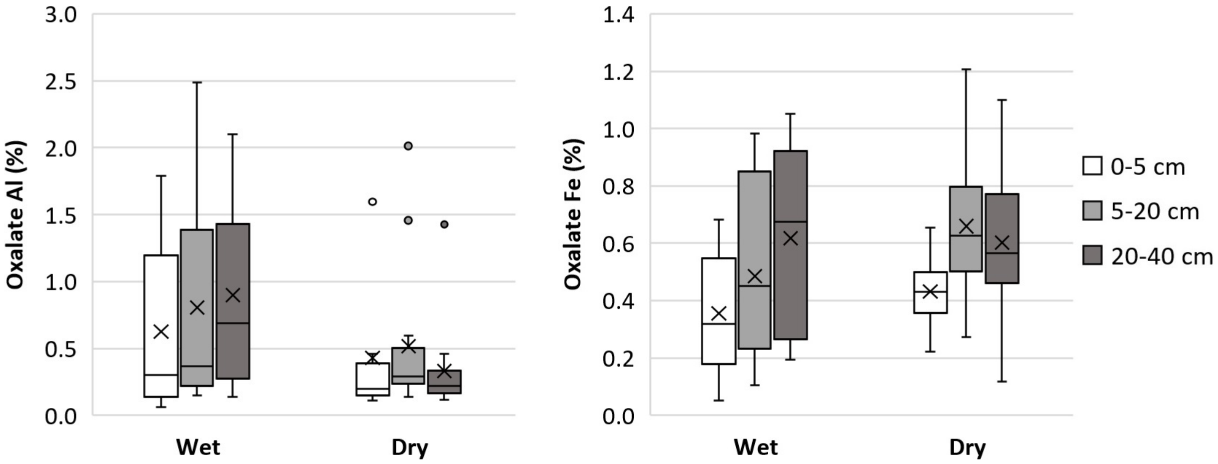


Figure 3.14: Distribution of oxalate Al and Fe in soil profiles at wet and dry sampling sites.

Table 3.5: Summary statistics for profile P and various soil properties at 0 – 5, 5 – 20, and 20 – 40 cm.

Soil Sample Depth: 0 - 5 cm

	n	Mean	Median	Min	Max	Standard Deviation	Standard Error
Extractable Soil Phosphorus							
Total P (mg/kg)	25	950	540	120	3930	987	197
Mehlich-3 P (mg/kg)	25	89.1	89	16	189	50.7	10.1
Mehlich-3/Total P (%)	25	14.8	12	3	39	10.1	2.02
Oxalate P (mg/kg)	24	634	370	60.0	3260	720	147
Soil Physicochemical Properties							
Bulk Density (g/cm^3)	50	1.35	1.37	0.504	2.2	0.355	0.0502
pH	24	5.83	5.71	4.88	7.28	0.644	0.132
TN (%)	25	0.195	0.15	0.0139	0.651	0.179	0.0359
TOC (%)	25	8.07	5.21	1.4	30.9	7.17	1.433
Sand (%)	24	91	92	70	99	6.4	1.3
Oxalate Al (%)	24	0.52	0.234	0.06	1.79	0.557	0.114
Oxalate Fe (%)	24	0.397	0.407	0.053	0.682	0.17	0.0347

Soil Sample Depth: 5 - 20 cm

	n	Mean	Median	Min	Max	Standard Deviation	Standard Error
Extractable Soil Phosphorus							
Total P (mg/kg)	25	840	580	100	3390	769	154
Mehlich-3 P (mg/kg)	25	78.2	75	7.00	217	49.6	9.92
Mehlich-3/Total P (%)	25	14.1	12.0	2.00	32.0	9.78	1.96
Oxalate P (mg/kg)	24	580	415	40.0	2190	555	113
Soil Physicochemical Properties							
Bulk Density (g/cm^3)	46	1.41	1.39	0.855	2.45	0.309	0.0456
pH	23	5.66	5.51	4.92	7.05	0.556	0.116
TN (%)	24	0.0682	0.0320	0.00	0.300	0.0861	0.0176
TOC (%)	24	3.89	2.67	0.791	13.1	3.33	0.680
Sand (%)	24	89	91	75	95	5.4	1.1
Oxalate Al (%)	24	0.649	0.320	0.135	2.49	0.674	0.138
Oxalate Fe (%)	24	0.580	0.550	0.106	1.21	0.277	0.0566

Soil Sample Depth: 20 - 40 cm

	n	Mean	Median	Min	Max	Standard Deviation	Standard Error
Extractable Soil Phosphorus							
Total P (mg/kg)	21	611	490	70.0	1820	424	92.4
Mehlich-3 P (mg/kg)	21	60.8	40.0	4.00	176	51.2	11.2
Mehlich-3/Total P (%)	21	12.5	11.0	0.00	39.0	9.51	2.08
Oxalate P (mg/kg)	21	397	380	20.0	890	263	57.4
Soil Physicochemical Properties							
Bulk Density (g/cm^3)	36	1.51	1.53	0.548	2.45	0.409	0.0682
pH	21	5.36	5.18	4.68	6.37	0.500	0.109
TN (%)	21	0.0279	0.00500	0.00	0.325	0.0703	0.0153
TOC (%)	21	2.04	1.27	0.499	10.7	2.27	0.494
Sand (%)	21	91	93	71	97	7.0	1.5
Oxalate Al (%)	21	0.576	0.272	0.117	2.10	0.587	0.128
Oxalate Fe (%)	21	0.608	0.591	0.117	1.1	0.274	0.0597

Table 3.6: Pearson's r correlations of extractable soil phosphorus pools with environmental variables, soil physicochemical properties, and O-horizon properties for profile soils (5 – 20 and 20 – 40 cm).

	5 - 20 cm				**20 - 40 cm**			
	Total P	**Mehlich-3 P**	**Mehlich-3 P/ Total P**	**Oxalate P**	**Total P**	**Mehlich-3 P**	**Mehlich-3 P/ Total P**	**Oxalate p**
Environmental Variables								
Slope	0.18	-0.08	-0.41*	0.15	0.0	-0.16	-0.33	-0.07
Elevation	0.17	-0.27	-0.49*	0.05	0.25	-0.17	-0.40	0.25
Aspect	-0.12	0.07	-0.11	-0.13	-0.12	0.12	-0.02	-0.10
Precipitation	0.15	-0.62**	-0.42*	0.10	0.17	-0.74***	-0.67***	0.13
Topographic Wetness Index	0.18	-0.14	-0.22	0.16	0.30	-0.11	-0.24	0.39
Soil Physicochemical Properties								
Bulk Density	-0.13	0.22	0.22	-0.12	-0.50*	0.33	0.53*	-0.49*
pH	0.08	0.07	-0.08	0.22	-0.10	0.01	0.0	-0.09
Total N	0.45*	-0.21	-0.53**	0.36	-0.14	-0.3	-0.18	-0.18
Total Organic C	0.36	-0.21	-0.45*	0.30	-0.06	-0.36	-0.27	-0.11
Sand Content	-0.55**	0.04	0.42*	-0.63***	-0.06	0.09	0.16	-0.06
Oxalate Al	0.88***	-0.03	-0.63**	0.86***	0.78***	-0.39	-0.62**	0.71***
Oxalate Fe	0.50*	0.34	-0.27	0.53**	0.69***	0.07	-0.19	0.72***
O-Horizon Properties								
Depth	0.08	-0.19	-0.38	0.14	0.22	-0.01	-0.20	0.34
Mass	0.01	-0.26	-0.4	0.07	0.12	-0.08	-0.22	0.27
Molybdate-Reactive WSP (mg/kg)	0.03	-0.12	0.04	0.05	0.01	-0.15	-0.08	0.01
Total WSP (mg/kg)	0.12	-0.07	0.0	0.14	0.09	-0.16	-0.1	0.06
Molybdate-Reactive WSP (mg/m^2)	0.13	-0.14	-0.31	0.18	0.19	-0.04	-0.17	0.28
Total WSP (mg/m^2)	0.14	-0.13	-0.32	0.19	0.20	-0.04	-0.17	0.29

*significant at $p < 0.05$; **significant at $p < 0.01$; ***significant at $p < 0.001$

3.4 Conclusion

In the Lake Tahoe Basin, the accumulation of highly concentrated P in forest floor O-horizons as a result of legacy fire suppression poses a threat to surface water quality (Miller et al., 2005). Previous research suggests that infiltration of O-horizon leachate may be generating hotspots of labile soil P in various parts of the Basin landscape (Johnson et al., 2011). Subsurface lateral flow through soils is often greater than surface runoff in undisturbed forest soils, and therefore it is a significant transport path for P delivery to surface waters (Elliot et al., 2015). Effective environmental management of soil P transport to surface water relies on tools that can predict these losses using associated landscape characteristics to identify sensitive source areas (Sharpley et al., 1993). Our results shed light on environmental variables and soil properties that influence the distribution of different extractable pools of soil P throughout the Tahoe Basin.

In the surface composite soils, the parent material underlying the soil is an important factor related to concentrations of TP and oxalate P in soil. Soils formed on volcanic and alluvial geology contained higher TP and oxalate P than those formed on glacial drift or granitic geology. These differences were likely associated with parent material influences on sand and poorly crystalline aluminum oxide content. Average soil M3P is not significantly different among the four different classes of underlying geology. Lower TP, lower oxalate P, and higher M3P was observed in dry compared to wet climates.

Regression analysis demonstrated that labile P (M3P) in the soil can be effectively predicted using the selected environmental variables and soil properties, and predictive power increases when grouping soils based on TP status. Differences in the distribution of labile P in soil profiles around the Lake Tahoe Basin mainly occurs in response to climate, with decreases occurring at depth at wetter sites possibly as a result of increased P retention capacity related to poorly crystalline aluminum and iron oxides.

Chapter 4: Key Takeaways for Science and Management of Forested Watersheds

Legacy fire suppression in the Lake Tahoe Basin has resulted in dense forests with thick organic (O) horizons rich in phosphorus that is easily mobilized in the environment, and may be an important source for eutrophication of the Lake (Fisher, 2012). Phosphorus (P) that is solubilized in forest O horizon leachate infiltrates into mineral soil throughout the Basin creating hotspots with above average P concentrations (Johnson et al., 2011). During spring snowmelt, large volumes of water move through subsurface flowpaths, interacting with Basin soils and potentially mobilizing P to surface waters (Elliot et al., 2015; Ohara et al., 2011). Consequently, understanding properties of Lake Tahoe Basin soils that promote accumulation of P in labile forms can support the targeted management of forest debris in sensitive watersheds.

Parent material is the most important soil forming factor influencing the amount of P in forest soils of the Lake Tahoe Basin. Soils formed in granitic and glacial drift parent materials tend to have larger fractions of adsorbed labile soil P (B1P and M3P), despite having lower TP. This was primarily associated with lower levels of poorly crystalline Fe and Al hydroxides, soil minerals that tightly sorb P. In Andisols and other andic soils derived from volcanic parent materials such as basaltic and andesitic tephra, higher amounts of poorly crystalline minerals, especially Al hydroxides, provide an increased capacity to sorb P. For this reason, a higher risk of P release to surface waters may be expected from granitic and glacial drift watersheds containing soils with a lesser ability to filter runoff and O horizon leachate.

Management of forest O horizons as a strategy to reduce P loading to Lake Tahoe may be more successful for granitic and glacial drift soils, especially in watersheds above the West Shore that receive higher mean annual precipitation and deliver more upland runoff. The Lake Tahoe Basin Management Unit uses prescribed burns with regulated intensity as a means for ecosystem restoration and fuel management. Reduction of forest O horizon thickness through prescribed burning is also an effective method for attenuation of watershed nutrient exports to surface waters (Zhang, 2017). Existing online tools such as Web Soil Survey (USDA Natural Resources Conservation Service, 2019) or the SoilWeb mobile applications (O'Green, 2019) are user-friendly interfaces which provide relevant information from the USDA-NCSS geographic soil survey database. Land managers in the Basin can use these tools, for example, to determine if andic properties are present in some level of taxonomy across a watershed's dominant soil types. If present, higher rates of P retention would suggest that labile soil P hotspots are not prevalent, and it would not be necessary to prioritize the watershed.

In some meadow stream environment zones (SEZ) in the Basin, we observed that more labile soil P was immobilized into microbial biomass rather than weakly adsorbed to soil surface exchange sites, regardless of parent material. Therefore, we expect that mechanisms regulating loss of P are different between meadow and forest soils. During the period of spring snowmelt when soils undergo rewetting, leaching of labile soil P from exchange sites to solution is primarily expected in forest soils, while in meadow soils, P may be most prone to solubilization and mineralization from microbes to solution. Further work to directly address contrasting dynamics and mobility of labile soil P between these ecosystems will help to understand potential impacts and rates of loading to surface waters in the Basin. Isotherm and column leaching experiments on corresponding forest and meadow soils from the same study watersheds by Chinmay Deval (results forthcoming) will provide valuable accompanying information to characterize the flux of P through these diverse soil ecosystems.

Results from Chapter 2 show that labile forms of soil P (water-soluble P; WSP) consist of dissolved unreactive forms in all soil types. USGS water quality data from streams near our study watersheds reveal that during periods of peak discharge, the majority of dissolved P is also made of unreactive forms. These soluble unreactive forms are typically associated with dissolved organic P compounds which are often not directly monitored in aquatic ecosystems, but may be bioavailable to some organisms like cyanobacteria in surface waters (Condron et al., 2005). A recent review article by Bol et al. (2016) asserts that dissolved organic P in forest soils is an important component of P loading to surface waters and requires further studying at the watershed scale. Results from the work presented in this thesis suggest that the focus of future science and management considers the role of organic P as a source for P loading in the Lake Tahoe Basin, and is likely an important source in other forested watersheds.

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Appendix A: Chapter 2

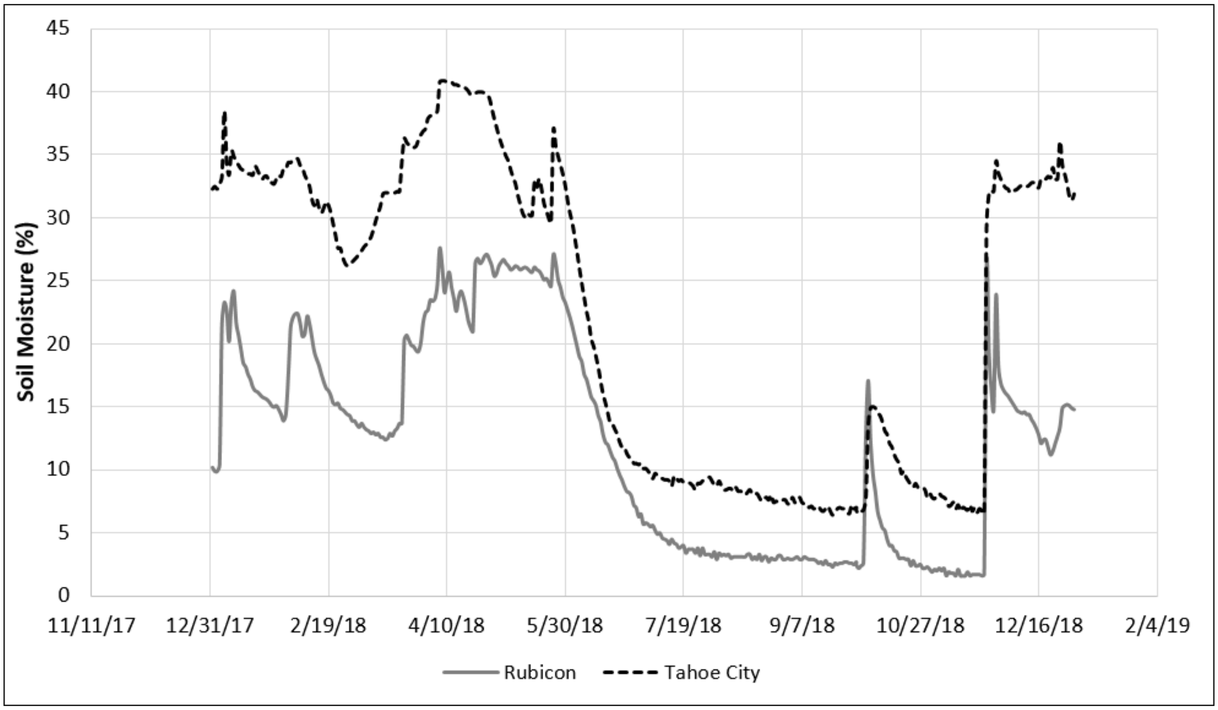

Figure A.1: Time-series of percent soil moisture (depth of 8 inches) at Tahoe City and Rubicon SNOTEL stations (USDA Natural Resources Conservation Service, 2019).

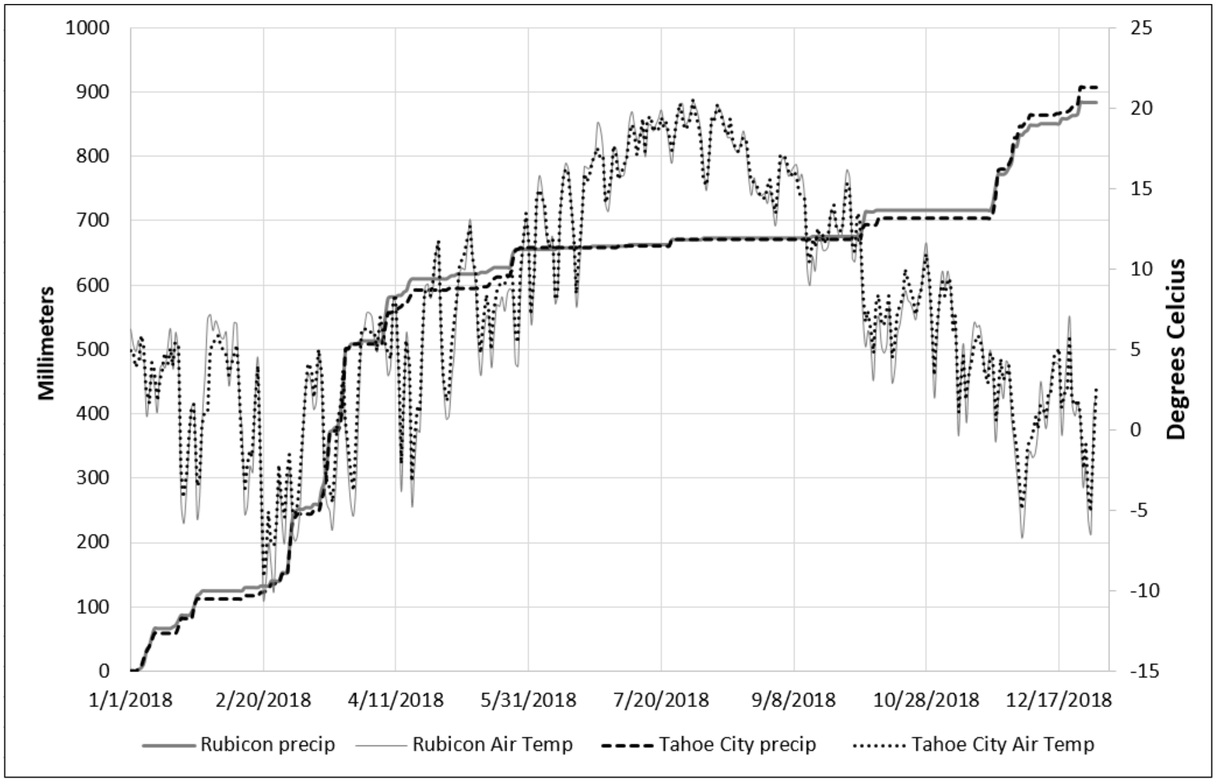

Figure A.2: Cumulative precipitation and daily average air temperature during calendar year 2018 from Tahoe City and Rubicon SNOTEL stations (USDA Natural Resources Conservation Service, 2019).

Appendix B: Chapter 3

Table B.1: Results of laboratory analysis for individual surface composite Basin samples collected during September 2016.

Sample	Coordinates X (degrees)	Coordinates Y (degrees)	Total Phosphorus (mg/kg)	Oxalate Phosphorus (mg/kg)	Mehlich-3 Phosphorus (mg/kg)	pH	Total Nitrogen (%)	Total Carbon (%)	Sand (%)	Oxalate Aluminum (%)	Oxalate Iron (%)	Oxalate Silicon (%)
U1-A3	-120.2053	39.1474	2100	1616	112	6.43	0.415	9.47	78	1.40	0.460	0.171
U1-A4	-120.1930	39.1427	870	613	39.0	4.13	0.272	6.29	78	1.16	0.642	0.272
U1-A5	-120.1986	39.1447	1010	722	33.0	6.02	0.223	6.00	75	1.29	0.543	0.233
U1-A7	-120.1299	39.0446	240	130	63.0	6.46	0.0876	3.01	91	0.178	0.164	0.017
U2-A10	-120.2055	39.1464	800	474	19.0	6.55	0.423	9.84	84	1.27	0.401	0.247
U2-A7	-120.1541	39.1651	2540	1799	105	6.71	0.245	8.83	72	1.06	1.04	0.158
U2-A8	-120.1938	39.1819	1000	562	49.0	6.46	0.190	5.08	78	0.501	0.619	0.0988
U2-A9	-120.1917	39.1446	1270	868	29.0	6.41	0.220	6.32	76	1.39	0.709	0.266
U3-A10	-120.1691	39.1973	1450	956	74.0	7.00	0.152	5.07	78	1.06	0.371	0.239
U3-A11	-120.1704	39.2171	1120	693	135	6.41	0.184	5.12	84	0.404	0.223	0.0256
U3-A7	-120.1383	39.1976	1240	797	11.0	5.96	0.147	3.70	76	1.16	1.60	0.265
U3-A8	-120.1371	39.1970	1500	1091	22.0	6.53	0.103	2.89	80	1.52	1.80	0.671
U4-A3	-120.1444	39.2018	1630	909	50.0	6.17	0.295	7.74	73	1.13	0.949	0.225
U4-A7	-120.1667	39.1620	2700	1857	77.0	6.68	0.215	6.67	75	1.68	0.720	0.298
U4-A8	-120.1508	39.1911	2350	1699	38.0	6.61	0.236	8.28	75	2.56	0.802	0.603
U4-A9	-120.1699	39.2160	910	501	70.0	5.73	0.124	4.68	82	0.570	0.299	0.0454
U5-A10	-120.1072	38.9880	110	65	43.0	5.56	0.0500	1.96	93	0.096	0.051	0.00408
U5-A7	-119.9577	38.7946	680	330	150	5.82	0.105	2.30	95	0.151	0.277	0.0108
U5-A8	-120.1229	39.0100	160	70.9	31.0	6.48	0.0417	2.03	90	0.134	0.144	0.0176
U5-A9	-120.1231	39.0090	140	182	87.0	7.33	0.0605	2.38	91	0.113	0.086	0.0142
U6-A10	-120.1234	39.0084	310	107	38.0	6.43	0.0541	2.15	90	0.128	0.120	0.00930
U6-A7	-120.1262	39.0163	130	50.8	14.0	7.05	0.0559	2.29	89	0.225	0.120	0.0229
U6-A8	-120.0046	38.7859	610	373	105	6.39	0.0541	2.45	89	0.249	0.271	0.0325
U6-A9	-119.9684	38.7964	920	465	81.0	5.78	0.147	3.50	86	0.155	0.612	0.0171
U7-A2	-119.8973	39.0807	900	330	111	4.05	0.0546	1.33	95	0.131	0.200	0.0120
U7-A6	-119.9396	38.9830	510	312	114	6.53	0.0988	2.37	92	0.159	0.243	0.0155
U7-A7	-119.9263	39.1708	660	378	131	6.23	0.103	3.22	88	0.159	0.635	0.00923
U7-A8	-119.9429	39.0326	550	334	143	4.63	0.0664	2.09	88	0.153	0.349	0.0152
U8-A1	-119.9210	39.2569	450	232	87.0	4.09	0.0634	2.16	92	0.070	0.484	0.0217
U8-A4	-119.9572	38.8653	610	434	131	6.34	0.0748	2.82	87	0.201	0.661	0.0184
U8-A6	-119.9258	39.1902	600	309	107	6.99	0.0654	2.27	90	0.121	0.470	0.0174
U8-A7	-119.9497	38.8954	750	484	176	5.84	0.0730	2.36	88	0.202	0.420	0.0133
U8-A8	-119.9695	38.8846	970	685	193	6.18	0.0992	2.97	92	0.319	0.324	0.0242
U9-A2	-119.9352	39.2786	1250	752	97.0	5.28	0.176	4.55	88	0.771	0.362	0.164
U9-A3	-120.0726	39.2603	1220	764	36.0	6.98	0.298	7.39	80	1.29	0.415	0.317
U9-A4	-120.0477	39.2619	2320	1784	89.0	7.54	0.157	4.07	75	1.73	0.598	0.398
U9-A6	-119.9109	39.0872	790	416	98.0	6.51	0.178	5.41	73	0.155	0.324	0.0116
U10-A2	-119.9390	39.2728	4520	1043	58.0	7.09	0.184	4.36	84	1.39	0.499	0.305
U10-A3	-119.9137	39.0881	760	302	53.0	6.54	0.0777	2.53	87	0.099	0.530	0.0171
U10-A5	-119.9336	39.2578	610	386	163	6.65	0.0632	2.04	87	0.237	0.611	0.0247
U10-A6	-119.9307	39.0838	320	147	23.0	7.72	0.0828	2.30	82	0.104	0.375	0.0348
U11-A1	-119.9710	38.8990	490	317	142	5.90	0.0642	2.33	91	0.207	0.574	0.0121
U11-A3	-119.9853	38.8704	390	312	145	5.40	0.0621	1.71	94	0.215	0.481	0.0125
U11-A7	-119.9532	38.8518	580	354	136	5.90	0.0339	1.32	90	0.184	0.730	0.0102
U12-A1	-119.9797	38.9008	350	246	70.0	5.94	0.108	3.35	90	0.216	0.958	0.00398
U12-A2	-119.9529	38.8502	670	365	78.0	6.16	0.189	5.24	83	0.159	0.450	0.00841
U12-A5	-119.9636	38.8881	510	287	132	5.93	0.0646	2.56	90	0.159	0.486	0.0110
U12-A7	-119.9563	38.8551	640	481	101	6.44	0.0675	2.38	83	0.419	0.724	0.0473
U13-A2	-120.1572	39.1455	1160	714	119	7.21	0.0889	2.41	86	0.590	0.698	0.137
U13-A4	-120.0230	38.9137	330	240	98.0	7.22	0.0820	2.12	76	0.263	0.181	0.0326
U14-A3	-120.1653	39.1110	1380	940	53.0	5.84	0.203	5.96	83	1.18	0.539	0.184
U14-A5	-120.1503	39.1512	2340	1488	48.0	6.90	0.129	4.27	80	1.73	0.629	0.473
U14-A7	-120.1624	39.1285	1340	817	34.0	5.02	0.313	6.93	77	0.923	1.03	0.323

Table B.2: Results of laboratory analysis for additional individual surface composite Basin samples collected during September 2016.

Sample	Coordinates X (degrees)	Coordinates Y (degrees)	Total Phosphorus (mg/kg)	Oxalate Phosphorus (mg/kg)	Mehlich-3 Phosphorus (mg/kg)	pH	Total Nitrogen (%)	Total Carbon (%)	Sand (%)	Oxalate Aluminum (%)	Oxalate Iron (%)	Oxalate Silicon (%)
O1	-119.9220	39.2575	600	311	141	5.73	0.0538	1.97	92	0.154	0.530	0.0104
O2	-119.9097	39.0877	490	136	28.0	5.83	0.0740	2.71	77	0.089	0.229	0.00736
O3	-119.8962	39.0803	1020	467	35.0	6.73	0.355	5.27	83	0.104	0.848	0.0617
O4	-120.2326	39.0764	1130	891	57.0	5.72	0.170	3.88	80	1.06	0.574	0.182
O5	-120.2207	39.0748	910	565	27.0	5.79	0.495	8.81	80	0.762	0.602	0.135
O6	-120.2161	39.0797	970	613	35.0	5.69	0.297	6.82	72	0.720	0.640	0.0854
O7	-120.2088	39.0914	1180	770	70.0	6.39	0.498	12.3	86	0.632	0.425	0.0514
O8	-120.2087	39.0922	1030	815	95.0	5.77	0.0916	1.89	83	0.616	0.467	0.0933
O9	-120.1958	39.1047	1420	963	48.0	6.10	0.225	7.16	77	1.17	0.592	0.205
O10	-119.9612	38.8904	560	352	53.0	6.81	0.125	3.15	90	0.100	0.653	0.0445
O11	-119.9718	38.8716	490	315	128	5.99	0.0917	2.89	88	0.199	0.510	0.0159
O12	-119.9534	38.8515	730	318	12.0	5.08	0.960	14.2	85	0.144	1.27	0.0632
O13	-119.9678	38.7965	540	224	18.0	6.88	0.146	3.68	87	0.120	0.583	0.0226
O14	-120.1509	39.1787	3600	2159	99.0	5.38	0.354	9.19	82	1.64	0.605	0.295
O15	-120.1131	39.0500	630	529	147	6.85	0.0965	2.68	91	0.342	0.453	0.0461
O16	-120.1493	39.0435	450	371	108	5.68	0.121	3.27	88	0.420	0.127	0.0887
O17	-120.1231	39.0500	410	247	70.0	4.59	0.0912	2.38	88	0.378	0.166	0.0625

Table B.3: Results of laboratory analysis for individual surface composite Basin samples collected during summer 2017.

Sample	Coordinates X (degrees)	Coordinates Y (degrees)	Total Phosphorus (mg/kg)	Oxalate Phosphorus (mg/kg)	Mehlich-3 Phosphorus (mg/kg)	pH	Total Nitrogen (%)	Total Carbon (%)	Sand (%)	Oxalate Aluminum (%)	Oxalate Iron (%)	Oxalate Silicon (%)
N1	-120.1682	39.0461	710	441	69.0	4.77	0.207	6.43	83	0.707	0.248	0.104
N2	-120.1556	39.0528	760	530	92.0	4.81	0.197	7.36	82	0.766	0.220	0.0866
N3	-120.1502	39.0581	680	491	19.0	4.23	0.248	4.68	64	1.02	0.353	0.250
N4	-120.0775	38.9332	1110	579	53.0	4.49	0.613	12.8	88	0.335	0.447	0.0380
N5	-120.0693	38.9218	320	86.3	11.0	4.00	0.359	13.2	81	0.403	0.395	0.100
N6	-120.0628	38.9314	640	463	69.0	5.36	0.445	9.77	87	0.728	0.228	0.116
N7	-120.0679	38.9215	990	663	54.0	5.23	0.353	8.67	75	1.05	0.245	0.168
N8	-120.0570	38.9258	810	483	141	5.00	0.162	4.81	86	0.388	0.202	0.0720
N9	-120.0452	38.9123	700	402	103	5.29	0.290	8.69	85	0.343	0.209	0.0352
N10	-120.0473	38.9028	310	94.6	17.0	5.50	0.318	12.2	86	0.197	0.149	0.0195
N11	-119.9846	38.9160	360	251	135	5.60	0.0140	1.57	90	0.133	0.171	0.0104
N12	-120.0266	38.8434	420	296	64.0	4.85	0.0815	2.50	82	0.436	0.318	0.0890
N13	-120.0156	38.8425	430	277	128	4.71	0.0839	3.71	95	0.172	0.175	0.00936
N14	-120.0164	38.8337	450	400	205	4.93	0.0380	2.27	90	0.253	0.329	0.0275
N15	-119.9934	38.8757	530	344	92.0	5.42	0.0455	2.56	86	0.402	0.302	0.0408
N16	-119.9526	38.9375	710	413	177	5.16	0.0394	2.17	88	0.189	0.387	0.0116
N17	-119.9212	38.9664	670	317	136	5.44	0.104	4.56	86	0.134	0.289	0.00868
N18	-119.9163	38.9702	560	246	106	5.30	0.0000	0.58	93	0.126	0.386	0.00790
N19	-119.9252	39.1387	810	382	196	5.6	0.0159	1.90	91	0.192	0.246	0.0181
N20	-120.0147	39.2439	1630	1140	159	5.81	0.238	7.39	86	0.768	0.320	0.102
N21	-120.0755	39.2431	1000	568	65.0	5.77	0.419	17.3	79	0.508	0.317	0.0592

Table B.4: Results of laboratory analysis on individual 0 – 5 cm samples.

Sample	**Total Phosphorus** (mg/kg)	**Oxalate Phosphorus** (mg/kg)	**Mehlich-3 Phosphorus** (mg/kg)	**pH**	**Total Nitrogen** (%)	**Total Carbon** (%)	**Sand** (%)	**Oxalate Aluminum** (%)	**Oxalate Iron** (%)	**Oxalate Silicon** (%)
U1-A7	180	101	22.0	6.50	0.166	7.13	96	0.225	0.178	0.0310
U2-A7	2940		153	5.35	0.242	10.6				
U3-A10	890	623	35.0	6.00	0.116	6.30	83	1.35	0.483	0.233
U4-A8	2890	2012	101	5.89	0.651	22.7	89	1.79	0.682	0.184
U4-A9	540	673	57.0	5.56	0.265	14.8	81	0.801	0.507	0.0647
U5-A10	120	60	27.0	4.88	0.0405	2.97	99	0.0595	0.0528	0.0030
U5-A8	200	74	16.0	5.03	0.272	12.8	90	0.137	0.189	0.0148
U5-A9	350	336	130	6.94	0.0459	2.31	92	0.134	0.143	0.0190
U6-A7	260	238	76.0	6.51	0.0139	1.98	92	0.302	0.202	0.0604
U6-A8	620	345	73.0	5.97	0.148	6.43	93	0.242	0.319	0.0264
U7-A2	830	601	128	5.72	0.186	5.21	99	0.202	0.430	0.0204
U7-A8	470	395	119	5.51	0.113	4.30	88	0.167	0.318	0.0115
U8-A4	360	248	39.0	5.32	0.162	7.32	93	0.112	0.486	0.0147
U8-A8	1030	865	189	5.48	0.0945	4.21	94	0.459	0.368	0.0359
U9-A4	3930	3261	155	7.28	0.0719	3.08	70	1.65	0.514	0.430
U9-A6	650	241	30.0	5.99	0.442	12.6	91	0.142	0.378	0.0137
U10-A2	1820	1259	83.0	5.21	0.150	4.82	86	1.60	0.592	0.340
U10-A5	470	395	108	6.00	0.0446	2.69	97	0.264	0.467	0.0225
U11-A1	360	167	35.0		0.589	30.9	95	0.197	0.383	0.0093
U11-A3	360	287	139	5.65	0.0289	1.40	92	0.197	0.448	0.0244
U12-A2	460	271	54.0	5.70	0.158	5.16	97	0.131	0.345	0.0129
U12-A5	430	268	89.0	5.22	0.0570	3.96	95	0.158	0.654	0.0117
U13-A2	1160	931	177	6.47	0.220	6.87	92	0.653	0.615	0.143
U13-A4	610	407	103	4.96	0.0570	3.03	85	0.315	0.223	0.0230
U14-A5	1830	1153	90.0	6.77	0.544	18.0	88	1.19	0.547	0.213

Table B.5: Results of laboratory analysis on individual 5 – 20 cm samples.

Sample	**Total Phosphorus** (mg/kg)	**Oxalate Phosphorus** (mg/kg)	**Mehlich-3 Phosphorus** (mg/kg)	**pH**	**Total Nitrogen** (%)	**Total Carbon** (%)	**Sand** (%)	**Oxalate Aluminum** (%)	**Oxalate Iron** (%)	**Oxalate Silicon** (%)
U1-A7	170	84	23.0	6.11	0	1.307117	95	0.230417	0.232	0.031153
U2-A8	1010	0	36.0							
U3-A10	990	527	41.0	5.30	0.300	13.1	85	1.39	0.582	0.231
U4-A8	3390	1867	70.0	5.01	0.222	8.26	83	2.49	0.871	0.381
U4-A9	680	472	33.0	5.14	0.0352	2.87	82	0.945	0.545	0.147
U5-A10	120	96	38.0	5.23	0.00464	1.74	89	0.170	0.106	0.0091
U5-A8	100	39	7.00		0.00259	1.56	91	0.220	0.291	0.0305
U5-A9	180	133	51.0	6.57	0.0222	1.98	94	0.150	0.245	0.0218
U6-A7	140	62	19.0	6.47	0.00179	1.82	91	0.337	0.194	0.0541
U6-A8	550	292	66.0	5.45	0.0784	5.64	91	0.369	0.450	0.0321
U7-A2	1320	507	119	5.85	0.0170	0.791	93	0.240	0.490	0.0272
U7-A8	650	480	144	6.12	0.0283	1.81	91	0.249	0.474	0.0209
U8-A4	460	391	76.0	5.52	0.00582	1.66	91	0.135	1.21	0.0255
U8-A8	1190	1020	217	5.53	0.0295	2.82	92	0.597	0.627	0.0601
U9-A4	2460	2193	117	7.05	0.0485	2.51	75	2.01	0.796	0.532
U9-A6	830	593	87.0	5.73	0.0819	4.03	80	0.237	0.547	0.0217
U10-A2	1020	597	27.0	5.21	0.0933	4.28	90	1.46	0.741	0.3758
U10-A5	580	352	109	5.51	0.0155	1.88	92	0.312	0.514	0.0322
U11-A1	380	371	111	4.92	0.0810	7.24	93	0.289	0.799	0.0230
U11-A3	430	400	139	5.24	0.00565	0.903	95	0.328	0.668	0.0438
U12-A2	520	313	33.0	5.14	0.266	10.1	95	0.210	0.552	0.0272
U12-A5	430	359	100	5.37	0.00802	1.55	93	0.231	0.890	0.0230
U13-A2	1180	804	111	6.33	0.0506	2.89	94	0.759	0.982	0.177
U13-A4	420	433	107	5.51	0.0634	3.18	82	0.414	0.273	0.0557
U14-A5	1790	1542	75.0	5.90	0.177	9.46	85	1.82	0.850	0.348

Table B.6: Results of laboratory analysis on individual 20 – 40 cm samples.

Sample	**Total Phosphorus** (mg/kg)	**Oxalate Phosphorus** (mg/kg)	**Mehlich-3 Phosphorus** (mg/kg)	**pH**	**Total Nitrogen** (%)	**Total Carbon** (%)	**Sand** (%)	**Oxalate Aluminum** (%)	**Oxalate Iron** (%)	**Oxalate Silicon** (%)
U1-A7	100	42	13.0	4.70	0.00	0.957	96	0.230	0.272	0.0257
U3-A10	870	523	18.0	4.76	0.0445	5.40	88	1.47	0.758	0.396
U4-A8	1820	888	9.0	5.08	0.0314	2.42	88	2.10	1.05	0.509
U4-A9	840	769	40.0	5.11	0.00584	1.00	81	1.39	0.675	0.411
U5-A9	70	28	9.0	6.10	0.00	0.800	95	0.138	0.258	0.0190
U6-A7	90	24	4.0	6.37	0.00	1.22	93	0.321	0.194	0.0458
U6-A8	430	214	30.0	4.68	0.00	1.06	91	0.420	0.500	0.0815
U7-A2	880	502	100	6.11	0.00224	0.582	95	0.272	0.448	0.0391
U7-A8	490	281	100	5.04	0.00410	0.915	91	0.165	0.452	0.0218
U8-A4	480	378	77.0	5.51	0.00619	2.16	92	0.137	1.10	0.0278
U8-A8	1030	741	176	5.30	0.00376	1.16	95	0.459	0.515	0.0748
U9-A6	740	472	118	4.87	0.0150	1.62	71	0.168	0.603	0.0174
U10-A2	810	458	15.0	5.20	0.0388	2.74	91	1.43	0.772	0.3744
U10-A5	350	200	64.0	5.59	0.00255	1.27	94	0.199	0.494	0.0299
U11-A1	420	375	163	5.71	0.00629	1.43	93	0.265	0.539	0.0339
U11-A3	350	332	93.0	5.11	0.000069	0.499	95	0.356	0.591	0.0571
U12-A2	600	480	67.0	5.18	0.0360	1.84	95	0.224	0.770	0.0402
U12-A5	430	315	94.0	5.16	0.00	0.855	95	0.210	0.821	0.0306
U13-A2	850	482	39.0	6.06	0.00528	1.34	97	0.689	0.899	0.176
U13-A4	70	20	7.0	5.10	0.325	10.7	73	0.117	0.117	0.0208
U14-A5	1120	835	40.0	5.81	0.0591	2.99	93	1.34	0.945	0.297

Table B.7: Bulk density of all individual samples (0 – 5, 5 – 20, and 20 – 40 cm) collected in September 2016

Name	**0 - 5 cm**	**5 - 20 cm**	**20 - 40 cm**	**Name**	**0 - 5 cm**	**5 - 20 cm**	**20 - 40 cm**
U1-A3	1.28			U8-A1	1.71	2.11	
U1-A4	0.850	0.888	0.875	U8-A4	1.56	1.63	
U1-A5	1.10	1.11	0.548	U8-A6	1.05	1.55	1.84
U1-A7	1.51	1.51	2.12	U8-A7	1.76	1.77	1.48
U2-A10	0.807	1.22	0.942	U8-A8	1.47	1.48	1.93
U2-A8	0.504			U9-A2	1.42	1.33	1.72
U2-A9	0.973	1.17		U9-A3	1.13	0.948	1.36
U3-A10	1.37	1.26	1.24	U9-A4	1.25		
U3-A11	1.29	1.38		U9-A6	1.27	1.46	1.35
U3-A7	1.35	1.18		U10-A2	1.05	0.975	1.08
U3-A8	1.30	1.15		U10-A3	1.87	1.11	1.64
U4-A3	0.743	0.957		U10-A5	1.55	1.33	1.74
U4-A7	1.17			U10-A6	1.60	1.46	1.41
U4-A9	1.89	1.39	0.923	U11-A1	1.48	1.78	1.60
U5-A10	1.64			U11-A3	1.62	1.33	1.64
U5-A7	1.70	1.63		U11-A7	2.20	2.45	2.19
U5-A8	1.35	1.51	1.28	U12-A1	1.90	1.71	1.34
U5-A9	1.64	1.36	1.62	U12-A2	0.973	1.32	1.61
U6-A10	1.33	1.23		U12-A5	1.43	1.75	1.63
U6-A7	1.43	1.47		U12-A7	1.47	1.63	2.01
U6-A8	1.46	1.44	1.31	U13-A2	1.09	1.48	1.20
U6-A9	0.900	1.35	1.44	U13-A4	1.48	0.951	1.69
U7-A2	1.38	1.59	1.57	U14-A3	0.903	1.28	1.08
U7-A6	1.42	1.56	2.45	U14-A5	1.34	1.37	1.34
U7-A7		1.70		U14-A7	0.522	0.855	1.20
U7-A8	1.84	1.56	2.08				

Appendix C: Phosphorus in Forest Seeps at the Granitic Watershed

During summer of 2017, several lysimeters and wells were installed throughout the Meeks Watershed in five of our meadow sampling sites as well as four seep locations, which were areas along the forested hillslope where water exfiltrates at or near the soil surface. Lysimeters, polyvinyl chloride (PVC) pipes with a porous cup on the bottom end, were installed to a soil depth of 15 cm and placed under vacuum between sample collections. Perforated PVC wells were installed to 30 cm, allowing for the collection of saturated soil water. Solution from lysimeters and wells were collected on a weekly basis during July and August, filtered through 0.45-µm PES membranes, analyzed for molybdate-reactive P in solution on a spectrophotometer using the Murphy-Riley method for dissolved reactive P in water (Kovar & Pierzynski, 2009). Soil samples were collected from the seeps three times during June, August, and October of 2018. To preserve potential anoxic conditions, soils from seeps were sampled by flushing with nitrogen gas for 3 minutes immediately after collection; then stored in secondary containment that was flushed for an additional 1 minute. Water-soluble P (WSP) and Bray-1 P (B1P) in the seep soils were measured according to methods outlined in Chapter 2 of this thesis.

Repeated measures of phosphorus from wells, lysimeters, and soil are summarized in Tables C.1, C.2, and C.3, respectively. Average solution phosphorus measured in wells from seeps was lower than that observed in wells from the meadow. Lysimeters in the meadows did not collect enough water for sample collection on most occasions; only three samples were collected during the 8-week period during the first half of July from three different sampling sites (Table C.2). Sample collection from lysimeters in seeps was much more consistent, and a steady discharge of P in solution was observed in seep lysimeters for the duration of the 8-week period, except at site G8. When excluding site G8, the average of lysimeter solution P was considerably higher than two of the three individual samples collected from meadow lysimeters.

Average Bray-1 phosphorus (B1P) and water-soluble phosphorus (WSP) was noticeably lower in seep soils compared to all other forest soils sampled (Table C.4). B1P and WSP were similar between seep and meadow soils. Continuous exfiltration of water and detectable lysimeter P from seep soils suggests that lower soil B1P concentrations in seeps compared to other forested hillslope sites could be the result of persistent leaching processes that are depleting the sink of labile adsorbed P. A more thorough investigation into solid-solution phase P dynamics in these soils would offer insight about the degree to which seeps may act as a perennial point-source of P being exported from the watershed.

Table C.1: Weekly measurements of dissolved molybdate-reactive phosphorus (mg/l) from wells installed in meadow and seep soils.

	Meadow Sites					**Seep Sites**			
Collection Date	G1	G2	G3	G6	G7	G4	G5	G8	G10
7-Jul	0.0128	(BDL)	-	0.0729	(BDL)	0.0128	-	-	0.0128
14-Jul	-	-	0.0363	0.0975	0.0404	-	-	-	-
21-Jul	0.0790	1.08	-	-	-	-	-	-	(BDL)
27-Jul	0.00847	0.0862	-	-	-	-	-	-	0.00847
4-Aug	(BDL)	0.203	-	-	-	-	-	-	0.0179
11-Aug	(BDL)	0.222	-	-	-	-	-	-	0.321
18-Aug	(BDL)	-	0.307	-	-	(BDL)	-	-	0.218
23-Aug	(BDL)	-	0.0563	-	-	0.0114	-	-	0.0393
Average	0.0172	0.319	0.133	0.0852	0.0227	0.00972	-	-	0.0882
(SD)	(0.0254)	(0.388)	(0.123)	(0.0123)	(0.0177)	(0.00338)	-	-	(0.118)

Minimum Detection Limit = 0.01 mg/l; dashes indicate no sample was collected; SD = standard deviation; A value of one-half the detection limit (0.005 mg/l) was attributed to measurements below detection limit (BDL) to calculate averages

Table C.2: Weekly measurements of dissolved molybdate-reactive phosphorus (mg/l) from lysimeters installed in meadow and seep soils.

	Meadow Sites					**Seep Sites**			
Collection Date	G1	G2	G3	G6	G7	G4	G5	G8	G10
7-Jul	-	-	-	0.0299	-	0.0128	(BDL)	-	(BDL)
14-Jul	-	-	-	-	-	0.118	0.0975	-	-
21-Jul	-	-	0.0762	-	0.281	0.174	0.221	(BDL)	0.0762
27-Jul	-	-	-	-	-	-	0.175	-	0.0862
4-Aug	-	-	-	-	-	0.153	0.153	-	0.0491
11-Aug	-	-	-	-	-	0.170	0.222	-	0.0452
18-Aug	-	-	-	-	-	-	0.218	-	0.0403
23-Aug	-	-	-	-	-	0.0393	0.162	-	0.0246
Average	-	-	N/A	N/A	N/A	0.111	0.157	N/A	0.0467
SD	-	-				(0.0634)	(0.0700)		(0.0259)

Minimum Detection Limit = 0.01 mg/l; dashes indicate no sample collected
A value of one-half the detection limit (0.005 mg/l) was attributed to measurements below detection limit (BDL) to calculate averages

Table C.3: Water-soluble phosphorus and Bray-1 phosphorus during June, August, and October at sites where lysimeters and wells were installed.

Sample Name	Month	Water-soluble Phosphorus	Bray-1 Phosphorus
Seep			
G4	June	0.531	9.23
	August	0.599	6.58
	October	0.110	8.28
	Average	0.413	8.03
	(SD)	(0.216)	(1.10)
G5	June	0.699	16.1
	August	0.560	8.75
	October	0.727	13.1
	Average	0.662	12.6
	(SD)	(0.0730)	(3.02)
Meadow			
G1	June	0.734	3.87
	August	0.498	3.60
	October	0.361	4.93
	Average	0.531	4.13
	(SD)	(0.154)	(0.572)
G2	June	0.985	10.5
	August	1.82	10.6
	October	1.28	8.74
	Average	1.36	9.94
	(SD)	(0.346)	(0.846)
G3	June	0.762	4.32
	August	0.354	2.82
	October	0.381	1.73
	Average	0.499	2.96
	(SD)	(0.186)	(1.06)
G6	June	0.387	11.4
	August	0.571	13.5
	October	0.435	9.33
	Average	0.464	11.4
	(SD)	(0.0779)	(1.70)
G7	June	0.508	4.78
	August	0.161	6.24
	October	0.334	1.85
	Average	0.334	4.29
	(SD)	(0.142)	(1.82)

Table C.4: Soil (WSP & B1P) and solution (lysimeter & well) phosphorus averages of repeated measures for each individual sampling site, as well as overall site-type averages.

Sample Name	**Water-soluble Phosphorus**	**Bray-1 Phosphorus**	**Lysimeter Phosphorus**	**Well Phosphorus**
	mg/kg		mg/l	
Seep				
G4	0.413	8.03	0.111	0.00972
G5	0.662	12.6	0.157	-
G8	-	-	0.00505[ˆ]	-
G10	-	-	0.0467	0.0890
Average	0.538	10.3	0.0800	0.0494
(SD)	(0.124)	(2.31)	(0.0452)	(0.0396)
Meadow				
G1	0.531	4.13	-	0.0172
G2	1.36	9.94	-	0.319
G3	0.499	2.96	0.0762[ˆ]	0.133
G6	0.464	11.4	0.0299[ˆ]	0.0852
G7	0.334	4.29	0.281[ˆ]	0.0227
Average	0.638	6.55	0.129	0.116
(SD)	(0.369)	(3.44)	(0.0945)	(0.111)
Forest Average[*]	1.89	106	-	-
(SD)	(0.828)	(50.9)	-	-

[ˆ] Only one sample collected during 8 week period, thus single observation presented

[*] Averages from all eight sampling sites across June, August, October from Chapter 2