TOWARD IMPROVED WATER QUALITY:

CONNECTING HYDROLOGY, MANAGEMENT AND COLLABORATION

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Authorization to Submit Thesis

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Abstract

Improvements in surface water quality require full understanding of the impacts of climate, land use, soils, and topography. An interdisciplinary approach was used to understand how to achieve reductions in pollutant loads. Chapter 1 analyzes extreme erosion events based on spatial, temporal, and climatic factors in Paradise Creek, Idaho. Peak discharge, event duration, and antecedent moisture conditions explained most of the variation in sediment load. Chapter 2 is a thorough literature review on effectiveness of agricultural best management practices, and presents a conceptual framework for identifying such practices for respective dominant hydrologic flow paths. Chapter 3 assesses the function and dynamics of collaborative stakeholder groups within three watersheds that have successfully reduced pollutant levels. Participants saw monitoring and data collection as the ultimate measure of success, despite funding constraints, lag time of pollutant response, and the inability to attribute successes to a specific entity or action.

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Dedication

I would like to thank my Guatemalan street dog, Patita, for surviving through my long days at the office with infrequent walks.

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Chapter 1: Characterization of Extreme Erosion Events: The Role of Spatial, Temporal and Climatic Forces in Generating the Greatest Sediment Loads

1. Abstract

Infrequent, large storms generate erosion events that disproportionately contribute to annual sediment loading totals. Extreme erosion events have a negative impact on downstream water quality and habitat, and contribute to the loss of agricultural topsoil. With climate change predictions of an increased frequency of high intensity rainfall events, it is important to understand and characterize the driving forces in order to better control for them. In this study, we analyzed 12 years of continuous streamflow, precipitation, and watershed data at two monitoring stations in Paradise Creek Watershed, Idaho. We identified 137 erosion events in the upstream rural section of the watershed and 191 events in the downstream urban section. We statistically analyzed the influence of two dozen explanatory variables on event sediment load. Using multiple linear regression and ANOVA, we found that peak discharge, event duration and antecedent baseflow explained most of the variation in event sediment load at both stations and for the watershed as a whole (R^2 between 0.725 and 0.775). In the rural area, saturated soils combined with spring snowmelt in March led to extreme events. The urban area load contribution peaked in January, which could be a re-suspension of streambed sediments from the previous water year.

2. Introduction

Soil erosion and the ensuing elevated sediment loads in surface water bodies may result in impaired water quality and unsuitable habitat for salmonid species and other cold water biota (Owens *et al.*, 2005). Chronic turbidity can negatively affect the growth of young salmonid species (Sigler *et al.*, 1984). Increased sediment loads also are known to be related to high nutrient levels in streams at downstream locations as phosphorus tends to bind to soil particles, leading to algae growth and eutrophication of water bodies (Withers and Jarvis, 1998; Sharpley *et al.*, 2000). In Idaho, sediment is recognized as the most common pollutant in streams (Rowe *et al.*, 2003). Furthermore, the loss of topsoil has economic impacts on agricultural communities (Boardman *et al.*, 2003).

In many streams and rivers, the majority of sediment loads is delivered during a small proportion of the year, specifically during a few large storm events (Larson *et al.*, 1997; Boardman, 2006). In a 28 year study of nine watersheds in the North Appalachian Experimental Watershed in Ohio, Edwards and Owen (1991) identified that the five biggest erosion events within each watershed produced 66% of the total erosion. In eastern Oregon, Zuzel *et al.* (1993) found, over a 12 year period, that 10% of events resulted in 60-70% of total erosion. González-Hidalgo *et al.* (2009), in an analysis of 27,857 daily erosion events from the USDA USLE database, determined that the top 10% of daily events, on average, accounted for 50% of eroded soil. For several Wisconsin watersheds, Danz *et al.* (2013) calculated that the largest 10% of events in each watershed represented 73-97% of total sediment load.

Agricultural best management practices (BMPs) for controlling erosion tend to work well for average storms but may not achieve desired results during large storms (Edwards and Owens, 1991; Larson *et al.*, 1997; González-Hidalgo *et al.*, 2009; Danz *et al.*, 2013) when hydrologically sensitive areas contribute the greatest amount of runoff and erosion (Walter *et al.*, 2000). In order to reduce sediment loads, the conditions that lead to extreme events need to be identified in order to appropriately target BMPs for the most sensitive time periods and locations. Specifically, temporal and spatial variability of erosion need to be considered to gain a better understand of how to control for extreme events (Boardman, 2006; González-Hidalgo *et al.*, 2009).

Many studies have examined hydrologic events to determine the factors that explain runoff response. High antecedent soil moisture is often cited as the driving force in runoff generation (Seeger *et al.*, 2004; Zehe and Blöschl, 2004; Merz *et al.*, 2006; Rodriguez-Blanco *et al.*, 2012), especially when saturation excess overland flow is the dominant runoff process (Dunne and Black, 1970; García-Ruiz *et al.* 2005). Precipitation depth and intensity play more prominent roles when infiltration excess overland flow is dominant (Lana-Renault *et al.*, 2007; Nadal-Romero *et al.*, 2008).

While the factors that lead to runoff generation are generally well understood, the intricacies behind erosion events still require more research (Boardman, 2006). Zuzel *et al.* (1993), in a study in the dryland wheat growing region in the Pacific Northwest, determined that 88% of runoff and erosion occurred between December and March. In an analysis of flood generation and sediment transport in the Central Spanish Pyrenees, García-Ruiz *et al.* (2008) observed that land cover impacted the seasonality and intensity of response. In an abandoned farmland catchment also in the Central Spanish Pyrenees, Lana-Renault and Regüés (2009) concluded that sediment yield was greater during the hydrologically-active spring and responded to high rainfall intensity and antecedent baseflow. Danz *et al.* (2013) statistically analyzed the explanatory power of several weather and watershed conditions on sediment load response in eight Wisconsin watersheds and found that rainfall depth and antecedent baseflow were the most important predictors of sediment load for rainfall events; for snowfall events, load response was predicted by antecedent baseflow combined with variables that indicated an addition of water to the system and melt conditions. Eder *et al.* (2013) found that, depending on the deposition characteristics of the previous event, bed sediments were re-suspended during the “first flush” period of the rising limb of event hydrographs. In the Paradise Creek Watershed, the location of the present study, stream bed sediments may contribute a significant proportion of the annual sediment load (Brooks *et al.*, 2010).

Climate change predictions suggest that both total rainfall and the frequency of high intensity rainfall events will increase (Nearing *et al.*, 2004; Boardman, 2006). Because the largest events contribute a disproportionate amount of the sediment load, an increase in the number of extreme events may lead to greater soil erosion rates. Additionally, greater total rainfall from global warming may lead to increased soil moisture over longer periods, also resulting in greater erosion rates. Already moist climates, such as the Pacific Northwest, may experience these impacts more heavily.

In this research, we look beyond the conclusion that large events contribute the majority of sediment loads by investigating the driving forces behind each event. The primary objective of this research is to characterize the “extreme erosion events” in a mixed land use watershed. Specific objectives for this study are to: (1) characterize the sediment delivery behavior of events with the greatest sediment loads as either transport- or supply-limited; (2) identify the climate, watershed and antecedent conditions associated with the biggest erosion events; (3) quantify and determine the seasonal distribution of sediment loads produced from rural and urban land uses.

3. Methods

3.1 Study Area

The Paradise Creek Watershed (PCW; 4,890 ha; HUC 17060108) is located in the Palouse River hydrologic basin within the Northwest Wheat and Range Region in north central Idaho and eastern Washington. This study focuses on the Idaho segment of the creek from its headwaters to the state line. The watershed is comprised primarily of rural areas (62%) with urban and forest making up 20% and 18%, respectively (Fig. 1.1). Elevation ranges from 770 m to 1,330 m and precipitation from 650 mm to 1,000 mm with 70% falling in the winter months as rain and snow (Brooks *et al.*, 2010). The headwaters of Paradise Creek are on Moscow Mountain, which is forested with limited logging and steep gradients. The creek then runs through approximately nine kilometers of agricultural lands comprised mainly of winter wheat – spring grain – pea rotations. Conservation tillage is readily practiced with some conventional tillage remaining. The predominant soil in the agricultural area is

Palouse silt loam, which is deep and well-draining. Soils with intermittent shallow argillic horizons are also present (e.g. Southwick, Garfield and Thatuna). Downstream of the agricultural lands, Paradise Creek passes through Moscow, ID, an urban area of approximately 24,000 people. The Moscow Wastewater Treatment Plant (WWTP) has an outflow into the creek at the end of the urban area, just above the state line with Washington (46°43'54.500"N; 117°02'04.297"W).

A Total Maximum Daily Load (TMDL) was written for Paradise Creek in 1997 for multiple pollutants, including nutrients, sediment, habitat alteration, pathogens, flow alteration and temperature (IDEQ, 1997). The main nonpoint source pollutant of concern in PCW is sediment because of the historically high sediment loads in the region (Kok *et al.*, 2009). In the TMDL, the annual allowable sediment load from nonpoint sources was set at 260 t/yr, requiring a 75% reduction from 1997 levels (IDEQ, 1997). That target has not yet been consistently achieved but significant improvements have been made (Brooks *et al.*, 2010) (Fig. 1.2). Since the writing of the TMDL, the target has been met in 2001, 2005, 2010 and 2013.

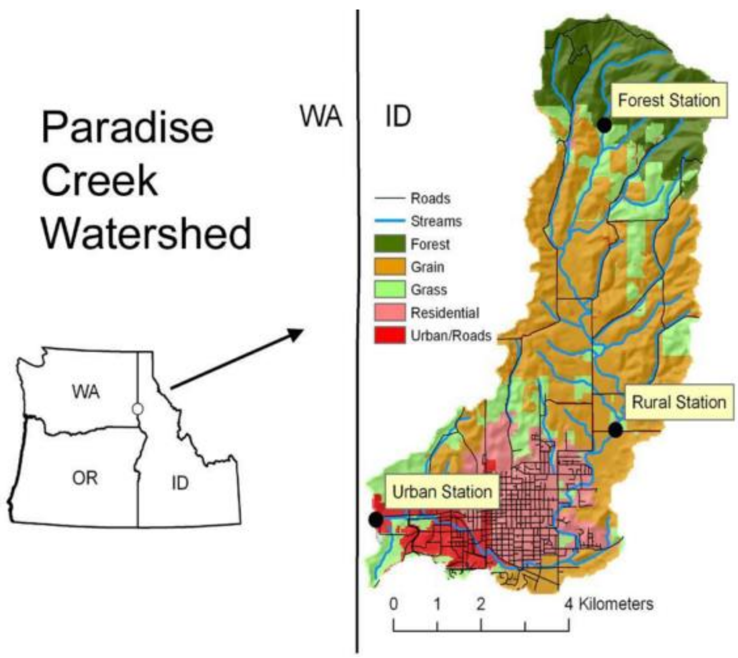


Figure 1.1 Location of the Paradise Creek Watershed in the Palouse region of the Pacific Northwest, and land use and sampling stations. The forest station was not used in the present study.

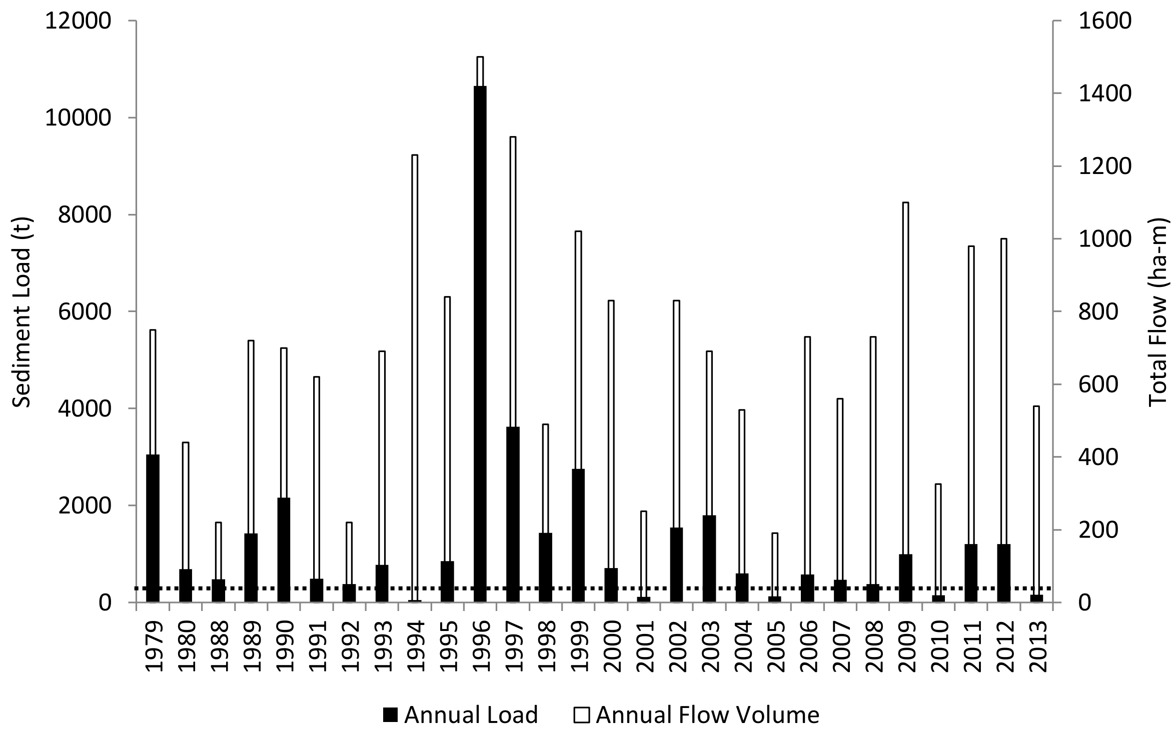


Figure 1.2 PCW annual sediment loads and flow, 1979-80 and 1988-2013 at watershed outlet (adapted from Brooks *et al.*, 2010). Dashed line indicates TMDL annual sediment load target for watershed outlet. Notice high load during 1996 flood year.

3.2 Monitoring Data

We utilized long-term, event-based data (2001 – 2013) from two monitoring stations on Paradise Creek, one located at the approximate transition between rural/agricultural and urban land uses (46°44'55.2048"N; 116°57'46.872"W), the other below the urban area above the outlet of the wastewater treatment plant (WWTP) (46°43'54.220"N; 117°2'4.394"W) near the Idaho-Washington state line (Fig. 1.1). Paradise Creek is ephemeral in the forested section, mainly flowing during snow melt periods when it contributes an insignificant sediment load.

Continuous 15-minute water quality and hydrological data have been collected at these stations since November 2000 (rural station) and November 2001 (urban station). In this analysis, we used stage height (m) and turbidity (NTU) from the continuous data (see Brooks *et al.*, 2010 for detailed sampling methods). We developed a rating curve based on manual discharge measurements (velocity-area method) during the collection period to obtain continuous discharge data from stage height. For 2011-13, we used discharge data from a USGS gaging station located near the urban station in order to remove noise in the data from WWTP backflow. We collected weekly and event samples based on height and flow-volume thresholds which were used to measure total suspended sediment concentration (TSS) (mg/L) using vacuum filtration standard method 2540D (Eaton *et al.*, 1995). To obtain continuous TSS data, we found a regression relationship between turbidity and measured TSS. We calculated sediment load (kg/15 min) by multiplying 15-minute sediment concentration and 15-minute discharge, and then summing for each event and each water year. We subtracted the load calculated at the rural station from the load at the urban station in order to isolate the influence of each land use on the event and annual sediment loads. We refer to those loads as *rural* and *urban* throughout the analysis, while we refer to the total load measured at the urban station as *watershed outlet*. The urban load is an approximation of the actual load because some sediment between the rural and urban station may have been deposited or re-suspended in the stream channel during a given event or year. A small tributary (Hog Creek) with rural land use before entering the urban area also was attributed to the urban load.

We collected precipitation (depth, timing and intensity) and snowfall data using a tipping bucket rain gauge with snow adaptor at the rural station. During periods when that station malfunctioned, we supplemented data from a nearby monitoring station at the Cook Agronomy Farm (CAF) (46°46'42.067"N; 117°5'21.533"W) near Pullman, WA using a correction factor calculated from total annual precipitation for each location. We obtained daily data for snowmelt, frozen soils, and snow cover from the University of Idaho Plant Science Farm in Moscow, ID (46°43'33.013"N; 116°57'18.388"W).

3.3 Event Analysis

3.3.1 Event Selection

Runoff and erosion events were selected during water years 2001 – 2013 by creating annual hydrographs for each station and visually identifying all peaks. We did not use a specific numerical cutoff in the initial selection of events and thus included all events that appreciably impacted the hydrograph or sediment load. Events started when discharge began to increase consistently above base flow with little noise in the data. Events ended when discharge returned to baseflow (either antecedent levels or a consistent elevated level due to seasonal baseflow increases), identified using the constant slope method (Hewlett and Hibbert, 1967). Generally, events that occurred early or late in the water year returned to antecedent baseflow. For events during late winter and spring, baseflow rarely returned to antecedent levels, so we selected the end of the event when the decrease in discharge was minimal to none, or when the next event began (if events were close to each other). Occasionally, small peaks occurred during the falling limb of a larger event which were included within the larger event. Multiple peaks in a short period of time occurred often during spring snow melt and were considered to be one event because discharge never approached baseflow levels. After visually approximating runoff events in the hydrograph, we found corresponding dates in the continuous streamflow data and recorded specific start and end times. Using a lower limit of 1 t for sediment load, we identified 137 events at the rural station and 191 at the watershed outlet.

We identified precipitation events as the start of precipitation to the end of a runoff event using the continuous climate data. We selected the start of a precipitation event at either the start of the runoff event, or the beginning of precipitation in the time period leading up to the runoff event if there was consistent precipitation (i.e., > 0.25 mm recorded hourly). We did not include sporadic small amounts of precipitation (< 0.5 mm greater than 2 hours apart) recorded prior to the start of the runoff event.

Variables analyzed for each event are described throughout the remainder of this section and listed in Table 1.1. Total precipitation depth (*P*) is the sum of precipitation from the start of the precipitation event to the end of the runoff event; we also identified peak precipitation intensity (*i*) and time to peak precipitation (t_p).We recorded snowfall (*S*) during runoff events using data collected at the University of Idaho Plant Science Farm that separated out snowfall (as depth of snow) in the observed precipitation. We summed daily amounts beginning one day prior to the start of the runoff event (if runoff began in morning hours) and the same day as the runoff start (if runoff began after 12:00 hours). We assumed a snow water equivalent (SWE) of 15% for new snow. We did not include snowfall occurring on the final day of the runoff event since it would not have had time to melt and contribute to streamflow. We recorded the presence of snow cover during the rising limb, falling limb or entirety of the event. We calculated snowmelt (*M*) as the difference between the depth of snow cover at the beginning of the event, including one day prior, and the depth at the end of the event. If the event ended in early morning hours (i.e., before solar radiation is present to melt the snow), we did not include the final day in this calculation since snow cover observations were recorded at 17:00 hours. We assumed a SWE of 30% for older snow.

We calculated net $P + M$, or the available inflow (*A*) for each event to account for both precipitation immediately contributing to the system and snowmelt:

$$A = P - S + M \tag{1}$$

where *A* is available inflow (mm), *P* is total measured precipitation (mm) using a tipping bucket rain gauge with snow adaptor, *S* is snowfall SWE (mm), and *M* is snowmelt SWE (mm). In a small number of events when measured snowfall was much greater than measured precipitation (perhaps due to instrument malfunction or precipitation differences between sites), a negative value resulted from $P - S$. For these events, we set net precipitation to zero, since snowfall was the dominant component, implying that snow density was lower (e.g. 10%) or that the newly fallen snow did not contribute to streamflow during that event.

Examining the relationship between *A* and streamflow depth (*D*) can indicate the proportion of direct flow during a runoff event (Merz *et al.*, 2006; Lana-Renault *et al.*, 2009). We found *D* by calculating the total runoff volume during an event and dividing it by the area of the catchment. We calculated the stormflow coefficient (*SC*) for each event by finding the ratio of streamflow depth to available inflow:

$$SC = \frac{D}{A} \tag{2}$$

The resulting value indicates the proportion of available inflow (precipitation and snowmelt) that enters the stream through direct flow rather than baseflow. A greater value indicates greater amounts of direct flow. In this research, we also used antecedent baseflow (*B*), 7 day antecedent precipitation (P_7), and water year cumulative precipitation (*CP*) as surrogates for antecedent moisture conditions in the watershed.

Using frost tube data at the Plant Science Farm we identified the presence of frozen (*F*) or thawing (*T*) soils during events. When the frost tube was not installed (typically from beginning of March to October or November), or not functioning, we used the 10 cm soil temperature data to determine the presence of frozen soil. We observed in the data that when maximum and minimum soil temperatures for a given day were both at 1.1 degrees Celsius or less, soils were typically frozen. Therefore, we applied that assumption when the frost tube data were absent.

3.3.2 Sediment Delivery Behavior

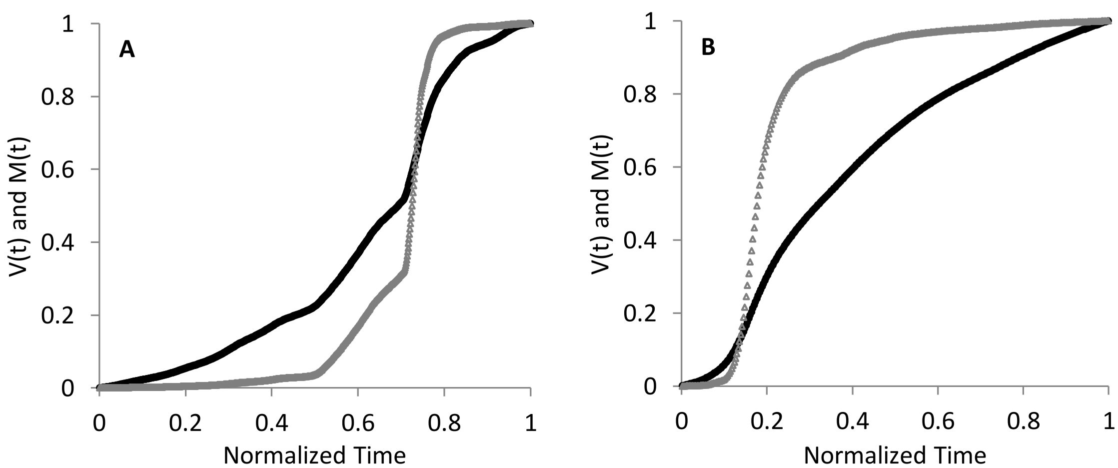
The “first flush” concept for sediment delivery, initially introduced by Helsel *et al.* (1979), can be used to identify when disproportionately high sediment delivery occurs during an event. This concept has most often been applied to urban systems (e.g. Cristina and Sansalone, 2003; Piro and Carbone, 2013) due to the characteristic “flushing” of impervious surfaces, but it was also applied to agricultural and forested systems (Helsel *et al.*, 1979). To determine when the sediment flush occurs, the normalized sediment mass, $M(t)$, and the normalized flow volume, $V(t)$, were calculated as:

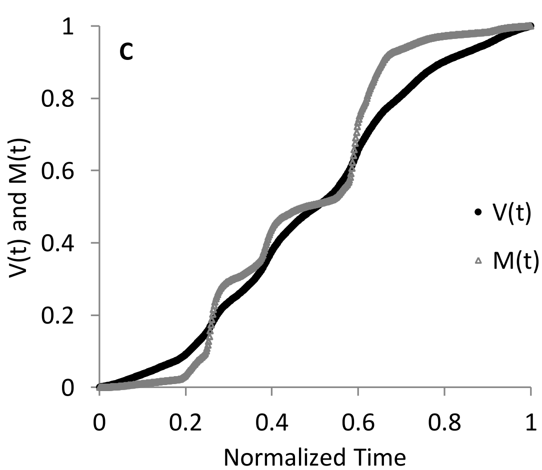
$$\mathrm{M(t)} = \frac{\sum_{i=0}^{k} \bar{Q}(t_i)\bar{C}(t_i)\Delta t}{\sum_{i=0}^{n} \bar{Q}(t_i)\bar{C}(t_i)\Delta t} \tag{3}$$

$$\mathrm{V(t)} = \frac{\sum_{i=0}^{k} \bar{Q}(t_i)\,\Delta t}{\sum_{i=0}^{n} \bar{Q}(t_i)\,\Delta t} \tag{4}$$

where $\overline{Q}(t)$ is the average flow rate and $\overline{C}(t)$ is the average sediment concentration during the time interval $k < n$ where $n =$ time at end of event (Helsel *et al.*, 1979; Cristina and Sansalone, 2003). When these curves are plotted together the sediment flush is at the point when $M(t) > V(t)$ (Fig. 1.3A-C).

Hydrologic systems can be classified as *transport limited* or *supply limited*. An erosion event is characterized as transport limited if there is sufficient sediment available, but the flow rate is the limiting factor (Piro and Carbone, 2013). An event is supply limited when there is not sufficient sediment available for transport, such as immediately following another event that flushed the system. To quantify when an event is transport or supply limited, the ratio of the area under the $M(t)$ and $V(t)$ curves is used (Cristina and Sansalone, 2003). When the area ratio of $M(t){:}\,V(t) > 1$, the event is transport limited. When $M(t){:}\,V(t) < 1$, the event is supply limited (Fig. 1.3A-C). We applied this concept to our event data set to determine if the Paradise Creek system is generally supply or transport limited.





Figures 1.3A-C Normalized sediment mass, M(t), and normalized flow volume, V(t), as a function of normalized time. When M(t) crosses V(t), a sediment flush occurs. **A.** Supply limited event because area ratio of M(t):V(t) < 1; **B.** Transport limited event because area ratio of M(t):V(t) > 1; **C.** No limiting factor because area ratio of M(t):V(t) = 1.

3.3.3 Statistical Analysis

We statistically analyzed the impact of multiple variables on event sediment load in PCW using multiple linear regression and ANOVA in the R statistical package (R development core team, 2008) for 328 events. The variables analyzed account for climate, watershed and antecedent conditions (Table 1.1).

Table 1.1 Variables found for each erosion event, including abbreviation of variable to be used throughout the text, measurement units, and description.

Variables	Abbreviation	Unit	Description
Total precipitation depth	P	mm	Depth of precipitation during and immediately leading up to event measured with rain gauge and snow adaptor
Snowfall SWE	S	mm	Depth of snowfall during and immediately leading up to event, multiplied by 0.15 for SWE
Snowmelt SWE	M	mm	Depth of snow that melted during event, multiplied by 0.3 for SWE
Available inflow	A	mm	Depth of water available to system considering precipitation and snow melt, calculated as $R_m = P - S + M$
Time to max rainfall intensity	t_p	mm	Time elapsed from start of rainfall event to maximum rainfall intensity
Maximum rainfall intensity	i	mm h^{-1}	Maximum rainfall intensity over 1 hour that occurred during event
Peak discharge	Q_{max}	m^3 s^{-1}	Peak stream discharge during event
Average discharge	Q_{ave}	m^3 s^{-1}	Average stream discharge during event
Event intensity	Q_{max}/ Q_{ave}	-	Relative intensity of event runoff
Streamflow depth	D	mm	Depth of streamflow calculated as total volume of water during event divided by catchment area
Event water volume	V	ha-m	Streamflow water volume during event calculated as average discharge multiplied event duration
Antecedent baseflow	B	m^3 s^{-1}	Stream discharge immediately before event start; used as surrogate for soil moisture
7 day antecedent precip.	P_7	mm	Total precipitation summed for seven days prior to event start; used as surrogate for soil moisture
Water year cumulative precip.	CP	mm	Total precipitation depth measured from Oct. 1 of water year to event start; used as surrogate for soil moisture
Time to runoff peak	t_r	min	Time elapsed from start of runoff event to peak stream discharge (length of rising limb)
Event runoff duration	t_d	min	Duration of runoff event from start to end
Relative rising limb length	t_r/t_d	-	Ratio of time to peak discharge to event duration; larger number indicates proportionally longer rising limb
Stormflow coefficient	SC	-	Stormflow coefficient indicates proportion of direct flow during event, calculated as $SC = D/A$
Snow cover	C	categorical	Presence of snow cover
Snowmelt event	SM	categorical	Presence of snowmelt
Frozen soils	F	categorical	Presence of frozen soils
Thawing soils	T	categorical	Presence of thawing soils
Limiting factor	lf	categorical	Transport or supply limited event
Land use	U	categorical	Event measured at rural or urban station

We log transformed the response variable (event sediment load) and removed variables from the model based on colinearity with another variable. Using stepwise regression with Akaike Information Criterion (AIC), we narrowed down the variables to those with statistical significance ($p < 0.05$) in explaining the response of event sediment load.

3.3.4 Land Use Contributions, Seasonality and Flush Timing

PCW is dominated by winter hydrology which results in significant storm events during late winter and early spring (December – April) when the catchment is saturated and precipitation generates greater levels of runoff. To determine the seasonal distribution of events we found average monthly sediment load and concentration and streamflow volume, and compared across sites. Additionally, we plotted the annual hydrograph with cumulative sediment load to show interaction of events.

We examined the difference in the timing of the sediment flush at each station to determine if the load measured at the urban station could be attributed to a flush of sediments from the rural area. For events that were observed at both stations, flush timing is the difference in timing of the sediment flush at each site compared to the estimated travel time of the sediment wave, which can suggest the origin of the sediment load. Average stream velocity is approximately equal to the travel time of the sediment wave (Williams, 1989). We used manual discharge measurements from 2000 – 2013 at the rural site to find stream velocity, which we calculated as discharge divided by stream cross-sectional area. We found average stream velocity for Paradise Creek's high runoff season (Jan – Mar, 0.51 m/s) and used it to estimate the travel time of the sediment wave. Stream distance between the two monitoring sites is 8850 m, therefore, the travel time of the sediment wave is, on average, 4.8 hours.

We found the timing of the sediment flush using "first flush" curves for each event. Those curves, because they are normalized by time, show at what percent through the event the sediment flush occurred, which we then used to find the actual time of the sediment flush during the event. We also compared the timing of peak sediment concentration between sites to confirm the timing of the sediment flush. We used both methods to increase confidence in determining when the sediment flush occurred at each site. When the sediment flush occurred first at the rural site and travel time of the sediment wave was less than or equal to the flush difference (i.e., the sediment wave from the rural area arrived at the urban station before a flush occurred there), the sediment measured at the watershed outlet likely came from the rural area. If travel time was greater than the flush difference, then the sediment measured at the watershed outlet cannot be attributed to the rural area. When the sediment flush occurred first at the watershed outlet, we considered the sediment load measured there to be from the urban area or the stream channel within the urban area. We then used the flush versus travel time analysis to re-approximate annual and monthly sediment loads from each land use and estimate how many urban events can potentially be attributed to the rural area.

3.3.5 Hysteretic Loops

The relationship between suspended sediment concentration and stream discharge is often manifested as a hysteretic loop which can indicate the dominant runoff process during an event and the proximity of the sediment source to the point of measurement (Williams, 1989; Seeger *et al.*, 2004; Lana-Renault *et al.*, 2009). Williams (1989) introduced this concept and observed five classes of relationships that can occur when sediment concentration is plotted versus stream discharge: 1) single-valued line; 2) clockwise loop; 3) counterclockwise loop; 4) single line plus a loop; 5) figure eight. A single-valued line indicates a continuous supply of sediment throughout the event (Williams, 1989). A clockwise loop shows depletion of sediment before the discharge peak, sediment sources limited to nearby areas (e.g. re-suspension of stream bed sediments), and saturation excess overland flow as the dominant runoff generating process (Williams *et al.*, 1989; Seeger *et al.*, 2004; Lana-Renault *et al.*, 2009; Eder *et al.*, 2013). A clockwise loop is often associated with the first flush of sediments (Eder *et al.*, 2013). A counterclockwise loop is also indicative of saturation excess overland flow, but with greater antecedent moisture conditions and sources spanning the entire catchment (Seeger *et al.*, 2004) because the travel time of the flood wave is shorter than the travel time of the sediment wave (Williams, 1989). A single-valued line with a loop is a combination of the previous classes. The figure eight shape occurs when water content is low and infiltration excess overland flow is the dominant process (Seeger *et al.*, 2004). We plotted hysteretic loops for the events in which we determined that the sediment at the watershed outlet potentially originated in the rural area (based on flush timing), providing another method to assess the probable origin of the sediment load.

4. Results

4.1 Sediment Delivery Behavior

Most events in Paradise Creek are transport limited (Fig. 1.4). Rural events are more transport limited than events at the watershed outlet, indicating a continual availability of sediment in the rural area as well as greater discharge at the watershed outlet. While we expected to observe that supply limited events occur mainly in the fall before the watershed is saturated or in the spring when crops have emerged, that did not occur. Furthermore, we did not notice any influence of the previous event on the subsequent one with respect to sediment delivery behavior. Specifically, we hypothesized that supply limited events would follow large events because the immediate availability of sediment would have been greatly reduced. However, that pattern was not apparent. This suggests that deposition may occur at the end of each event, thus replenishing stream bed sediments. In transport limited events, the sediment flush occurred early in the event, generally in the first 20%. In supply limited events, the sediment flush occurred closer to the middle of the event. In all events, the majority of the sediment load was transported during the flush period.

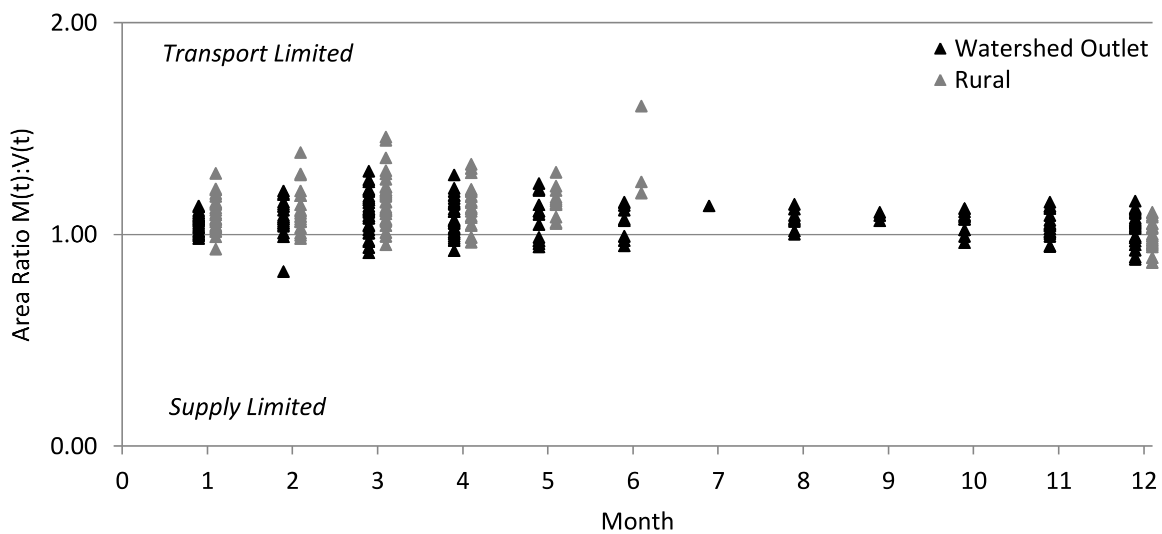


Figure 1.4 Area ratio of normalized sediment mass and flow volume curves, M(t):V(t), in PCW for watershed outlet and rural events by month. The *equal area line* is at 1.00; above that line, events are considered transport limited and below it, events are considered supply limited.

4.2 Characterization of Extreme Events

Each erosion event is unique. While a statistical analysis of the variables that explain sediment load for all events provides useful insight to processes and conditions that lead to high loads, the characteristics of individual extreme events point to specific climate and land use conditions that have the greatest impact on the sediment budget. In PCW, one event contributes, on average, 33% of the annual sediment load but only accounts for 2% of the time in a year (Fig. 1.5A-B). All events account for 86% of the average annual sediment load and 16% of the time in a year, leaving the remainder for the baseflow period. The five largest events in the watershed during the study period occurred on 17 – 27 February 2002, 10 – 16 March 2002, 28 January – 5 February 2003, 8 – 14 March 2003, and 20 March – 4 April 2012. These events each exceeded 285 t at both monitoring stations and were within the top six at each station. (28 January 2004 was the fifth largest for rural and 12

January 2011 the fifth largest for watershed outlet, but each one was not in the top six for the other station and so was not included.)

The event in March 2012 was the largest recorded event in the rural area, where it was much larger than at the watershed outlet (954 t compared to 666 t). According to the National Climatic Data Center Storm Events Database, heavy rainfall occurred on March 25 – 26 following a period of moderate to heavy snowfall (NCDC, 2014). This long duration rain-on-snow event resulted in the fourth highest crest level on record for Paradise Creek because large quantities of water from rain and snowmelt (201.9 mm; Table 1.2) were added to an already saturated watershed. This event exhibited two extremely high discharge peaks and sustained high flows between the peaks (Fig. 1.6E). The majority of the sediment load was transported during the hydrograph peaks. The March 2003 event was also a rain-on-snow event with saturated soil conditions that resulted in a greater sediment load in the rural area than at the watershed outlet (Fig. 1.6D).

The January 2003 event produced the greatest sediment load observed at the watershed outlet during the study period and was a result of heavy rain and snowmelt (NCDC, 2014). Similar to the March 2012 event, two extremely high discharge peaks occurred in January 2003 (Fig. 1.6C). Peak discharge at the watershed outlet was approximately equal to that of the March 2012 event (Table 1.2); rural peak discharge was lower than in March 2012 but still well above mean and median values for all events. For the five extreme events, peak discharge, event duration, and available inflow are greater than the mean and median event values (Table 1.2). Discharge throughout the event is always greater at the watershed outlet than the rural area, but for the two largest events (Jan. 2003 and March 2012) the difference was minimal. For some extreme events, antecedent baseflow and maximum rainfall intensity are greater than the mean and median event values, but not consistently across all extreme events. In 2002 and 2003, two extreme events were observed. In 2002, the second event was much smaller than the first. The same is true in 2003 for the watershed outlet; however, at the rural station, the two events are of similar size (531 and 544 t).

Cumulative water year precipitation for the five extreme events ranges from 228 – 397 mm (Table 1.2). Compared to all events during the study period, a similar window of cumulative precipitation (228 – 467 mm) is apparent for most large events (> 100 t) (Fig. 1.7). This is the period of time when the watershed is saturated enough to produce runoff and spring crops have not yet emerged.

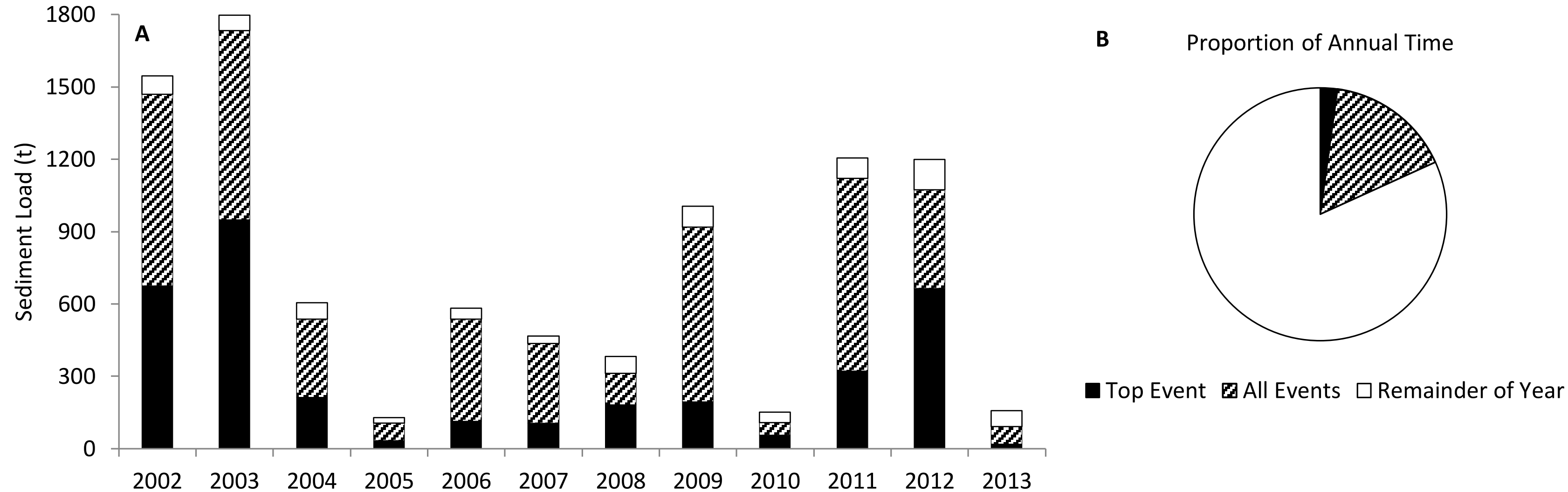
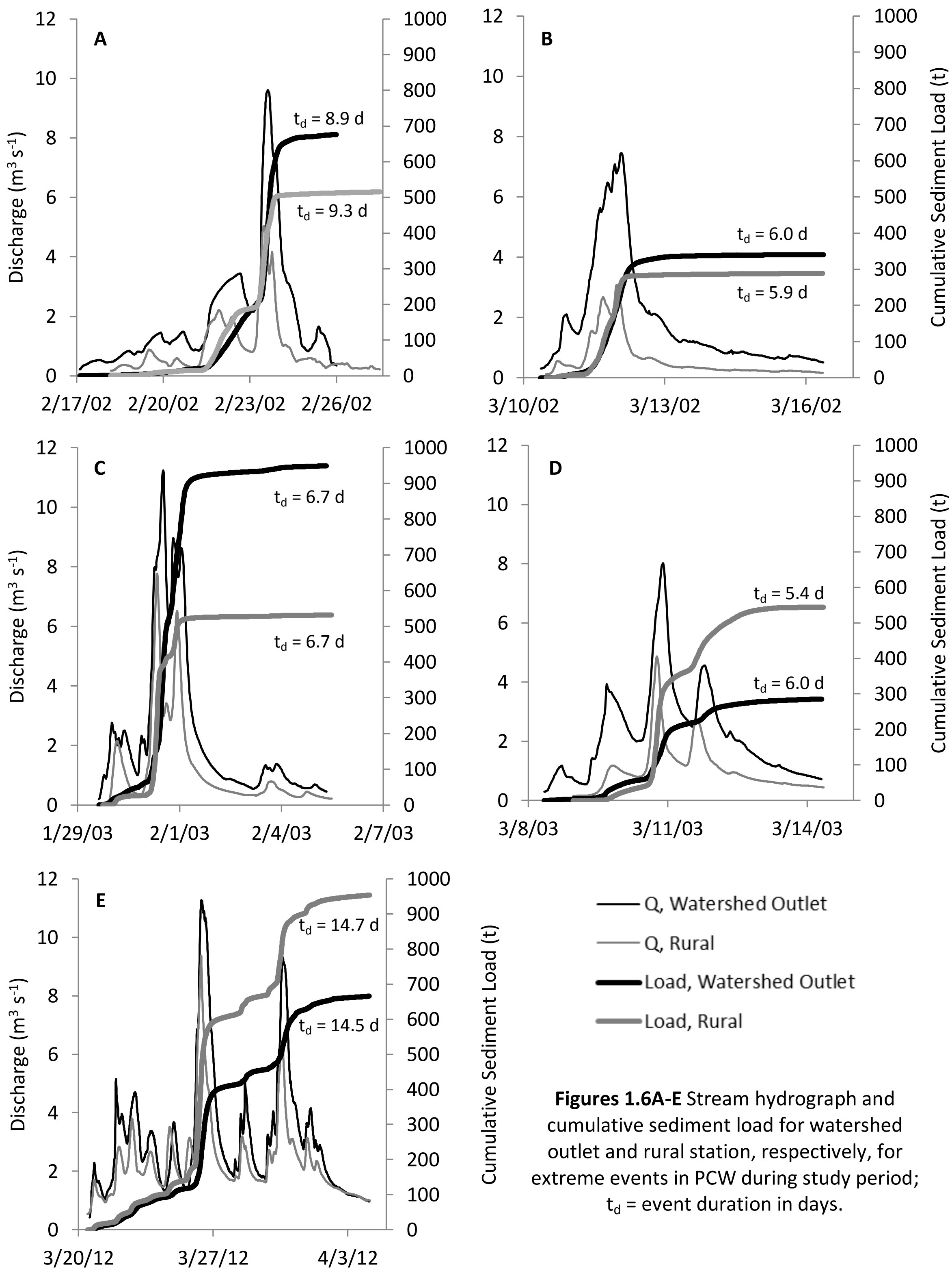


Figure 1.5A-B. A. Contribution of erosion events to annual sediment load in PCW as measured at watershed outlet for the top erosion event, all events, and the entire water year, respectively. **B.** Average proportion of time during the year represented by those events.

Table 1.2 Selected variables for extreme events in PCW during study period. Mean and median values are for all erosion events > 1 t. Observations are included for each monitoring station, rural (R) and watershed outlet (WO). Snow cover is during entire event (Yes), during rising limb (Rising), or not at all (No).

	Feb 2002		**Mar 2002**		**Jan 2003**		**Mar 2003**		**Mar 2012**		***Mean***		***Median***	
	R	**WO**	**R**	**WO**	**R**	**WO**	**R**	**WO**	**R**	**WO**	***R***	***WO***	***R***	***WO***
Sediment Load (t)	515	676	289	341	531	949	544	286	954	666	*43.7*	*44.2*	*7.0*	*9.2*
Peak Discharge ($m^3 s^{-1}$)	5.04	9.60	3.08	7.45	7.77	11.23	4.87	8.02	9.37	11.27	*1.53*	*2.70*	*1.01*	*2.00*
Event Duration (d)	9.3	8.9	5.9	6.0	6.7	6.7	5.4	6.0	14.7	14.5	*4.3*	*3.6*	*3.3*	*3.0*
Available Inflow (mm)	40.1	42.3	27.4	27.4	75.8	75.8	78.0	78.0	201.9	201.9	*32.5*	*31.6*	*21.3*	*21.8*
Ant. Baseflow ($m^3 s^{-1}$)	0.16	0.21	0.11	0.30	0.04	0.19	0.15	0.29	0.53	0.40	*0.19*	*0.20*	*0.13*	*0.10*
Max Precip. Intensity (mm h^{-1})	3.6	3.6	3.3	3.3	7.1	7.1	13.2	13.2	7.4	7.4	*4.5*	*4.9*	*3.8*	*4.1*
Cumulative Precip. (mm)	249.1	251.2	294.4	294.4	228.4	228.4	363.7	363.7	397.1	397.1	*396.1*	*356.3*	*384.7*	*363.7*
Snow Cover	Yes	Yes	No	No	Rising	Rising	Rising	Rising	Rising	Rising	-	-	-	-



Figures 1.6A-E Stream hydrograph and cumulative sediment load for watershed outlet and rural station, respectively, for extreme events in PCW during study period; t_d = event duration in days.

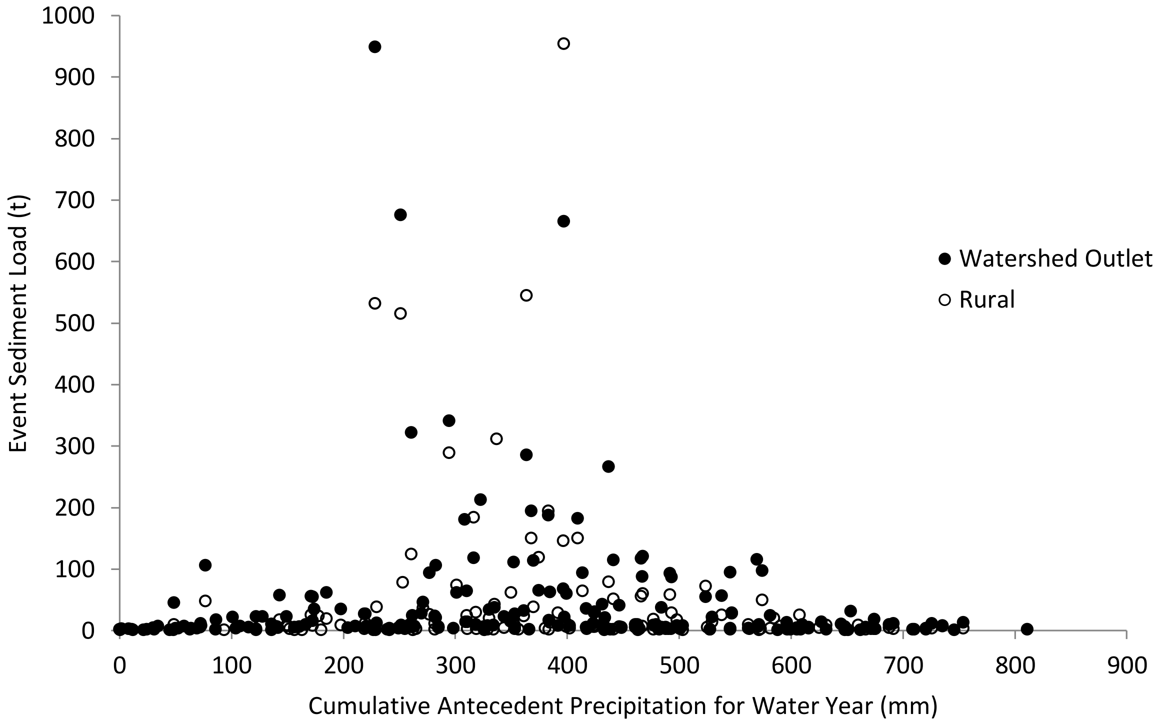


Figure 1.7 Event sediment load as a function of cumulative antecedent precipitation beginning on October 1st of given year in PCW. Most events of > 100 t occur in a window of 228 to 467 mm of cumulative antecedent precipitation.

4.3 Statistical Analysis

The results from the statistical analysis generally confirm the observations made for the extreme events in the watershed. Using multiple linear regression, peak discharge after log transformation explains a large proportion of the variation in event sediment load. Models were created for all events (watershed outlet and rural combined) (R^2 = 0.725), rural (R^2 = 0.775), and watershed outlet (R^2 = 0.767). Peak discharge explains 63.7%, 87.6% and 60.0% of event sediment load for all, rural, and watershed outlet events, respectively (Tables 1.3 – 1.5). The strength of the relationship between peak discharge and event sediment load can be seen in Figure 1.8. A linear relationship was examined in the multiple linear regression models, but on an individual basis, the relationship between these two variables is best explained with a polynomial function.

After peak discharge, event duration and antecedent baseflow are important factors. Duration explains approximately one quarter of the variation for all events and watershed outlet events whereas for rural events, it only explains 3.5% of variation. Similarly, antecedent baseflow explains 7.1% and 8.8% of variation for all and watershed outlet events, respectively, but only 3.3% for rural events. The final variables of importance for all events and rural events include maximum precipitation intensity and the presence of thawing soils, and the presence of thawing soils and frozen soils for urban events.

Other streamflow-related variables (e.g., event water volume and streamflow depth) also have strong positive correlations with event sediment load in single variable linear regression, but not as strong as peak discharge (Fig. 1.9). (See Appendix A for correlation matrices.) When considered in the multiple linear regression models, because of colinearity with peak discharge, the effects of water volume and streamflow depth manifest as negative correlations with load. For that reason, they were not used in the multiple linear regression models but should still be considered important factors. The effect of all other variables listed in Table 1.1 on event sediment load was examined using the stepwise function in R, but did not produce statistically significant relationships.

Table 1.3 Multiple linear regression performed in R for all events. n = 328; residual SE = 0.354 on 318 DF; adjusted R^2 = 0.725; F-statistic = 171.6 on 5 and 318 DF; p-value < 2.2E-16. Results are significant at $p < 0.05$.

Variable	DF	SS	F	P	Estimate	SE	Variation Explained
Q_{max}	1	31.96	254.50	< 2.20E-16	1.95E-01	1.22E-02	63.7%
t_d	1	13.01	103.58	< 2.20E-16	5.67E-05	5.57E-06	25.9%
B	1	3.57	28.41	1.87E-07	5.12E-01	9.61E-02	7.1%
T	1	1.14	9.04	0.00285	2.59E-01	8.60E-02	2.3%
i	1	0.54	4.29	0.0390	1.32E-02	6.38E-03	1.1%

Table 1.4 Multiple linear regression performed in R for rural events. n = 137; residual SE = 0.321 on 129 DF; adjusted R^2 = 0.775; F-statistic = 93.25 on 5 and 129 DF; p-value < 2.2E-16. Results are significant at $p < 0.05$.

Variable	DF	SS	F	P	Estimate	SE	Variation Explained
Q_{max}	1	20.67	200.90	< 2.20E-16	3.19E-01	2.25E-02	87.6%
t_d	1	0.82	7.95	0.00557	2.20E-05	7.81E-06	3.5%
B	1	0.77	7.52	0.00697	4.28E-01	1.56E-01	3.3%
i	1	0.71	6.88	0.00978	2.37E-02	9.05E-03	3.0%
T	1	0.63	6.15	0.0144	2.94E-01	1.18E-01	2.7%

Table 1.5 Multiple linear regression performed in R for watershed outlet events. n = 191; residual SE = 0.329 on 185 DF; adjusted R^2 = 0.767; F-statistic = 125.9 on 5 and 185 DF; p-value < 2.2E-16. Results are significant at $p < 0.05$.

Variable	DF	SS	F	P	Estimate	SE	Variation Explained
Q_{max}	1	17.26	159.96	< 2.20E-16	1.77E-01	1.40E-02	60.0%
t_d	1	7.73	71.62	7.82E-15	6.58E-05	7.77E-06	26.9%
B	1	2.52	23.35	2.82E-06	5.40E-01	1.12E-01	8.8%
F	1	0.74	6.82	0.00977	2.01E-01	7.68E-02	2.6%
T	1	0.51	4.73	0.0310	2.34E-01	1.08E-01	1.8%

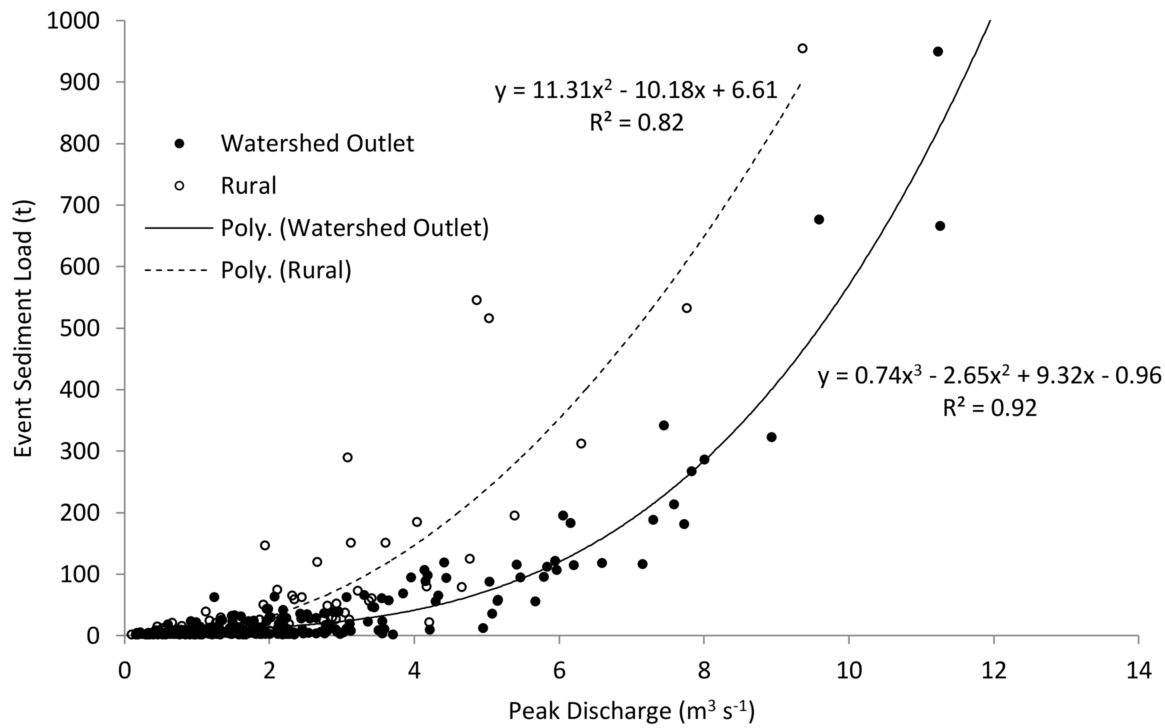


Figure 1.8 Event sediment load as a function of peak discharge for both watershed outlet and rural events in PCW. Polynomial relationship and R^2 values are shown for each site.

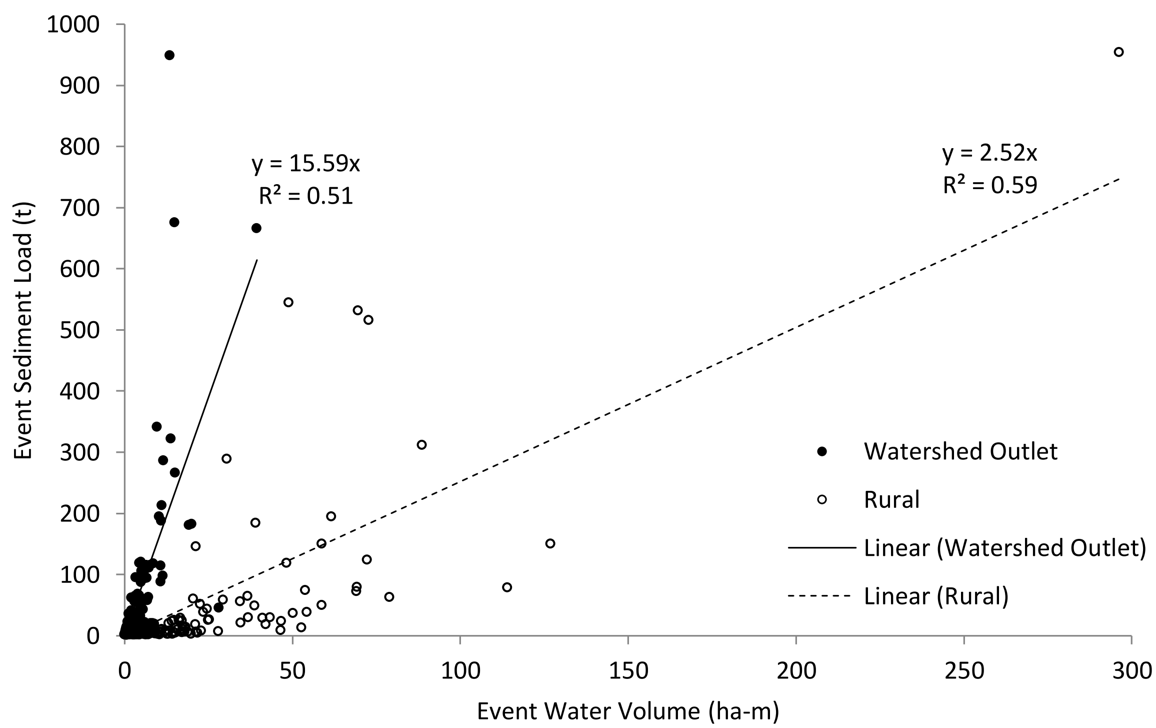


Figure 1.9 Event sediment load as a function of event water volume for both watershed outlet and rural events in PCW. Linear trendlines and R^2 values are shown for each site.

4.4 Land Use, Seasonality and Flush Timing

We observed an alternating influence from urban and rural land uses throughout the study period. The load measured at the watershed outlet was greater than the load measured in the rural area for all years except 2012 and 2013 (Fig. 1.10). For a rough estimation of the urban contribution to sediment load, we subtracted the load measured at the rural station from the load measured at the watershed outlet. This assumes that all sediment measured at the rural station traveled all the way through the urban area during the given water year. The urban load was greater than the rural load in 2005 – 2007 and 2010 – 2011. The rural load was greater than the urban load in the remaining years. In 2012 and 2013, the rural load was greater than the watershed outlet load, which indicates that deposition occurred in the stream channel in the urban area.

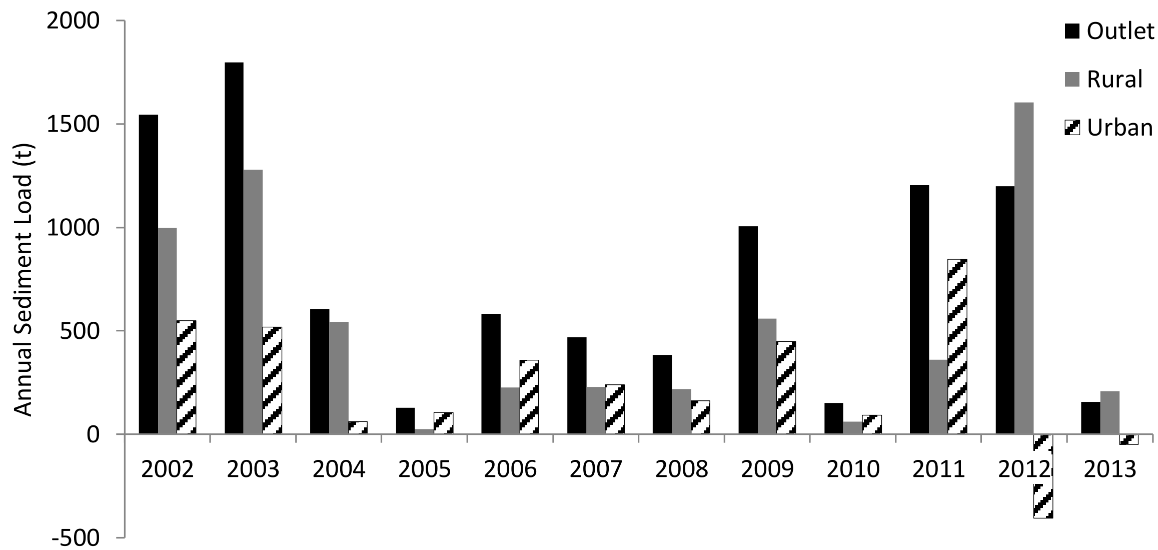


Figure 1.10 Annual sediment loads measured in PCW. Black bar is total load measured at the watershed outlet. Gray bar is total load measured at rural station. Diagonally hashed bar is load attributed to urban land use by subtracting rural load from watershed outlet load. Negative value indicates deposition.

Winter hydrology is apparent in PCW as the majority of the sediment load is delivered in December through April, with peaks in January and March (Fig. 1.11). A clear difference exists between the peak sediment load contributions from each land use. The urban area supplies the majority of its load during January, whereas the rural area peaks in March. Insignificant sediment loads are recorded during the summer at both sites. At the beginning of the water year, the urban load begins to accumulate before the rural load does; the urban load remains greater than the rural load until January. From January to April, the rural load is greater. When observed as average sediment concentration by month, the rural site peaks in March and the watershed outlet has sustained high concentrations from January – May (Fig. 1.12). Varying from the monthly sediment load distribution, however, the rural sediment concentration is only greater than the watershed outlet sediment concentration in March.

The largest annual load at the watershed outlet and the largest single event at that site (1/29/03 – 2/5/03) were recorded in 2003. Seasonal patterns of sediment loads at the two sites in 2003 are very similar, but with much greater loads measured at the outlet (Fig. 1.13A). The cumulative sediment load exhibits a two-step pattern; nearly six weeks separate those two significant events. The second large rural event (early March) was approximately the same magnitude as the first (late January) despite a substantially lower peak discharge. Conversely, in the remainder of March, several moderately-sized flows separated by a few days did not result in as large of an increase in sediment load. For 2012, the largest rural annual load and single event (3/20/12 – 4/4/12), the cumulative load exhibits a more continuous increase from January through March and a clear dominance from the rural area in March (Fig. 1.13B). In agreement with Figures 1.11 and 1.12, the rural contribution is dominant in March in both 2003 and 2012 (Fig. 1.13A-B).

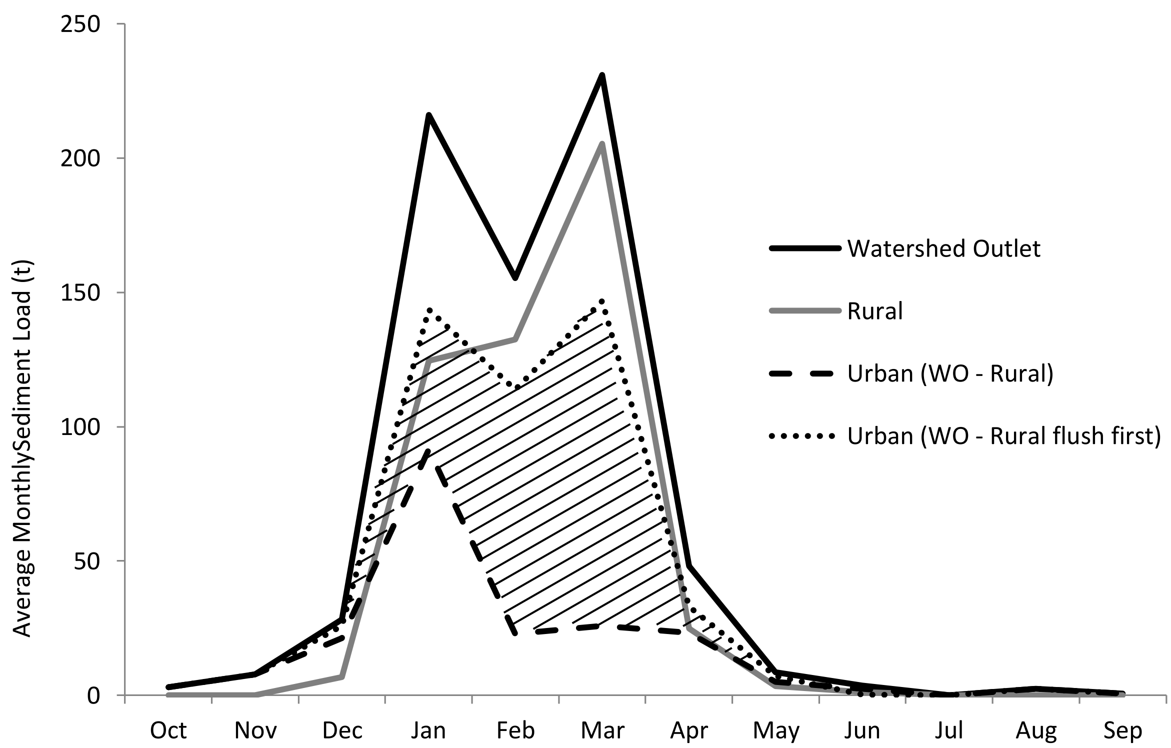


Figure 1.11 Average monthly sediment loads in PCW from 2002-2013. Black line indicates the average monthly load measured at watershed outlet. Gray line indicates the average monthly load measured at the rural station. Dashed line indicates the average monthly urban load when subtracting rural load from the load measured at watershed outlet. Dotted line indicates average monthly urban load when subtracting rural load only when indicated by flush timing. The area between the dashed and dotted lines represents the estimated range for the urban load.

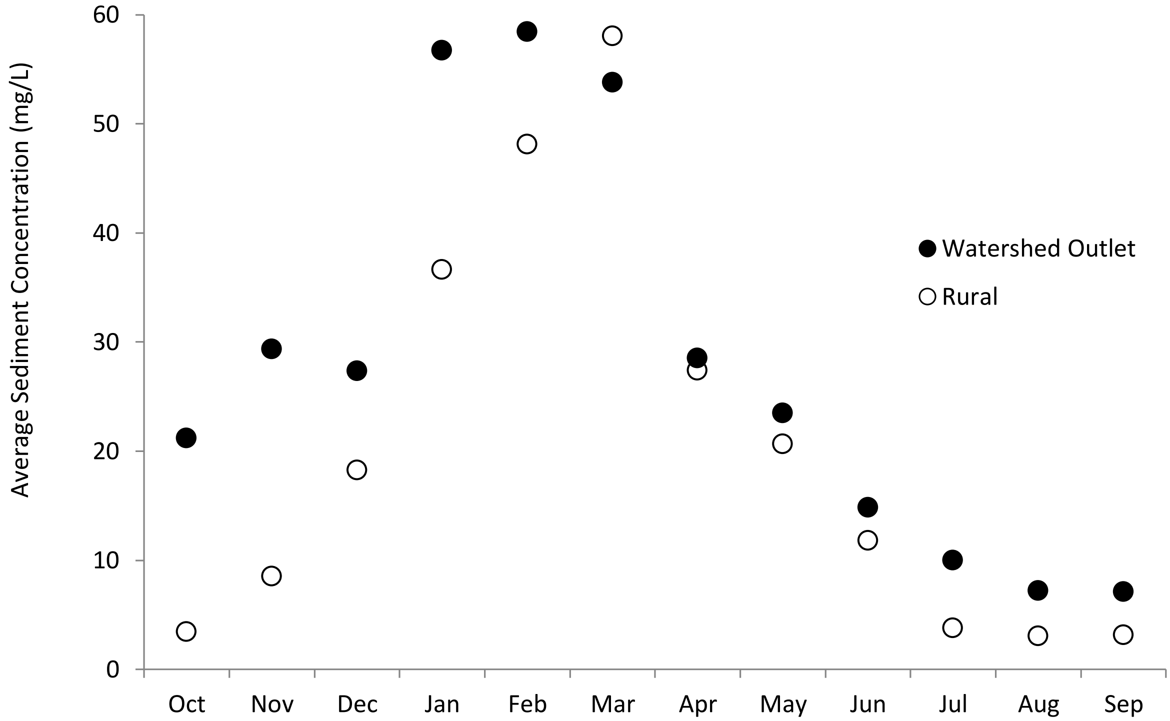
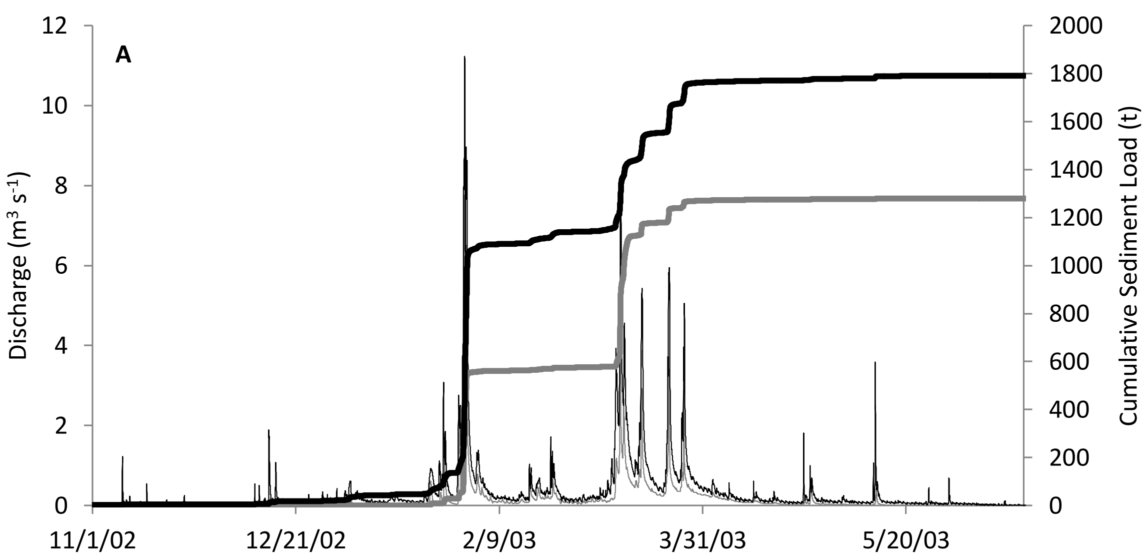
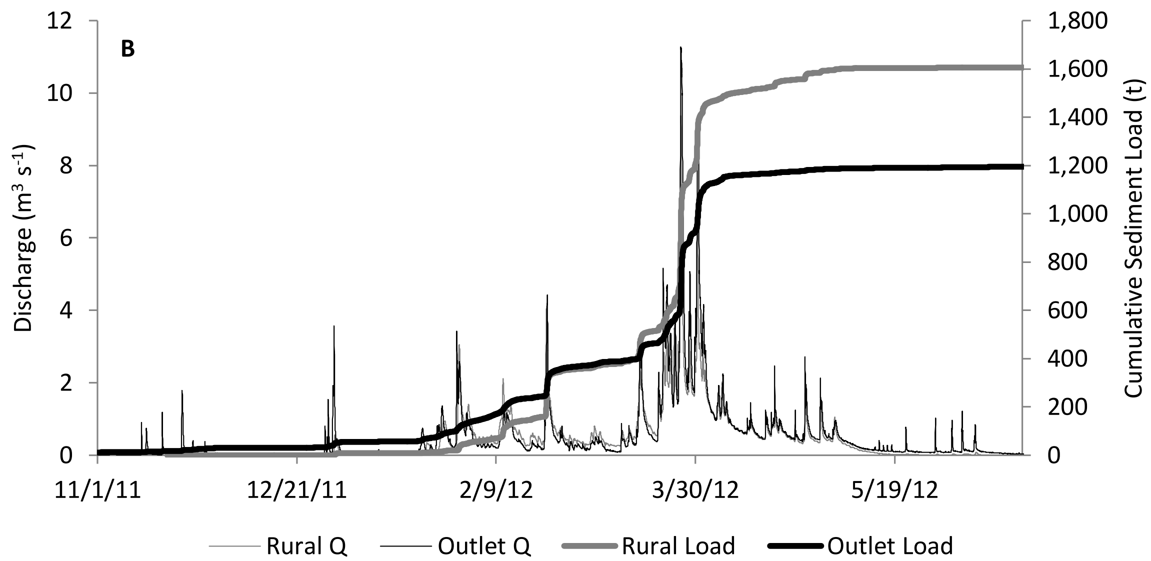


Figure 1.12 Average 15-minute sediment concentration by month in PCW from 2002-2013 at the watershed outlet and rural sites. Sediment concentration is higher at the watershed outlet in all months except March when it peaks in the rural area.





Figures 1.13A-B Cumulative sediment load and hydrograph for PCW, rural and watershed outlet for **A.** 2003, the largest annual sediment load at the watershed outlet; and **B.** 2012, the largest annual rural sediment load.

For 53 of 209 events that occurred at both sites, the sediment wave from the rural site could have reached the watershed outlet before its sediment flush and concentration peaks occurred. When this analysis is applied to annual loads by subtracting only these events from the total load measured at the watershed outlet, the large amount of sediment deposition in the urban stream channel for 2012 – 13 is not present and the urban contribution is greater than in the original method (Fig. 1.14). Additionally, consideration of flush timing in the analysis of average monthly sediment loads results in greater urban than rural loads in January, and a more significant influence of urban loads in March compared with the original method (Fig. 1.11).

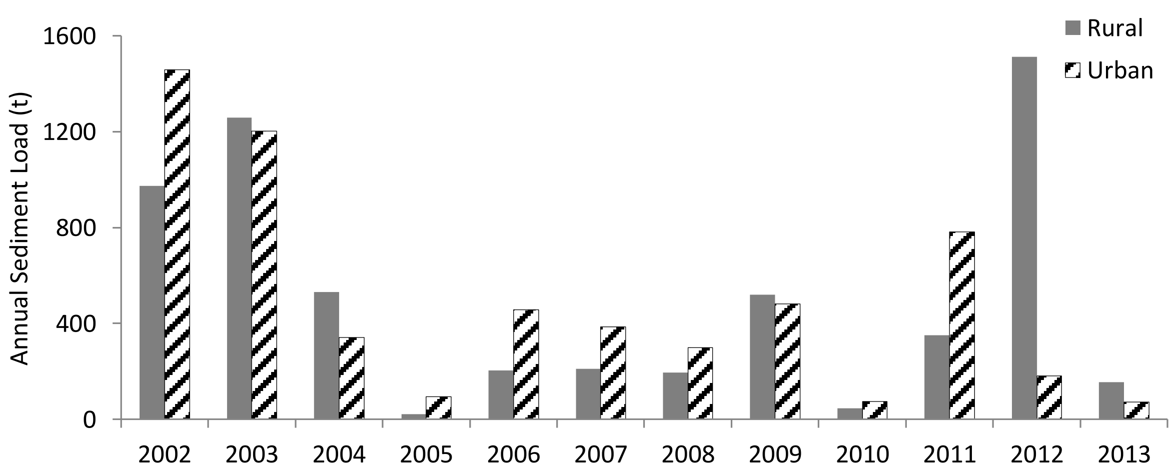


Figure 1.14 Annual sediment loads in PCW by land use considering flush timing. Urban load was calculated as watershed outlet minus rural when rural flushes first and travel time of sediment wave is greater than the flush difference.

As a method to confirm the potential source of sediment at the watershed outlet for the 53 events from the flush timing analysis, we plotted the hysteretic loops for each since the direction of the loop can indicate the source of the sediment. Five were clockwise loops which rules out a rural contribution at the watershed outlet because it indicates a nearby sediment source. The remaining exhibited counterclockwise loops (20), figure eights (25), and straight lines (3). A counterclockwise loop is expected if we are attributing the sediment load at the watershed outlet to the rural area since it indicates a distant sediment source. A figure eight hysteretic loop suggests a short duration event, low water content, infiltration excess overland flow, and rapid extension of contributing areas, potentially in the rural area. A straight line (equal on rising and falling limb) indicates that the sediment contribution during the event was continuous, and thus could come from both locations.

5. Discussion

Like many agricultural areas, Paradise Creek is generally a transport limited system in the rural portion with a continual supply of sediment from fields and the stream channel (Brooks *et al.*, 2010; Wagenbrenner *et al.*, 2010). When transport capacity and stream power increase due to high peak discharge and sustained elevated flows, the available sediment is readily eroded from the upland areas and stream banks or re-suspended in the stream channel. Flow volume positively correlates with sediment load, but not as strongly as peak flow, indicating that the extra force exerted by high peak flows is important in detaching or re-suspending sediment, and generating and carrying greater loads. High peak discharge also leads to increased sediment concentrations because of the likelihood of expanding contributing areas, as seen with the variable source area concept (Walling and Webb, 1982). In such events, the contributing area expands to locations that are not reached in smaller magnitude events and thus have stores of easily erodible sediment. The interaction and related recovery period of events also dictates the resulting sediment loads. An event with high peak discharge may not generate an equivalently large sediment load if it is preceded by another large event with a short recovery period between the two (Walling and Webb, 1982). We see this in March 2003 with a series of four high peak discharge events with recovery periods of less than seven days (Fig. 1.13A). The first event in the series produced a much greater load than the subsequent events. Furthermore, that first March event generated the same sediment load as a January 2003 event that had a much higher peak discharge. This is likely a result of the 38 day recovery period that occurred after the January event when stream runoff was relatively low.

In PCW, when high antecedent soil moisture conditions are paired with high peak flows, event sediment loads are greater (Tables 1.3 – 1.5). Through both quantitative and qualitative data, we observed saturated soils in mid to late winter which aligns with the winter hydrology of the region. Events in the fall do not generate as much runoff or erosion because most precipitation infiltrates into the relatively dry soil, rather than contributing to streamflow response (Lana-Renault *et al.*, 2007). Both antecedent baseflow and antecedent precipitation were statistically analyzed as indicators of soil moisture conditions, as many previous studies have used either or both (e.g., Merz *et al.*, 2006; García-Ruiz *et al.*, 2008; Lana-Renault and Regues, 2009; Danz *et al.*, 2013). For PCW, we found that antecedent baseflow was statistically significant in predicting sediment load while antecedent precipitation was not. In a watershed dominated by saturation excess processes, a logical connection can be made between elevated baseflow levels and saturated soils ready to generate surface runoff. In areas with saturation excess, antecedent conditions have been related to an expansion of contributing areas (Zehe and Blöschl, 2004; García-Ruiz *et al.*, 2005). Furthermore, when soils are saturated, or "primed", it is expected that more erosion will occur because subsurface soil cohesion is reduced as water content increases (Kemper and Rosenau, 1984; Pelletier, 2012). With reduced soil cohesion, or increased soil erodibility, detachment potential increases with increased shear velocity causing rill and gully erosion (Govers, 1985).

Precipitation depth during erosion events was not a significant predictor of event sediment load in PCW, which is consistent with other studies conducted in areas with saturation excess overland flow. González-Hidalgo *et al.* (2009) found that daily and mean precipitation were not good predictors of soil erosion. Seeger *et al.* (2004) suggested that precipitation had less influence on runoff generation than antecedent moisture conditions for both saturation excess and infiltration excess overland flow. However, some studies indicated that combining precipitation depth and antecedent moisture conditions better predicted runoff response. For example, Rodríguez-Blanco *et al.* (2012), in a study in a temperate humid catchment with variable runoff processes, found that soil water content was more important than precipitation in runoff response, but that pairing the two variables explained more variation in runoff coefficients than soil moisture alone. Merz *et al.* (2006) also showed that coupling the two variables influenced watershed response under mixed runoff generation processes. To explain the variation in sediment loads during runoff events, Danz *et al.* (2013) found in several Wisconsin watersheds with low to moderate infiltration rates and moderate to high runoff rates, that both rainfall depth and antecedent baseflow were influential during rainfall events, while predictors of soil moisture or snowmelt were most important during snowmelt events.

The three largest events in the watershed had relatively high maximum precipitation intensities: 7.4, 7.1 and 13.2 mm h^{-1}, compared to mean and median values during the study period of less than 5 mm h^{-1} (Table 1.2). Maximum precipitation intensity was statistically significant at explaining a small amount of the variation in the rural event sediment loads (Table 1.4). Nadal-Romero *et al.* (2008) found a significant correlation between maximum precipitation intensity and peak discharge. Through experiments in an area of low intensity rainfall and permeable soils, Dunne and Black (1970) showed that only when soils were fully saturated, increased precipitation intensity generated more runoff through an increased proportion of overland flow.

We observed that the majority of annual sediment loads in Paradise Creek are measured in January – March (Fig. 1.11), coinciding with the period of high peak runoff observed by Brooks *et al.* (2010). While agricultural BMPs have produced significant results in recent decades, when winter wheat farmers follow conventional practices, soil is left bare during the vulnerable winter period, increasing erosion potential (Kok *et al.*, 2009). Interestingly, Brooks *et al.* (2010) showed that in the rural area of PCW, average annual runoff volume peaks in February, whereas we found that average annual sediment load peaks in March. This disparity between the timing of runoff and sediment loads is likely due to high antecedent moisture conditions and rain-on-snow events in March which decreases surface storage. We expect that soil saturation peaks in March after a winter of snowfall, thereby increasing erosion. Additionally, rain-on-snow events that are characteristic of March increase direct runoff (Kattelmann 1997; Marks *et al.*, 1998; Merz *et al.*, 2006) and, therefore, peak flows which lead to large sediment loads, as seen with the two largest events in PCW during the study period. The urban contribution was greatest in January, despite peak runoff occurring in March, which may be more characteristic of a supply limited system. Elevated January loads could be a result of channel flushing of deposited sediment from the previous year. By January, flows are high enough to re-suspend channel sediments and flush the system, reducing sediment available in the stream channel during the remainder of the winter that would be measured as contributing to the urban load. Elevated loads in January could also be partially attributed to the material laid out on city roads to improve traction for driving on snow and ice. The City of Moscow currently uses approximately 125 tons of clean chip rock annually, the majority applied in December and January, with an estimated 75% recovery rate (T. Palmer, personal communication, 27 March 2014).

The stream channel appears to play a large role in the Paradise Creek sediment budget (Brooks *et al.*, 2010; Rittenburg, 2014). Many of the events in Paradise Creek exhibited an early flush as characterized by “first flush” curves and hysteretic loops. Such behavior is typically due to a flush of channel sediments (Eder *et al.*, 2013). The availability of those sediments is dependent upon the deposition characteristics of the previous event while the re-suspension of them is dependent upon the stream power of the current event. For example, an event with a relatively low peak flow may be more likely to deposit sediments rather than flush them through the system. Whereas an event with a relatively high peak flow may flush more sediment out of the system because of increased stream power. However, since stream power is a function of both discharge and stream gradient, the low stream gradient in the rural (0.5%) and urban (0.01%) sections likely leads to deposition after many of the events. Subsequently, re-suspension of streambed sediments occurs after those events. Erosion of the stream channel itself may also be a factor, particularly in the urban area. As downstream urban sediment sources increase with development and upstream rural sources decrease with improved BMPs, channel erosion in the urban area may increase (Wolman, 1967) which is potentially attributable to increased magnitude and number of peak flows associated with urban runoff.

In the Paradise Creek TMDL (IDEQ, 1997), only 5% of the sediment load was attributed to the urban area. Brooks *et al.* (2010) showed that an average of 43% of annual loads in Paradise Creek comes from the urban area. We found that the urban and rural land uses alternate in their influence on the sediment budget in Paradise Creek, which is likely due to annual differences in activities and watershed conditions. For example, a short-term construction site could significantly increase the urban load for a given year and even exceed agricultural yields (Wolman, 1967); alternatively, an increased proportion of fallow fields (Zuzel *et al.*, 1993) or increased conventional tillage (Gaynor and Findlay, 1995) could result in a larger rural load. Recent years were the most interesting with high annual loads in 2011 and 2012 after several years of loads that met or were close to the TMDL target (Fig. 1.2). The urban influence was much greater in 2011, followed by a much greater rural influence in 2012 (Fig. 1.10). In 2011, Paradise Creek restoration projects were implemented in the urban area, including re-routing the stream channel on the University of Idaho campus. That activity likely led to the high urban load. The high rural load in 2012 was likely due to high antecedent moisture conditions resulting from the previous year's fallow fields. In 2011, half of Latah County was not planted due to an extremely wet spring. The absence of crops reduced the influence of transpiration on the water balance and thus soils remained wetter through the summer and fall (Sivapalan *et al.*, 2005) leading to wet antecedent soil conditions going into the winter of 2011 – 12. We suspect that the wetter than normal conditions in the watershed led to high erosion in the rural area in 2012. Additionally, in 2012, the sediment load recorded at the rural station was much greater than the sediment load recorded at the watershed outlet, indicating deposition in the stream channel. We would expect that in 2013, the stream channel deposits from 2012 would have resulted in an increased urban load. However, 2013 was a very low water year (Fig. 1.2), so stream power and transport capacity may have been too low to detach and carry the deposited sediment to the outlet.

6. Conclusions

This study, conducted in a small, mixed land use watershed dominated by winter hydrology and saturation excess overland flow, sought to characterize extreme erosion events based on (1) sediment delivery behavior; (2) climate, watershed and antecedent conditions; and (3) land use and seasonality. The implications of our findings can be extended to similar watersheds that have the rural area upstream of the urban area. We found that, due to the transport-limited nature of the watershed, peak discharge was the driving force behind sediment loads, with event duration, antecedent conditions, and precipitation intensity also explaining some of the variability. The influence of the rural and urban land uses alternated throughout the study period, depending partly on antecedent conditions and activities in the watershed. The urban contribution to Paradise Creek annual sediment loads peaked in January, while the rural contribution peaked in March, coinciding with soil saturation in the watershed. We suspect a cycle of deposition and re-suspension of stream bed sediments occurs in Paradise Creek, but more research is needed to determine the magnitude of the stream channel contribution to annual loads.

Annual sediment loads in Paradise Creek, and the Northwest Wheat and Range Region as a whole, are declining due to improved management practices (Kok *et al.*, 2009; Brooks *et al.*; 2010). Correspondingly, we observed that most extreme events in PCW occurred early in the study period (2002 and 2003). Despite the declining trend in event sediment loads, the single largest event (largest rural event, third largest urban event) occurred late in the study period (March 2012). That extreme event was a result of a perfect storm of influential factors impacting the watershed: peak discharge, event duration, antecedent baseflow, available inflow (rain + melt), and precipitation intensity were all well above mean and median values. It was also a rain-on-snow event and occurred in 2012 when the watershed itself was highly erodible due to wet antecedent moisture conditions.

Our findings suggest that in order to further reduce sediment loads, best management practices (BMPs) designed to target the factors characteristic of the largest events may be effective at further reducing annual sediment loads. Specifically, BMPs that increase infiltration in upland areas, such as conservation or no tillage, residue cover, and cover crops, could dampen the response of peak discharge while also reducing the impact of high intensity precipitation. When possible, not leaving fields fallow or planting cover crops to increase transpiration may reduce antecedent moisture conditions.

Currently, TMDL targets for sediment are focused on annual loads. Because of the seasonality of erosion, re-focusing load targets on monthly or seasonal periods could better elucidate the sediment delivery behavior of a stream system and provide insight to when loads can be reduced. In watersheds with similar climate patterns to Paradise Creek, late winter and early spring should be specifically targeted for reducing sediment loads. With climate change predictions of increasing precipitation in the Pacific Northwest, saturated soils may become more prevalent, which could reverse the declining trend in sediment loads. Future research should investigate the timing of planting and crop rotations to target the periods most vulnerable to erosion and the effectiveness of BMPs at reducing the impact of extreme events. To better understand the role of extreme events in other watersheds, study designs should include event-based monitoring to be able to determine if peak flows play a strong role in generating sediment loads. Additionally, within the Paradise Creek data set, a time series analysis could show how events are impacted by the magnitude of the previous event and the recovery period between events.

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Chapter 2: BMP Effectiveness and Dominant Hydrological Flow Paths: Concepts and a Review

1. Abstract

We present a conceptual framework that relates BMP effectiveness with dominant hydrological flow paths. We use the framework to analyze plot, field and watershed scale published studies on BMP effectiveness to develop transferable recommendations for BMP selection and placement at the watershed scale. The framework is based on the location of the restrictive layer in the soil profile and distinguishes three hydrologic land types. Hydrologic land type *A* has the restrictive layer at the surface and BMPs that increase infiltration are effective. In land type *B1*, the surface soil has an infiltration rate greater than the prevailing precipitation intensity, but there is a shallow restrictive layer causing lateral flow and saturation excess overland flow. Few structural practices are effective for these land types, but pollutant source management plans can significantly reduce pollutant loading. Hydrologic land type *B2* has deep, well-draining soils without restrictive layers that transport pollutants to ground water via percolation. Practices that increased pollutant residence time in the mixing layer or increased plant water uptake were found as the most effective BMPs in *B2* land types. Matching BMPs to the appropriate land type allows for better targeting of hydrologically sensitive areas within a watershed, and more significant reductions of NPS pollutant loading.

KEY TERMS: best management practices (BMPs), nonpoint source (NPS) pollution, watershed management, agriculture

2. Introduction

As large-scale industrial agriculture became prominent in the 1960s in the American landscape, greater use of chemicals and intensive soil management proved essential to increase food and fiber production. A consequence of this intensive agriculture continues to be nonpoint source (NPS) pollution, which impairs water bodies from addition of sediment, nutrients, or pesticides. In recognition of this threat to aquatic ecosystems, agricultural conservation practices have been developed to manage pollutant sources and to prevent the transport and delivery of pollutants from both overland and subsurface flow to water bodies (Mulla *et al.*, 2005). The Natural Resource Conservation Service (NRCS) has developed many pollution prevention techniques and quantitative standards of best management practices (BMPs) that can be found in the NRCS Field Office Technical Guide (NRCS, 2011).

Despite widespread implementation, the success of BMPs to control nonpoint source pollution has been mixed and the reported effectiveness of a single BMP can vary greatly. For example, Liu *et al.* (2008) showed sediment trapping efficacy from over 80 buffer studies varied from 54 to 100%. Often times these variable BMP efficiencies are a result of poor BMP placement with respect to critical pollutant source areas, timing or flow pathways (Tomer and Locke, 2011), or a poor match of the BMP to the physical characteristics of the landscape. Most published BMP studies have not clearly identified the physical characteristics (e.g., soil type, topography, land cover and treatment, and climate) responsible for BMP effectiveness. In the studies that did, authors found BMP effectiveness to be dependent on local hydrologic and climatic conditions (e.g., Walter *et al.*, 2001; Ghidey *et al.*, 2005; Deasy *et al.*, 2007).

Therefore, achieving BMP effectiveness requires knowledge of the local climate and site characteristics that trigger activation of the hydrologic flow paths (i.e., overland flow, subsurface lateral flow, and percolation and saturated flow in ground water). Various groups have suggested "targeting" BMPs for optimal placement and timing for maximum pollution reduction by understanding pollutant transport via hydrologic flow paths (Veith *et al.*, 2004; Mulla *et al.*, 2005; Galzki *et al.*, 2008). Walter *et al.* (2000) suggested limiting potentially polluting activities in hydrologically sensitive areas (HSAs), which occur when runoff is produced in so-called hydrologically active areas (HAAs). In their case, HSAs linked saturated areas to surface water bodies via overland flow. In this paper, we extend the definition of HAAs to include infiltration, perched water tables, and subsurface flow.

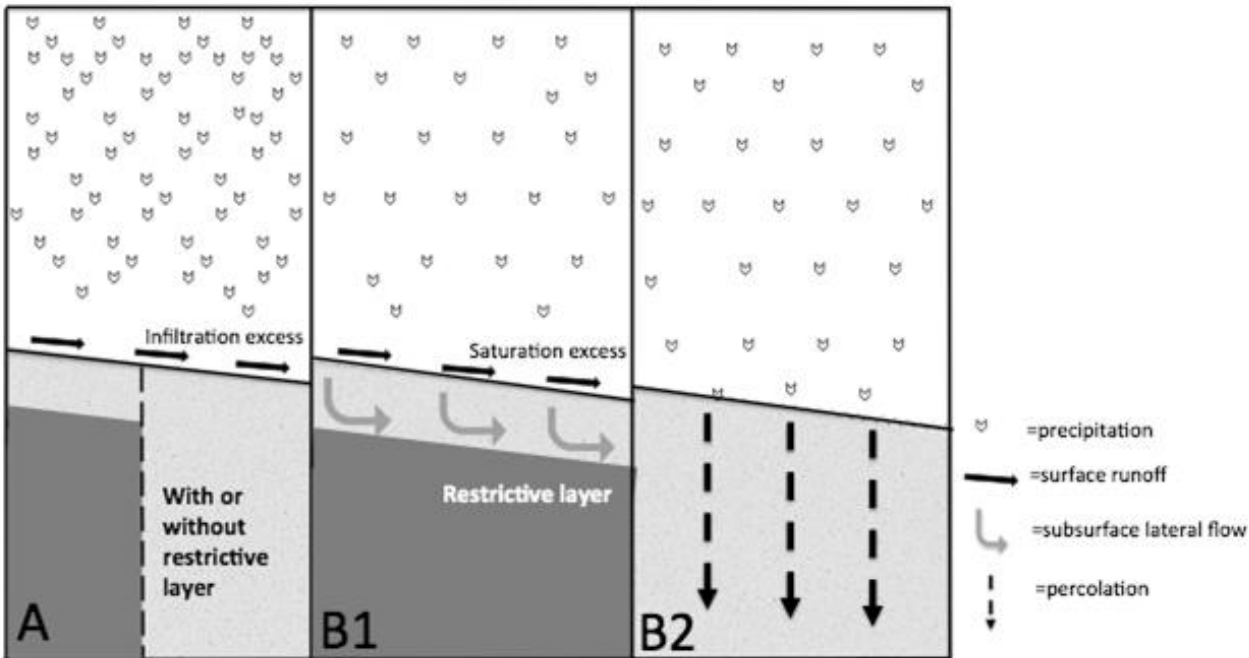
Studies that linked flow paths to the effectiveness of BMPs began with Stewart *et al.* (1975). Stewart *et al.* (1975) developed charts linking climate, topography, soils, and farming practices with effective management practices. Walter *et al.* (1979) extended this further and noted how soil and water conservation practices affect the flow path. More recently, Mulla *et al.* (2005) identified a significant opportunity to optimize BMP type and placement within a landscape through targeting. In addition, Pannell *et al.* (2006) and Walter *et al.* (2007) indicated that targeting BMPs in HSAs appears more cost effective and can reduce pollutant loading more than indiscriminately implementing BMPs across a watershed because the BMP is matched to the local hydrologic and climatic conditions. Gburek *et al.* (2000) identified similar potentials for reducing phosphorus loading if critical source areas are targeted.

Despite efforts over the past four decades, there is still not a clear consensus of the actual effects of BMPs at the watershed scale. Tomer and Locke (2011) reviewed Agricultural Research Service (ARS) benchmark watersheds and corroborated findings by numerous other studies (Dillaha, 1990; Tim *et al.*, 1995; Mulla *et al.*, 2005) and by researchers in the USDA Conservation Effects Assessment Project (CEAP) watersheds. As noted by several studies, many BMP assessments are designed in such a way where the effectiveness of a practice cannot be easily quantified due to, among other factors, the inherent physical lag time between implementation and water quality improvements (Gregory *et al.*, 2007; Meals *et al.*, 2010). Other studies find that some BMPs are simply not effective due to structural failure or incorrect implementation of the management practice. However, in this work we will focus on the need for a better method of targeting BMPs within a watershed by both identifying HSAs and matching BMP effectiveness with dominant flow paths. A number of BMP review papers have been published, however most do not relate BMP effectiveness to HSAs, and therefore do not provide translatable recommendations for selecting BMPs for a given HSA based on the pollutant flow path.

Therefore, our objective is to relate BMP effectiveness to the hydrological characteristics of a particular landscape location in order to minimize pollutant loading at the watershed scale. To do this, we (1) develop a conceptual framework that relates BMP effectiveness with dominant hydrological flow paths that are a consequence of land and climate characteristics; (2) determine how BMP effectiveness reported in past plot and field scale studies fit within the framework; and (3) review BMP effectiveness in watershed scale studies and how cumulative effects of BMPs at this scale can be explained within the conceptual framework.

3. Conceptual Framework for Targeting BMPs

In this framework, we first determine the soil and climate characteristics in landscapes that drive dominant hydrologic flow paths. From there, we define three hydrologic land types that are characterized by these specific flow paths. Next, we simplify the classification of major pollutant types, and relate pollutant transport to probable flow paths and land types. The conceptual framework of hydrological land types, and their respective dominant flow paths, probable pollutant transport, and BMP recommendations can be found in Figure 2.1.



	Precipitation intensity>infiltration capacity	Precipitation intensity<infiltration capacity	Precipitation intensity<infiltration capacity
Soil depth	Shallow or Deep	Shallow	Deep
Dominant Flow Paths	**Infiltration excess**, regardless of presence of restrictive horizon or slope	**Lateral flow** and/or **saturation excess; tile drainage**	**Percolation** and/or **tile drainage**
Probable Pollutant Transport	Pesticides, phosphorus, soil loss	Pesticides, nitrates, phosphates, soil loss	Pesticides, nitrates, phosphates
Effective BMPs	Mulch tillage, no tillage, terraces, CRP, effective buffers, cover crops, NMPs, IPMs	Mulch tillage, no tillage, CRP, cover crops, tile drainage, NMPs, IPMs	Cover crops, controlled drainage, crop rotations, forested buffers, NMPs, IPMs

Figure 2.1 Land type conceptual framework (abbreviations: NMP: Nutrient Management Plan; IPM: Integrated Pest Management plan; CRP: Conservation Reserve Program).

3.1 Hydrologic Flow Paths and Land Types

We argue that BMP effectiveness is greatest when placement is fundamentally based on the dominant hydrologic flow path that is associated with the pollutant of concern within a landscape. The hydrology in our conceptual framework is based on how water flows in the landscape. All excess rain (i.e., precipitation that does not evaporate or transpire) eventually flows laterally towards a surface water body or percolates vertically to ground water. The lateral path that the excess rain follows depends very much on soil characteristics such as permeability and where the restrictive layer occurs in the landscape (Figure 2.1).

The restrictive layer is at the soil surface when the precipitation intensity is greater than the infiltration rate of the soil, and surface runoff occurs (sometimes called Hortonian flow). This can occur for soils that have low organic matter, surface crust formation, fine texture, are degraded with decreased macroporosity (Horton, 1933), or are frozen. Hortonian overland flow usually occurs during storms and shortly thereafter. In our conceptual framework we identify areas in a landscape where the precipitation intensity is greater than the infiltration capacity due to a surface restrictive layer as Hydrologic Land Type *A* (Figure 2.1).

When the infiltration rate at the surface is greater than the precipitation intensity, precipitation will infiltrate the soil and percolate downwards either to the groundwater or to a restrictive layer at some depth in the profile (Hornberger, 1998). When the infiltration rate is greater than the precipitation intensity, locations fall into the broader category of Hydrologic Land Types *B* (Figure 2.1). If there is a restrictive layer in the profile, it can be either at shallow depths or at much greater depths. The surface layer of soils exhibiting a restrictive layer typically has either high organic matter content and is well structured, has a significant portion of sand, is from volcanic origin, has been glaciated, or has intensive impacts from moldboard or large disc plows. In these soil types, due to various processes, restrictive layers can form.

When a restrictive layer is at shallow depth, water will flow laterally over the restrictive layer with a driving force equal to the slope (Hydrologic Land Type *B1* in Figure 2.1). Soils become fully saturated when the lateral transport capacity becomes less than the incoming lateral or vertical flux. This occurs in times of excess rainfall at locations where the slope decreases, or where the conductivity of the soil decreases. In the saturated location, the excess subsurface lateral flow is transported as (saturation excess) overland flow. The driving force of the subsurface lateral flow is approximately equal to the slope of the restrictive layer when the slope is about 0.5 - 1%. Otherwise, the time it takes for the water to flow laterally through the soil to the stream channel depends on the slope and hydraulic conductivity, but generally occurs within 5 to 10 days after the rainfall event, after the threshold is exceeded. When installed, tile lines may accelerate the flow path within Hydrologic Land Type *B1.*

When the restrictive layer is deep or altogether not present (deeper than the elevation of the surface of the water body) water will flow predominately downwards via matrix and preferential flow until it reaches the water table and then flows under its own weight (i.e., piston flow) to the stream channel as base flow. This percolation flow path is typically slower than overland flow paths.

Percolation occurs within the Hydrologic Land Type *B2* in Figure 1**.** For ease of writing we will leave off the word hydrologic and call them simply "Land Types".

3.2 Hydrological Classification of Pollutants

Effectiveness of BMPs also depends on the chemical and biophysical characteristics of a particular pollutant. Although classification and terminology used to describe pollutant types vary throughout the literature, we are mainly interested in how they behave with respect to the flow path they follow. The type of flow path, the time that the pollutant is in the flow path, and the pollutant characteristics determine the dominant form of pollutant transport. Pollutants are transported as either sorbed (e.g., attached to soil particles) or dissolved in water. Therefore, we classify pollutants as either particulate or dissolved at the time of transport. When pollutants are weakly or strongly adsorbed, they are "two-phased" and occur in both dissolved and particle-attached states. Nutrients and pesticides can go through transformations and be present in various forms at the time of transport. In this review, we focus on pollutants typically associated with agriculture: phosphorus, nitrogen, pesticides and sediment. For weakly adsorbed pollutants we are more concerned with the dissolved form and for strongly adsorbed pollutants we are more concerned with the particulate form.

3.2.1 Pollutants of Concern

Phosphorus (P) can occur in multiple forms (Haygarth and Sharpley, 2000), but in order to determine what flow path P will follow, we classify P as soluble reactive phosphorus (SRP; i.e., dissolved) and as particulate phosphorus (PP). Sediment is transported through overland flow and erosion, and serves as a surrogate for the transport of PP and strongly adsorbed pesticides due to these contaminants' tendency to bind to soil particles.

Nitrogen occurs in many forms, as particulate organic matter, dissolved organic matter and various inorganic forms such as nitrate and nitrite (non-adsorbed), ammonium (strongly adsorbed), and can be rapidly transformed between states. The most common form of nitrogen (N) in water is nitrate (Logan *et al.*, 1980). Under aerobic conditions, other mobile forms of N (e.g. ammonium and nitrite) quickly transform to nitrate in the soil (Wild and Cameron, 1981).

Pesticides are classified as non-adsorbed, weakly adsorbed, or strongly adsorbed (Walter *et al.*, 1979; Smith and Ferreira, 1987; Reichenberger *et al.*, 2007). As stated above, weakly to strongly adsorbed pesticides are two-phased contaminants.

3.2.2 Pollutant Transport by Hydrologic Flow Paths

Strongly adsorbed chemicals are predominately associated with the particulate form and move in overland flow with particulates. In the subsurface path, strongly adsorbed chemicals adhere to the soil matrix. As a result, they are generally considered immobile. If adsorbed to very fine colloidal soil particles, small quantities can move significant distances. Non-adsorbed chemicals move at a similar rate as water.

In order to understand the movement of the two-phased (weakly to strongly adsorbed) chemicals, we need to understand what portion of the chemical is in the water phase and acts as dissolved and how much is attached to the sediment. Under equilibrium conditions (i.e. no net change in the balance of chemicals between dissolved and particulate phases), and with a low adsorption coefficient of 1, there is 100 times as much pesticide in the water phase as in the particulate phase. For an adsorption partition coefficient of 100 (intermediately adsorbed), half of the pollutant is transported with the sediment and half in the water. Only for strongly adsorbed chemicals, such as PP and some pesticides, is the transport principally associated with the particulate form. For strongly adsorbed chemicals, reducing erosion is an effective way to prevent pollutants from entering a water body.

For subsurface flow, the residence time in the flow path influences contaminant transport (Walter *et al.*, 1979; Logan *et al.*, 1980; Wild and Cameron, 1981; Reichenberger *et al.*, 2007). Subsurface flow can occur through preferential or matrix flow paths. In preferential flow paths, transport is fast and there is not sufficient time for equilibrium conditions to form within the soil matrix, and chemicals are transported rapidly downward, independent of adsorption until the terminus of the preferential path (Wild and Cameron, 1981; Gaynor and Findlay, 1995; Flury, 1996; Sharpley *et al.*, 2000; Gelbrecht *et al.*, 2005; Kim *et al.*, 2005; Reichenberger *et al.*, 2007). Matrix flow is a more gradual transport pathway and soil acts as a barrier to slow contaminant transport. Given a sufficiently long half-life, small but still significant quantities of pollutants can move down to groundwater (Walter *et al.*, 1979). Tile drainage causes an artificial preferential, lateral flow path, transporting weakly and non-absorbed pollutants in dissolved form and strongly adsorbed pollutants associated with fine soil colloids (once they arrive at the tile line) to water bodies.

3.2.3 Transport of Agricultural Chemicals as a Function of Hydrological Land Types

In our framework, we characterized three land types by specific flow paths and we simplified the classification of major pollutant types so that we can understand their transport along the major flow paths in each land type. In this section, we apply the framework to transport of agricultural related chemicals and sediment. We will then compare results of published field and plot scale studies with our conceptual framework to develop BMP recommendations based on how pollutants and sediment are transported.

Land type *A* (Figure 2.1), occurs, for example, in the Midwest in soils under conventional tillage practices where the high intensity summer convective storms are often present (Gassman *et al.*, 2010). To understand how chemicals move in land type *A* soils, we note that rain mixes with the soil solutes in a thin mixing layer of 0.5-2 cm at the soil surface where overland flow can pick up solutes **(**Ahuja *et al.*, 1981; Steenhuis *et al.* 1994, Gao *et al.*, 2004; Sanchez and Boll, 2005**).** Typically, infiltration rates in type *A* soils initially are high and then decrease over time, so there is a delay between the start of the rainfall and the initiation of runoff (Horton, 1933). In this initial period of infiltration, dissolved chemicals will move downward out of the mixing zone and are less available for transport by overland flow. Therefore the loss of nitrate in surface runoff is often less than the amount lost to leaching pathways (Walter *et al.* 1979; Wang and Zhu, 2011). Two-phased pesticides and SRP move downward much more slowly than nitrate resulting in increased availability when runoff occurs and can be more readily mobilized in overland flow (Saia *et al.*, 2013). Overland flow also induces erosion, which can carry strongly adsorbed pesticides, ammonia and PP.

Land type *B1* with shallow restrictive layers are found all over the US and are particularly expansive in the clay pan soils of the Midwestern U.S. (Ghidey *et al.*, 2005; Mudgal *et al.*, 2010a, b), as the dense glacial till layers in New York (Easton *et al.*, 2008), as argillic layers in the Palouse region in the Pacific Northwest (Brooks *et al.*, 2010), and many other locations in the U.S. (McDaniel *et al.*, 2008). These soils increase overland flow and subsurface lateral flow by inhibiting deep vertical infiltration, causing perched water tables. In many areas with such soils, tile drainage has been installed thereby reducing overland flow and the presence of saturated areas but providing a direct pathway for pollutants to surface water bodies (Strock *et al.*, 2010). Lateral flow in type *B1* landscapes occurs when a saturated lens forms atop of the restrictive layer and the hydraulic gradient is greater than the matric potential. Therefore places in the watershed that have steep slopes, conductive soils and small contributing areas (i.e., upslope areas) are usually unsaturated at the surface. In contrast, soils with a large contributing area and relatively flat slopes are saturated during periods in the year when there is more precipitation than evaporation. These periodically saturated soils occur in areas that are concave and in the bottom of the watershed near the stream channel.

Lateral flow primarily carries dissolved contaminants down the slope. The saturated areas producing saturation excess overland flow behave similarly as the runoff areas in land type *A* and can be source areas for SRP, PP (see Sanchez and Boll, 2005), organic N, pesticides and sediment. The main difference with land type *A* soils is that dissolved chemicals can originate from the entire subsurface lateral flow zone above the restrictive layer (Steenhuis and Muck, 1988). In addition, *B1* land types can be saturated for an extended period, and are, therefore, a source of denitrification in the landscape (Kuo, 1998). Subsurface lateral flow that exfiltrates in the saturated areas can mobilize SRP on its way to the stream. Water bodies in *B1* landscapes are therefore characteristically high in SRP and low in nitrate (Sanchez and Boll, 2005; Flores-Lopez *et al.*, 2010).

Land type *B2* has deep soils (Figure 2.1) and is common in many parts of the US such as in Michigan, Wisconsin and Minnesota. Leaching of nitrate is the most prominent concern because the subsurface soil structure allows deep infiltration. In these land types, transport of nitrates via the soil matrix and dissolved forms of pesticides via preferential flow paths is most common (Steenhuis and Parlange, 1990). Note that nitrate also moves in preferential flow paths, however the amount is generally substantially less than the fraction moved in matrix flow (e.g., Zhang *et al.*, 2011; Parn *et al.*, 2012; Spence *et al.*, 2012; Ballantine *et al.*, 2013). Nitrate in preferential flow is therefore less of a concern in groundwater. However, for pesticides, where even trace concentrations in groundwater are problematic, a small application of 2 kg/ha is sufficient to reach a level of a few parts per billion in ground water. Therefore, pesticide transport via preferential flow is a concern for groundwater quality.

We recognize that this framework is simplified and that there are many borderline cases in which water can follow different flow paths depending on the rainfall intensity, the condition of the soil, and topography. A review of rainfall characteristics can be used to assess the importance of temporal changes in land type characterization (Walter *et al.*, 2003, 2005). Because restrictive layers and soil depths are often not uniformly distributed across the landscape, it is likely that a watershed contains a patchwork of *B1* and *B2* land types. A good example is the Mohantango watershed in Pennsylvania, where the two sites draining in the East Mohantango creek are completely different. One has shallow soils with a restrictive layer and the other has deep soils (Needleman *et al.*, 2004). Similarly, the shallow upland soils and karst geology in the Lincoln Lake CEAP Watershed in Arkansas create spatial diversity in *B1* and *B2* land types (Edwards *et al.*, 1996). A patchwork of land types like this requires different BMPs throughout the watershed.

In many landscapes, regardless of land type, concentrated flow (e.g., rills and gullies) may occur due to topography and tillage/cropping practices (Grissinger, 1996). Effectiveness of BMPs (e.g., grassed waterways and sedimentation basins) that reduce pollutant transport through concentrated flow paths that are already formed is outside the scope of this review. However, BMPs that address overland flow in this review do aid in reducing the formation of concentrated flow paths. In the next section, we define BMP effectiveness based on the land types.

4. BMP Effectiveness Studies

We divided the literature on BMP effectiveness into plot and field studies, and watershed scale studies. Since hydrological land types and associated flow paths tend to be more uniform at a smaller spatial scale, we first considered the interaction of BMPs and flow paths within a land type from plot and field studies. Only experimental or physically-based model studies that included physical site characteristics were used in this analysis to ensure that results could be related to land types *A*, *B1* or *B2*. Physical site characteristics that we looked for included soil type, surface infiltration capacity or saturated hydraulic conductivity, soil depth, and climate information such as annual precipitation amount and precipitation intensity. In order to define a land type, we needed information about seasonal precipitation, types of dominant flow paths present, and soil type. We used the flow chart in Figure 2.2 to determine the land type using information contained in the study, and publicly available climate and soil survey data. Specific, quantitative thresholds of climatic and landscape characteristics in each land type cannot be reported here because these characteristics varied too much for each study and in each region.

To determine the dominant flow paths, we sought out studies that had sufficient soil and precipitation data or we further investigated publicly available soil survey and climate data. When we could not infer the dominant flow path, we eliminated the study from our analysis. Of the over 300 BMP effectiveness plot and field studies reviewed, approximately 80 were suitable for our analysis. Because this review presents a conceptual, process-based approach for understanding BMP effectiveness, rather than in depth analyses of individual BMPs, pollutants, or regions, we do not directly address narrowly focused BMP review papers in this analysis. Transferring lessons learned from empirical modeling studies was particularly difficult because models are typically calibrated to a specific area and the physical processes and dominant flow paths are not always discussed with respect to the physical mechanisms controlling losses. We considered only sites with natural,

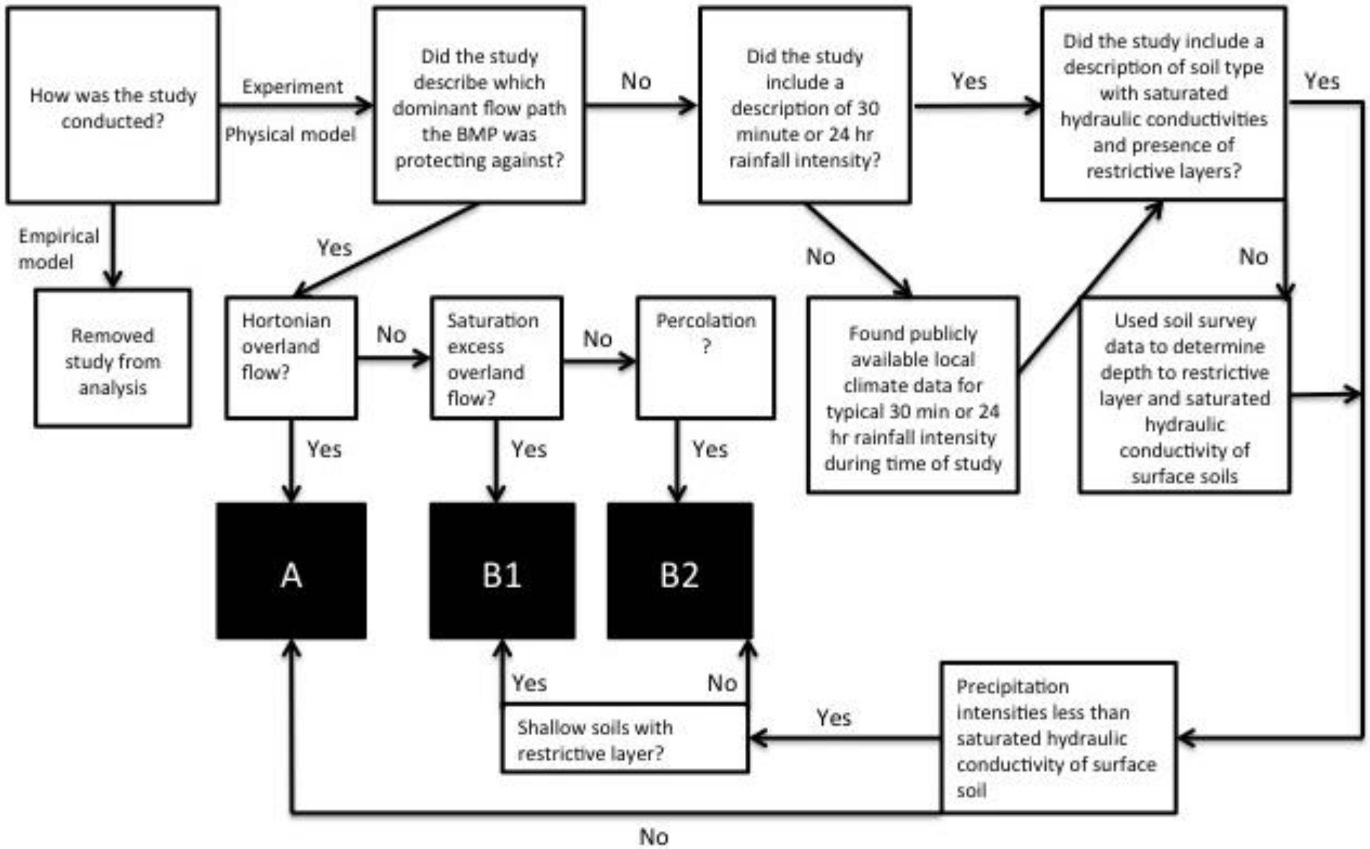


Figure 2.2 Land type classification decision flow chart for determination of *A*, *B1*, and *B2* land types for BMP effectiveness studies.

relatively fixed, soil and water characteristics. For this reason, irrigated systems were not considered in this study.

In the watershed scale analysis our conceptual framework was applied to Paradise Creek Watershed, ID, Lincoln Lake Watershed, AR, Cannonsville Watershed, NY, and Little River Experimental Watershed, GA. We made one or two-day site visits to these watersheds in 2009-2011. The information from plot and field studies was included in the analysis of the watershed scale studies, where BMP interactions become more complex. These four watersheds are examples of how the framework can be used as a tool to select an appropriate suite of BMPs for a given watershed.

4.1 BMP Effectiveness Analysis: Plot and Field Scale Studies

Plot and field scale BMP effectiveness study locations were grouped by *A*, *B1*, and *B2* land types to synthesize trends of BMP response. Specific findings for each land type and the most effective BMPs for the dominant flow paths associated with each land type are presented. Effective BMPs for each land type and their effect on the physical processes that target the dominant flow path for each land type are listed in Table 2.2 (e.g. slow overland flow velocity, increase infiltration capacity to reduce overland flow). The BMPs included in Table 2.1 and described in the text are the most commonly cited and demonstrated practices. We understand that there are effective BMPs that have not been evaluated in the published literature. The Conservation Reserve Program (CRP), which encourages farmers to convert highly erodible agricultural land into vegetative cover, and contour farming, for example, are likely effective for all land types, but plot and field scale studies that demonstrate their effectiveness are not readily published for all land types. Contour farming has been cited as effective (Kenimer *et al.*, 1997 cited in Reichenberger *et al.*, 2007; Gerontidis *et al.*, 2001; Deasy *et al.*, 2007), but it was difficult to parcel out the effectiveness of the individual practice as it was often combined with other practices (Van Doren *et al.*, 1950). While field observations by both producers and researchers suggest that gully plugs reduce overland flow and erosion in *B1* land types, the current literature lacks evidence that they are effective, and thus gully plugs are omitted from this review.

Table 2.1 Commonly effective BMPs for *A*, *B1*, *B2* land types and supporting studies.

Best Management Practice	A	B1	B2
Mulch Tillage	Ghidey *et al.* 2005 Deasy *et al.* 2007	Tollner *et al.* 1984 Ghidey *et al.* 2005 Lerch *et al.* 2010	-
No Tillage	Mostaghimi *et al.*1991 Forney *et al.* 2000 Ghidey *et al.* 2005 Lerch *et al.* 2011	Mostaghimi *et al.* 1991 Forney *et al.* 2000 Ghidey *et al.* 2005 Lerch *et al.* 2011	-
Terraces	Alberts *et al.* 1978 Dano and Siapno 1992 Chow *et al.* 1999	-	-
Conservation Reserve Program (CRP)	Udawatta *et al.* 2006 Jiang *et al.* 2007	-	-
Cover Crops	Wendt and Burwell 1985 Zhu *et al.* 1989 Kaspar *et al.* 2001	Wendt and Burwell 1985 Zhu *et al.* 1989	Meisenger *et al.* 1991 Fielder and Peel 1992 Staver and Brinsfield, 1998 Shepard 1999 Kay *et al.* 2009
Effective buffers*	Robinson *et al.* 1996 Cole *et al.* 1997 Clausen *et al.* 2000 Lee *et al.* 2003	-	Hubbard and Lowrance 1997 Mendez *et al.* 1999 Vellidis *et al.* 2001 Borin *et al.* 2004 Miller *et al.* 2010
NMP/IPM	Coelho *et al.* 2006 Ghidey *et al.* 2005 Mostaghimi *et al.* 1991 Mickelson *et al.* 1998	Mostaghimi *et al.* 1991 Isensee and Sadhegi 1993 Mickelson *et al.* 1998 Walter *et al.* 2001 Ghidey *et al.* 2005 Coelho *et al.* 2006	Isensee *et al.* 1990 Shepard 1996 Goulding *et al.* 2002 Vellidis *et al.* 2002
Tile Drainage	-	Dinnes *et al.* 2002	-
Drainage Ditch	-	Frankenberger *et al.* 1999 Zhang *et al.* 2013	-
Controlled Drainage	-	-	Feset *et al.* 2010 Skaggs *et al.* 2010 Lalonde *et al.* 1996 Evans *et al.* 1979
Crop Rotations	-	-	Randall *et al.* 1997

*Buffers must meet suitability components described in Table 2.3 to be considered effective for each land type.

Table 2.2 Summary of physical processes enhanced by each BMP. CRP refers to the Conservation Reserve Program. NMP refers to nutrient management plans and IPM to integrated pest management plans.

BMP	Decreases or changes timing of surface application	Slows overland flow velocity	Increases infiltration capacity	Increases surface roughness	Increases residence time in the mixing layer	Increases plant uptake
Mulch Tillage	-	Tollner *et al.* 1984 Ghidey *et al.* 2005 Deasy *et al.* 2007 Lerch *et al.* 2010	Tollner *et al.* 1984 Ghidey *et al.* 2005 Lerch *et al.* 2010	Ghidey *et al.* 2005 Deasy *et al.* 2007 Lerch *et al.* 2010	-	-
No Tillage	-	Mostaghimi *et al.*1991	Mostaghimi *et al.*1991 Forney *et al.* 2000	Mostaghimi *et al.*1991 Ghidey *et al.* 2005	-	-
Terraces	-	Alberts *et al.* 1978 Dano and Siapno 1992 Chow *et al.* 1999	-	-	-	-
CRP	-	Udawatta *et al.* 2006 Jiang *et al.* 2007	Udawatta *et al.* 2006 Jiang *et al.* 2007	Udawatta *et al.* 2006 Jiang *et al.* 2007	-	-
Cover Crops	-	Wendt and Burwell 1985 Zhu *et al.* 1989 Kaspar *et al.* 2001	Wendt and Burwell 1985 Zhu *et al.* 1989 Meisenger *et al.* 1991 Fielder and Peel 1992 Staver and Brinsfield 1998 Shepard 1999 Kaspar *et al.* 2001	Wendt and Burwell 1985 Zhu *et al.* 1989 Meisenger *et al.* 1991 Staver and Brinsfield 1998 Shepard 1999 Kaspar *et al.* 2001	Meisenger *et al.* 1991 Fielder and Peel 1992 Staver and Brinsfield 1998 Shepard 1999	Meisenger *et al.* 1991 Fielder and Peel 1992 Staver and Brinsfield 1998 Shepard 1999
Effective buffers*	-	Robinson *et al.* 1996 Cole *et al.* 1997 Clausen *et al.* 2000 Lee *et al.* 2003	Robinson *et al.* 1996 Cole *et al.* 1997 Lee *et al.* 2003	Robinson *et al.* 1996 Miller *et al.* 2010	Hubbard and Lowrance 1997 Vellidis *et al.* 2001	Hubbard and Lowrance, 1997 Mendez *et al.* 1999 Vellidis *et al.* 2001 Borin *et al.* 2004 Miller *et al.* 2010

Table 2.2, continued: Summary of physical processes enhanced by each BMP. CRP refers to the Conservation Reserve Program. NMP refers to nutrient management plans and IPM to integrated pest management plans.

BMP	**Decreases or changes timing of surface application**	**Slows overland flow velocity**	**Increases infiltration capacity**	**Increases surface roughness**	**Increases residence time in the mixing layer**	**Increases plant uptake**
NMP/IPM	Isensee *et al.* 1990 Mostaghimi *et al.* 1991 Isensee and Sadhegi 1993 Shepard *et al.* 1996 Mickelson *et al.* 1998 Walter *et al.* 2001 Goulding *et al.* 2002 Vellidis *et al.* 2002 Ghidey *et al.* 2005 Coelho *et al.* 2006	-	-	-	-	-
Tile Drainage		Dinnes *et al.* 2002	Dinnes *et al.* 2002	-	-	-
Drainage Ditch	-	Frankenberger *et al.* 1999 Zhang *et al.* 2013	-	-	-	-
Controlled Drainage	-	-	-	-	Skaggs *et al.* 2010	Lalonde *et al.* 1996 Skaggs *et al.* 2010
Crop Rotations	-	-	-	-	-	Randall *et al.* 1997

*Buffers must meet suitability components described in Table 2.3 to be considered effective for each land type.

4.1.1 Effective BMPs for A Land Types

In *A* land types, where Hortonian flow dominates, effective BMPs can be generally grouped into three categories: 1) BMPs that increase the soil's ability to infiltrate and store water, reducing overland flow, 2) BMPs that reduce application of the pollutant at the source, and 3) BMPs that reduce the overland flow water velocity, once generated. Of the 30 studies characterized as *A* land types, properly placed buffers, terraces, nutrient management plans (NMPs), Integrated Pest Management (IPM), the Conservation Reserve Program (CRP), cover crops, mulch tillage, and no-till were the most effective BMPs (see Table 2.1 and Table 2.2).

4.1.1.1 Structural and Vegetative BMPs

Buffer strips have been studied in many places, and are widely implemented, but they are not always effective. Examples of successful wetland construction and controlled drainage were also found, but these studies were not as prevalent. We grouped all vegetated filter strips, grassed waterways, and riparian buffer strips into the buffer strip category. Based on a review of buffer strips on Virginia farms in which 36% of buffers were completely ineffective (Hayes and Dillaha, 1992), we created a checklist to determine if a site is suitable for a buffer based on field slope, soil loss rates, site maintenance, and overland flow concentrations. We adapted this to create buffer strip suitability recommendations based on land type (Table 2.3). Buffer strips are effective for targeting overland flow paths if they reduced overland flow velocity, increased infiltration capacity, and increased surface roughness allowing deposition of sediment and removal of nutrients and pesticides. Buffers may be more effective for removing PP than SRP from surface flow (Uusi-Kamppa *et al.*, 2000). Buffer strips were effective when overland flow was shallow, dispersed, and uniform through the buffer (Dillaha *et al.*, 1986a; Dillaha *et al.*, 1986b; Robinson *et al.*, 1996; Cole *et al.*, 1997; Clausen *et al.*, 2000). A buffer strip study in northeastern Iowa (Robinson *et al.*, 1996) found that 70% of the sediment was removed in the first 3 meters of the buffer, with little removal observed with increased width. In this study, 67% of the total rainfall observed was during high intensity storms, triggering infiltration excess overland flow. The upland area adjacent to the buffer was an 18 m fallow strip, and was tilled every three weeks to minimize concentrated flow before overland flow entered the buffer strips. The steepest buffer strips in the study (slope gradient of 12%) were the least effective at reducing soil losses and overland flow. A comprehensive review paper of buffers ranging from 3-10 m, suggested that a buffer slope greater than 9.2% does not slow overland flow velocity enough for the buffer to effectively infiltrate overland flow or allow sediment deposition (Dillaha *et al.*, 1986a).

Table 2.3 Suitability of a site for buffer strips (modified from Hayes and Dillaha 1992 as cited in Barling and Moore 1994) and appropriate buffer characteristics for land types

Land Type	Landscape Factor	Suitability component for site and buffer	Comments
For all land types	Upland Characteristics	Slope limitations (e.g. Hayes and Dillaha (1992) found that slope of field must be less than 9.2% and greater than 1%, Franti *et al.* (1997) found the slope range to be between 3-12%)	Sites with higher slopes are not suitable for buffers; runoff velocity will be too high, reducing trapping efficiency to unacceptably low values. Sites with very small slopes are not suitable for buffers because hydraulic gradient is insufficient[1]
		Field cannot have excessive soil loss rates (e.g. Hayes and Dillaha (1992) found that soil loss rates must be less than 22.5 Mg/Ha)	If soil loss rates are excessive, other in field conservation practices must be used to reduce soil loss to acceptable levels otherwise rate of sedimentation in buffer will exceed the buffer trapping efficiency[1]
		The ratio of field area to buffer area must not be too great (e.g. Hayes and Dillaha (1992) found that this ratio must be less than 50:1)	If the ratio is greater, the site is unsuitable unless the soil erosion rates are very low. [1]
		Internal drainage ways such as rills and gullies must be targeted with vegetated filter strips, conservation tillage, or gully plugs before flow enters buffers	Buffers can only effectively filter pollutants if overland flow is dispersed through sheet flow, and is not in the form of concentrated flow.[1]
		Predict and identify overland flow source areas before designing and implementation of buffer using topography, soil characteristics, vegetation, and climate[2]	Buffers must intercept the dominant flow path that transports pollutants to be effective
	Maintenance	Landowner/ operator must be willing and able to maintain buffer	This includes mowing, controlling growth of undesirable weeds; inspection and repair after major storm events, excluding grazing and vehicle disturbance, especially during buffer establishment[1]
A	Infiltration excess overland flow path	Buffer strip must reduce overland flow velocity increase infiltration capacity and increase surface roughness	The buffer must allow for deposition of sediment and removal through uptake of N and P, denitrification, or degradation of pesticides
B1	Saturation excess overland flow path	Buffers are rarely effective for B1 land types due to perched water tables and concentrated flow	Forested buffers and constructed wetlands can provide suitable increase in uptake, infiltration capacity, retention and denitrification and degradation of pollutants
B2	Leaching flow path	Deep rooted, forested buffers effective only if leaching flow path is to a shallow ground water source[3]	Pollutants can be reduced only when the soil root zone is deep enough to intercept shallow ground water subsurface flow[4]

1. Hayes and Dillaha, 1992; 2. Barling and Moore, 1994; 3. Lowrance et al. 1997; Mendez et al. 1999; Borin et al. 2004; Miller et al. 2010; 4. Hubbard and Lowrance, 1997; Vellidis et al. 2002; 5. Franti et al. 1997

The benefits of terracing for *A* land types was demonstrated in a study in southwestern Iowa, where seasonal discharges of overland flow, nutrients, and sediment were all reduced tenfold in the terraced scenarios (Gassman *et al.*, 2010). Authors cited the terraces as being most effective during critical erosion periods, which is likely when infiltration excess overland flow is generated during extreme rainfall. Terraces were also highly effective on *A* land types with steep slopes (30-60%) (Dano and Siapno, 1992) and moderately steep slopes (Chow *et al.*, 1999). Terraces are likely most successful in *A* land types with moderately steep to steep slopes because they slow overland flow enough to increase water residence time within the terrace, allowing for infiltration and increased soil moisture storage for plant uptake (Chow *et al.*, 1999) as well as possible degradation of pesticides, deposition of sediment, and denitrification. As seen in the Iowa study, however, adding tile lines during construction of terraces can lead to increased nitrate losses (Gassman *et al.*, 2010).

4.1.1.2 Source Management

Nutrient management plans and IPM were highly variable in effectiveness based on the application amount, timing, and strategy (Table 2.1 and Table 2.4). Incorporation of pesticides and herbicides into the mixing zone (A horizon) of soil through mulch tillage or chisel-plowing and the direct injection of manure reduced pollutant loss in overland flow (Withers *et al.*, 1998; Sharpley *et al.*, 2000). In a manure application management study in Iowa, Coelho *et al.* (2006) optimized the rate and method of side-dressed injection application of liquid swine manure in corn fields to match the nitrogen uptake of the crop and minimize nitrate losses to ground and surface waters. When the application exceeded the crop demand for N, nitrate concentrations increased in both the topsoil and drainage water. Injection application was recommended because it maximized the crop yield while minimizing nitrate loss to soil, groundwater, and surface water.

Ghidey *et al.* (2005) found that incorporating soil-applied pesticides below the upper 2-5 cm of the soil is one of the most effective ways to reduce overland flow of pesticides. They studied the effect of split application of pesticides based on the hypothesis that multiple smaller applications would reduce loss. However, that did not prove true. Since pesticide loss was greatest immediately following application, split application created multiple periods of vulnerability for overland flow particularly during large storm events, despite the smaller quantities of application. They concluded, therefore, that timing of application is far more important than rate of application. However, it is logical that any reduction in application rate will reduce potential pollution by pesticides (Hall *et al.*, 1972, cited in Mickelson *et al.*, 1998).

These NMP and IPM examples provide evidence that even within the *A* land type, there is not a single NMP or IPM strategy. Site specific plans that are well managed may provide greater success, especially when applications do not coincide with large precipitation events, and are applied when crops can uptake the chemicals or there is enough organic matter and residue in the soil to either immobilize or bind them allowing for biodegradation (see Table 2.4).

Table 2.4 Summary of source BMP effectiveness findings by land type: NMPs and IPM.

Effectiveness component	**Prioritize for Land Types**	**Selected Studies**
Alter ratio of N and P in animal feed	*A,B1,B2*	Heathewaite *et al.* 2000. Powell *et al.* 2001; Sharpley *et al.* 2007; Swink *et al.* 2010
Install exclusionary fencing in grazing sites	*A,B1,B2*	Line *et al.* 2000; James *et al.* 2007; Kay *et al.* 2009; Rao *et al.* 2009; Flores-Lopez *et al.* 2010
Optimize side-dressed manure injection	*A,B1,B2*	Jokela *et al.* 1996; Coelho *et al.* 2006
Reduce application rate; optimize for crop uptake and denitrification and degradation processes	*A,B1,B2*	Isensee and Sadeghi, 1993; Edwards *et al.* 1996; Forney *et al.* 2000; Coelho *et al.* 2006
Avoid application on HSAs or Variable Source Areas	*A, B1*	Walter *et al.* 2001; Heathewaite *et al.* 2005
Incorporate fertilizer/ pesticides into the mixing zone	*A and B2*	Mickelson *et al.* 1998; Withers *et al.* 1998; Sharpley *et al.* 2000; Ghidey *et al.* 2005; Lerch *et al.* 2011
Avoid application before large precipitation events, minimize time between application and planting	*A, B1, and tile drained land types*	Steenhuis and Walter 1980; Blackmer and Sanchez 1988; Edwards *et al.* 1997; Randall *et al.* 1997; Withers *et al.* 1998; Higgs *et al.* 2000; Dinnes *et al.* 2002; Ghidey *et al.* 2005; Reichenberger *et al.* 2007; Sharpley *et al.* 2007

4.1.1.3 Tillage and Crop Management Practices

Reduced tillage practices increase soil infiltration capacity, reducing infiltration excess overland flow during high precipitation events. In the examples where no-till was effective in reducing overland flow (Mostaghimi *et al.*, 1991; Forney *et al.*, 2000; Ghidey *et al.*, 2005; Lerch *et al.*, 2011), the topography was flat, and the soil type was primarily silt loam, and in some of the pesticide cases, incorporation of the pesticides coupled with the no-till operations ensured greater reductions in pesticide transport. In the no-till example of Mostaghimi *et al.* (1991), nitrate concentrations in overland flow were reduced by 50% compared to conventional tillage and the nitrate yield in overland flow was comparable to control plots without any fertilizer application. Mickelson *et al.* (1998) also found that incorporation of pesticides into a silty clay loam soil in Knoxville, IA was an effective method for reducing pesticide overland flow.

With the introduction of reduced tillage operations on *A* land types, the dominant flow path can shift from infiltration excess to saturation excess in shallow soils or vertical leaching in land types with deep soils. A study in Maryland (Isensee and Sadehgi, 1993) compared the response of corn plots to no-till and conventional tillage over a 2-year study period. During the summer months, the effectiveness of no-till versus conventional tillage was primarily a factor of soil infiltration rates and antecedent soil moisture conditions. Overland flow rates were greater from no-till plots when the time between rainfall events was less than 7 days, overland flow from conventional tillage was greater when the time between events was more than 7 days. In this particular study, the dominant flow path may have switched from saturation excess during multiple rainfall events that occurred within 7 days to infiltration excess when rainfall occurred on drier soils with lower infiltration capacities. The study also found that slow release pesticide application on both tillage systems resulted in lower pesticide concentrations in overland flow and was a good option for land types that alter between infiltration and saturation excess.

There are multiple cases where no-till reduced overland flow and associated pollutants such as soil loss and/or TP, but as a result, enhanced leaching of other pollutants such as nitrates or soluble pesticides due to the increased infiltration capacity (Flury, 1996; Carter, 1998; Forney *et al.*, 2000; Shipitalo *et al.*, 2000). The national movement towards mulch tillage practices in the past century has effectively reduced sediment loss from upland areas across the country (Blevins *et al.*, 1998). Mulch tillage is slightly less effective at reducing pollutant delivery than no-till (Lerch *et al.*, 2011), but there is also less risk of mulch tillage triggering new flow paths, and it is less costly to implement. Before a BMP is implemented to address infiltration excess overland flow and erosion issues, the potential land type transition and dominant flow path change to saturation excess, subsurface lateral flow, or leaching should be examined. This can be done through a modeling approach (See Brooks *et al.*, *this issue*), through soil type analysis, or from observations of producers who have already implemented the BMP in a similar land type and have witnessed how the BMP impacts the dominant flow path. As discussed in the next two sections, soil depth is a key characteristic for the potential to trigger contradictory flow paths.

When cover crops are established prior to major hydrological events, they improve infiltration capacity, uptake, surface roughness, and they slow overland flow velocity, as cited in multiple studies on *A* land types (Wendt and Burwell, 1985; Zhu *et al.*, 1989; Kaspar *et al.*, 2001). The USDA CRP has also been highly successful in removing *A* land types with high erosion rates from agricultural production (Hansen, 2007; Tomer and Locke, 2011). With the reintroduction of perennials in these areas, the hydraulic conductivities of the soils are often enhanced (Jiang *et al.*, 2007), thus reducing overland flow. As well, the added vegetative cover can reduce erodibility.

4.1.2 Effective BMPs for B1 Land Types

In a *B1* land type, saturation excess and subsurface lateral flow are dominant flow paths, and in hilly terrain formation of gullies may occur. After source management through NMPs and IPM, tile drainage best addresses these dominant flow paths (Table 2.1). However, while artificial drainage may decrease the overland flow volume, it may create a subsurface flow path, which does not necessarily reduce the overall pollutant transport. *B1* land types are challenging to manage due to the presence of perched water tables above shallow restrictive layers. The flow path cannot easily be altered. The literature lacks convincing evidence of effective BMPs for land types dominated by saturation excess processes (Table 2.1 and Table 2.2).

4.1.2.1 Structural and Vegetative BMPs

Tile drainage is a common practice to manage perched water tables because it enhances infiltration thus reducing surface flow path activation. However, it also increases the likelihood of subsurface transport of pollutants, especially in soils of high hydraulic conductivity (Sharpley *et al.*, 2000; Dinnes *et al.*, 2002). An in-depth review of BMP effectiveness in tile-drained landscapes suggests employing multiple management strategies, including NMPs, diversifying crop rotations, cover crops, and conservation tillage to reduce the impact of tile drainage on pollutant export (Dinnes *et al.*, 2002). Removing N once delivered to the stream through wetlands and biofilters is also a recommended practice. In addition to tile drainage, diversion ditches also have been constructed to intercept subsurface lateral flow in perched systems and reduce downslope soil saturation (Frankenberger *et* *al.*, 1999; Easton *et al.*, 2008; Zhang *et al.*, 2013). Upslope drainage which intercepts clean water above a potential pollutant source area, such as a confined feeding lot or barnyard, is also highly recommended (Scott *et al.*, 1998). Examples of effective buffers for *B1* land types were not found (see Table 2.3).

4.1.2.2 Source Management

Because perched water tables are so difficult to manage, BMPs that target the source of the pollutant application like NMPs and IPM are generally the most effective practices for overall reductions (Table 2.1). Moving the pollutants out of the periodically saturated areas to the uplands where there is lateral flow through the soil is effective as has been proven in the New York City source watershed in the Catskill Mountains (Bishop *et al.*, 2005; Easton *et al.*, 2009). A study in the Catskills region by Walter *et al.* (2001) illustrated just how beneficial NMPs can be for a *B1* land type where shallow lateral flow and saturation excess overland flow are predominant. This study identified sources of overland flow generation at a hillslope scale, both spatially and temporally, and avoided spreading manure in these areas. Phosphorus levels were reduced at the watershed scale using this spatial and temporal NMP. Table 2.4 illustrates other key “source” BMP findings.

4.1.2.3 Tillage and Crop Management Practices

Depending on the depth to the restrictive layer, overland flow volume may not decrease with enhanced infiltration capacity due to the saturation excess overland flow mechanism inherent in these systems. The addition of organic matter and surface residue from conservation and no-till practices are beneficial for denitrification, degradation (Fawcett *et al.*, 1994), and sorption processes. Increased surface roughness also reduces the risk of soil erosion (Walter *et al.*, 1979; Fawcett *et al.*, 1994; Mudgal *et al.*, 2010a). The study by Ghidey *et al.* (2005) described above in the *A* land type section showed that reduced tillage can prevent Hortonian overland flow. Because this study was done in an area with restrictive layers, once the reduced tillage improved the infiltration capacity, this *A* land type transitioned to a *B1* land type. In soils prone to overland flow such as the claypan soils with restrictive layers studied by Ghidey *et al.* (2005) in Missouri, no-till was not effective for reducing herbicide loss through overland flow unless pesticides were incorporated into the soil profile. Conservation tillage is more effective for soils with restrictive layers. Soils in Ghidey *et al.* (2005) were characterized as silt loam and silty clay loam, with an argillic layer less than 36 cm deep leading to perched water tables. For this *B1* land type, the dominant flow path was saturation excess overland flow because the field area was flat and there was not enough of a gradient to promote subsurface lateral flow. In a situation like this, buffers and terraces are not effective because there is not enough slope gradient. Both no-till and conservation tillage were found to be effective in this area in plot and field scale studies, but to differing degrees, because these intensively cultivated soils likely had a reduced infiltration rate. In a 1993 to 2001 plot scale study, Lerch *et al.* (2011) found that no-till without pesticide incorporation reduced atrazine by 90 g/ha, which was three times the reduction of mulch tillage with incorporation. Metolachlor was reduced by 31 g/ha; this was double the reduction that mulch tillage had even with herbicide incorporation.

Another plot scale study by Ghidey *et al.* (2005) in the same area from 1997 to 2002 not only found greater pesticide reductions from no-till than mulch till, but also found that no-till management actually had greater overland flow rates than mulch tillage. The authors suspected that the mulch tillage practice broke up the sealed soil surfaces that can occur in silt loam soil, resulting in more micro relief and faster drying of the soil whereas the no-till systems did not significantly develop preferential flow paths, perhaps due to the restrictive layers. No-till increases the infiltration capacity of the soil, but unless water can percolate through the restrictive layer or the soil structure has enhanced its water holding capacity, the soil profile will eventually saturate and create saturation excess overland flow. No till systems should increase surface residue, prevent soil crusting, increase micro relief in the soil, and increase the drying out of the soil (Ghidey *et al.*, 2005). It is possible that in the study period, these soil transformations had not yet occurred.

The plot and field scale studies in Missouri (Ghidey *et al.*, 2005; Lerch *et al.*, 2011) are good examples of situations where mulch tillage and no-till were effective for surface pollutants, but did not reduce vertical leaching of dissolved pollutants because of the restrictive layer, and did not promote subsurface lateral flow because of the flat terrain and shallow soil. The lessons learned in Missouri regarding no-till and mulch tillage can be applied to other locations with flat terrain, restrictive layers, and high rainfall intensity that are transitioning to mulch tillage and no-till practices (Table 2.1).

4.1.3 Effective BMPs for B2 Land Types

Deep percolation is the dominant flow path in *B2* land types due to deep, well-drained soils with high infiltration capacities. To best reduce the leaching of pollutants of concern, application at the source should be reduced and the residence time in the mixing layer (0.2-5cm from soil surface) should be increased to enhance denitrification, degradation, and crop uptake of pollutants (Fielder and Peel, 1992; Flury, 1996; Shepard, 1999; Dinnes *et al.*, 2002; Reichenberger *et al.*, 2007; Kay *et al.*, 2009). BMPs like buffers, controlled drainage, conservation tillage paired with cover crops or source BMPs most effectively enhanced residence time in the mixing zone (Table 2.2).

4.1.3.1 Structural and Vegetative Practices

Buffers were effective only if the leaching flow path was to a shallow groundwater source such that the buffer vegetation interacted with the dominant flow path (Lowrance *et al.*, 1997: Mendez *et al.*, 1999; Borin *et al.*, 2004; Miller *et al.*, 2010; see Table 2.3). Studies of effective riparian buffers near Tifton, Georgia indicated that when the buffer root zone was deep enough to intercept shallow subsurface flow from upslope contributing areas, nitrates and pesticides could be removed in *B2* land types (Hubbard and Lowrance, 1997; Vellidis *et al.*, 2002).

Controlled drainage is a commonly used practice in *B2* land types (Table 2.1). Studies showed that by maintaining a saturated root zone, pollutant concentrations and leaching could be reduced by increasing residence time in the mixing zone and increasing plant uptake (Evans *et al.*, 1979; Lalonde *et al.*, 1996; Skaggs *et al.*, 2005; Feset *et al.*, 2010). In a North Carolina study with sandy soil, Dukes *et al.* (2003) tested controlled drainage systems on both *B1* and *B2* land types. In the *B2* land types, controlled drainage led to a 73% reduction in nitrates in shallow groundwater. The authors hypothesized that the reductions were due to enhanced denitrification deeper in the soil profile.

4.1.3.2 Source Management

In a three-year experimental study in Iowa (Blackmer and Sanchez, 1988), 49-64% of fertilizer applied in the fall was lost to leaching rather than plant uptake from corn production. The loss of nitrogen through the top 1.5 m of the soil profile appeared to be due to precipitation events paired with the lack of cover crops to uptake the excess nitrogen. To best manage *B2* land types, source management BMPs like NMPs and IPM should be implemented as shown by Goulding *et al.* (2000) and Vellidis *et al.* (2002). In an event-based nitrogen leaching study at a 157-year-old agricultural experiment site in Rothamsted, UK, Goulding *et al.* (2000) found that NMPs that increase N efficiency decrease N leaching by 74% compared to 120 years ago. They also found that even plots that had not received fertilizer in over 150 years still leached N after rain events during the beginning of the water year. This study highlights that even with increased N uptake and efficiency in agriculture, rainfall events may still release pollutant residue from prior applications. In the study by Isensee *et al.* (1990) in Maryland, pesticide concentration in leachate was greater when application occurred prior to a large rainfall event. Authors stated that during these rainfall events, preferential flow paths may have been triggered, enhancing the leaching rate. Reduction of application and enhancement of crop uptake can help to buffer the overall effects of event-based leaching.

4.1.3.3 Tillage and Crop Management Practices

Similar to the *B1* land type, addition of organic matter and surface residue from conservation and no-till practices are beneficial for denitrification, degradation, and sorption processes in *B2* land types (See Table 2.1 and Table 2.2). In land areas with deep soils, conservation tillage and no-till can convert *A* land type flow paths to *B2* land type flow paths by increasing the infiltration capacity. Cover crops may be added to *B2* land types to targe2.t both when the landscape is a seasonal *A* land type to increase infiltration capacity, and to increase time in the mixing zone when it is a seasonal *B2* land type. Cover crops have increased residence time in the mixing layer in studies by Meisenger *et al.* (1991) and Staver and Brinsfield (1998).

While conservation tillage increases infiltration capacity compared to conventional tillage, Shipitalo *et al.* (2000) found that the difference between leaching rates in conservation tillage and conventional tillage were minimal. Conservation tillage typically transports the greatest amount of solutes during the first precipitation event after chemical application via macropores, followed by reduced solute transport in subsequent events. Therefore, if NMPs and IPM align application of agrichemicals after a major precipitation event in conservation tillage systems, solute leaching may be less if time between events is sufficiently great.

4.1.4 Plot and Field Scale Studies Summary

Patterns observed for effective BMPs by land type, the temporal and spatial variability for land type transitions, and recommendations for future plot and field scale BMP effectiveness studies emerged from the BMP analysis at plot and field scale by land type.

Overall, regardless of scale or land type, site-specific “source” BMPs that included NMPs and IPM, on the basis of this review, were determined to be the most effective way to reduce pollution (Logan, 1990; Edwards *et al.*, 1996; Lord and Mitchell, 1998; Coelho *et al.*, 2006, 2007; Kay *et al.*, 2009). By reducing excess application, and avoiding periods of high precipitation, there were fewer pollutant issues (Edwards *et al.*, 1997a; Edwards *et al.*, 1997b Dinnes *et al.*, 2002). Key findings from source BMP effectiveness studies are illustrated in Table 2.4. Buffer strips were widely implemented, but were often not effective (Table 2.3). Management for high intensity rainfall events or for rainfall after soil crusting has occurred may be nearly impossible, as highlighted in Isensee and Sadhegi (1993), Isensee *et al.* (1990), and Goulding (2000), although from author observations, some growers in northern Idaho use a harrow to break up soil crusting after planting. However, as indicated, slow release pesticides, cover crops, or application planning can provide protection against the impact of these rainfall events.

Hydrological land types may be variable in space and time. Saturation excess overland flow may be the dominant flow path after spring infiltration excess overland flow. Or, for example, later in the season, an area with deep soils may transition to a *B2* land type with leaching as the primary flow path if high intensity rainfall patterns that cause infiltration excess overland flow dissipate. By implementing a BMP to address infiltration excess overland flow, a grower may inadvertently trigger a leaching pathway. When managing for both surface and subsurface transport of dissolved and particulate pollutants, conservation tillage paired with source BMPs that address timing and quantity of application can effectively reduce surface transport. At the same time, a suite of source and conservation tillage BMPs can prevent subsurface pollutant transport through preferential flow by overall reduction of application (Shipitalo *et al.*, 2000).

4.2 BMP Effectiveness Analysis: Watershed Scale Studies

Targeting BMPs within a watershed with limited conservation dollars is a common challenge. When extending beyond the plot and field scales, it can be difficult to determine which BMPs will be most effective in what locations. To aid in this effort, we now examine BMP effectiveness in watershed scale studies and how cumulative effects of BMPs at this scale can be explained within our conceptual framework. We included watersheds in this examination

if information about the following four characteristics was available: spatial variability of soils, temporal variability of climate, identification of HSAs, and the type of BMPs installed. If one or more of these characteristics was not included in published studies, it was difficult to determine land types across the watershed or to critique the placement of BMPs. For some watersheds, that information was found in alternate sources (described below). We reviewed studies of BMP implementation in 18 watersheds and found that few provided the necessary information to determine land types present in the watershed or to determine the effectiveness of each type of BMP utilized. If future watershed scale studies include this information, the land type conceptual framework can be applied more broadly. We focused on four NIFA-CEAP watersheds that some or all of the authors had visited and for which information was available. These include Paradise Creek watershed in Idaho (ID), Lincoln Lake watershed in Arkansas (AR), Cannonsville watershed in New York (NY), and Little River Experimental watershed in Georgia (GA). We used these watersheds as examples of how to apply the conceptual framework to target BMP placement at the watershed scale.

Prior to site visits to these watersheds, a land type classification was prepared based on watershed descriptions in publications and readily available data sources. Preliminary classifications were made by examining the publications and then were confirmed using outside data sources, including the rainfall frequency atlas for the US (Hershfield, 1961) and the NRCS Soil Survey. Monthly maximum 30 min rainfall rates for ID, AR, NY, and GA based on 30-year climate data (Figure 2.3) were used to determine precipitation characteristics for land type classification. In addition, NRCS Soil Survey data on soil depth to a restrictive layer, slope range, and the A-horizon hydraulic conductivity (K_{sat}), adjusted for conventional, mulch tillage and no-till conditions (Table 2.5) were also used. Depth to a restrictive layer was identified as depth to bedrock, an argillic or fragipan horizon, or the depth of an abrupt increase in clay content and/or bulk density. The no-till K_{sat} was the value found in the NRCS Soil Survey. K_{sat} for conventional tillage was determined based on soil texture (Flanagan and Livingston, 1995). For conservation tillage, the K_{sat} value was the average of the no-till and conventional tillage value. During the site visits, the land type classifications were presented and our estimates of dominant flow paths were discussed with local watershed scientists. In Table 2.5, land type classifications were based on soil depth, K_{sat}, and the monthly 30 min rainfall intensity shown in Figure 2.3 for the predominant runoff period of December-March. In each NIFA-CEAP watershed, we received confirmation from local scientists and watershed managers that the land type classifications were correct and that we correctly derived the dominant flow paths from the land type classification. Slight differences in hydrological understandings were due to local knowledge, such as presence of karst hydrology in Lincoln Lake watershed.

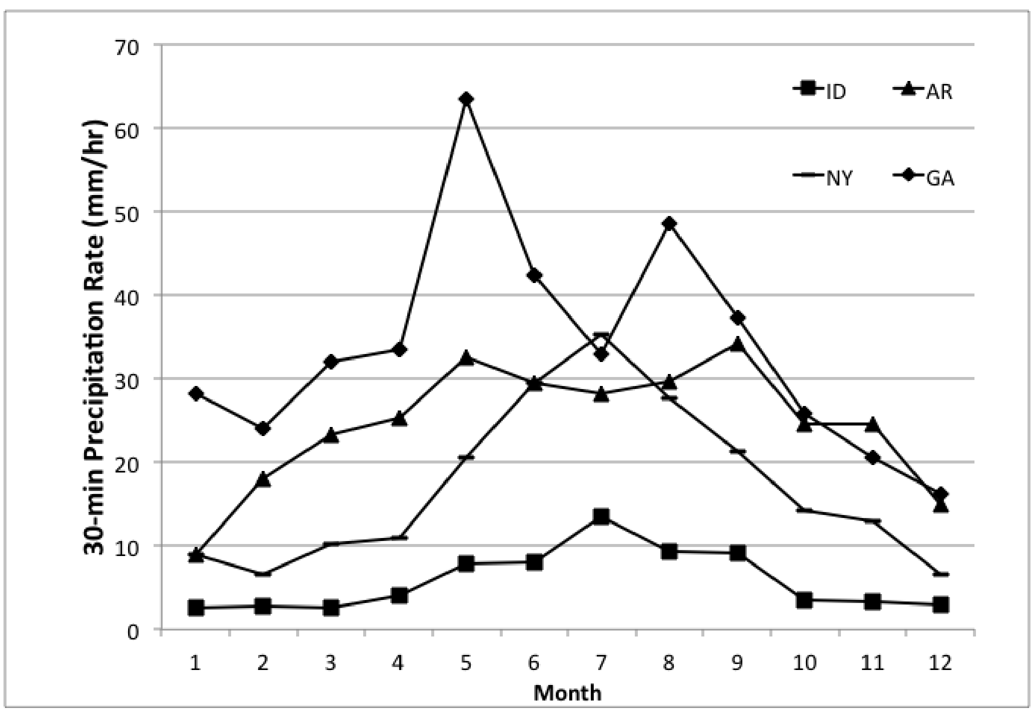


Figure 2.3 Mean Monthly Maximum 30 min Rainfall Rates for Idaho, Arkansas, New York, and Georgia Based on 30 Year Climate Data (source: Hershfield, 1961. Rainfall frequency atlas of the US).

Table 2.5 Typical Soil Depth to Restrictive Layer, Land Slope, and A-Horizon Hydraulic Conductivity for Soils in Four Watersheds, with Land Type Based on Monthly Mean 30 min Rainfall Rates during Major Runoff Periods in each Watershed Region.

Watershed & Soil Type	Soil Depth (cm)	Land Slope (%)	CT A-horizon K_{sat} (mm/hr) - LT[1]	MT A-horizon K_{sat} (mm/hr) - LT[1]	NT A-horizon K_{sat} (mm/hr) - LT[1]
Paradise Creek Watershed, ID (threshold precipitation intensity: 10 mm/hr)					
Palouse silt loam	150	2 - 5	2 - A	18 - B2	32 - B2
Southwick silt loam (Argillic of fragipan)	97	5 - 8	5 - A	18 - B2	32 - B2
Taney silt loam (fragipan)	69	8 - 35	4 - A	18 - B1	32 - B1
Garfield silty clay loam	20	8 - 20	6 – A	8 - A	10 - B1
Lincoln Lake Watershed, AR (threshold precipitation intensity: 20 mm/hr)					
Pembroke silt loam (Argillic)	200	0 - 2	5 - A	18 - A	33 - B2
Linker silt loam (Argillic)	89	3 - 8	9 - A	50 - B1	330 - B1
Johnsburg silt loam (fragipan)	60	0 - 6	5 - A	18 - A	33 - B1
Captina silt loam (fragipan)	51	1 - 8	5 - A	50 - B1	324 - B1
Cannonsville Watershed, NY (threshold precipitation intensity: 10 mm/hr)					
Elka channery silt loam (stony)	180	15 - 35	10 - B2	21 - B2	79 - B2
Lackawanna silt loam (fragipan)	71	0 - 55	10 - B1	21 -B1	32 - B1
Collamer silt loam (lake plain)	53	0 - 25	5 - A	18 - B1	32 - B1
Little River Watershed, GA (threshold precipitation intensity: 30 mm/hr)					
Lakeland sand	216	0 - 12	33 - B2	181 - B2	330 - B2
Tifton loamy sand	99	0 - 8	31 - B2	181 - B2	331 - B2
Cowarts fine sandy loam (perched water)	70	1 - 25	20 - A	61 - B1	100 - B1

[1]CT = Conventional Tillage; MT = Mulch Tillage; NT = No Tillage; LT = Land Type.

4.2.1 Paradise Creek Watershed, ID

Paradise Creek watershed (4,890 ha) in north central Idaho is characterized by a patchwork of predominantly *B1* and *B2* land types due to well-drained silt loam soils, often with shallow argillic layers, and low intensity precipitation (see Figure 2.3). However, when conventionally tilled or frozen, a soil crust forms reducing the infiltration capacity creating *A* land types (Table 2.5). When subjected to freezing, the *A* land type causes very high overland flow and erosion rates resulting from low infiltration capacities of frozen soil. Steep slopes (up to 35%), also characteristic of this region, lead to converging overland flow, which creates gullies (Brooks *et al.*, 2010).

Implementation of conservation tillage and contour farming starting in the mid-1970s and CRP in the 1980s (Carlson *et al.*, 1994; Kok *et al.*, 2009) drastically altered the dominant hydrologic flow path within this watershed to those associated with *B1* and *B2* land types (Table 2.5), (Brooks *et al.*, 2010). That shift on upland fields facilitated increased infiltration and reduced erosion. Long-term monitoring showed significant reductions in watershed sediment loading from the time of conventional tillage practices (Brooks *et al.*, 2010), but with the need for further reductions.

Between 2000 and 2003, various BMPs were installed throughout the rural and urban parts of the watershed including sedimentation basins (also called gully plugs), buffer strips, no-till, stream bank stabilization, riparian and wetland restoration, and bridge crossings. Some functioned well, such as gully plugs that reduced soil loss from upslope contributing areas in converging parts of the landscape. But not all BMPs were placed in HSAs. For example, buffer strips were installed by willing landowners, but not necessarily where conditions were optimal, such as below steep slopes, where concentrated flow paths are present, or in areas continually inundated during the winter (as observed by the authors). Based on recommendations (e.g. Stewart *et al.*, 1975; Veith *et al.*, 2004; Mulla *et al.*, 2005), and from our conceptual framework to improve targeting, we infer that basin-wide reductions would be more significant if land types had been considered when BMPs were placed. For example, placing buffer strips in *A* land types with low to moderate slopes and not in *B1* land types, and converting farming practices on *A* land types to either no-till or CRP, would further reduce sediment loads at the watershed outlet.

4.2.2 Lincoln Lake Watershed, AR

The Lincoln Lake Watershed (3,240 ha) in northwestern Arkansas exhibits a complex land type configuration as well as temporal shifts due to high intensity precipitation particularly during May through September (Figure 2.3). The watershed has shallow soils in the upland areas. Deeper soils with moderately good to excessive drainage comprise 70% of the land area, primarily near Lincoln Lake and in forested areas (Edwards *et al.*, 1996). Deep leaching to a karstic groundwater system is, therefore, dominant in the majority of soil types and thus the watershed is predominantly *B1* and *B2*. However, during high intensity precipitation events from May to September, *A* land types can result (Table 2.5). Land use in the watershed is primarily grass/hay and pasture land, and poultry operations (Edwards *et al.*, 1996, 1997; Chaubey *et al.*, 2010; Gitau *et al.*, 2010).

NMPs promoting manure application rates to meet crop N requirements reduced nitrate losses by 35 to 75% in the Lincoln Lake Watershed (Edwards *et al.*, 1996). Furthermore, NMPs were most effective when manure was not applied during wet antecedent conditions, in order to prevent nitrates from moving with subsurface lateral flow, and when manure application was prohibited within 10 m of surface waters (Edwards *et al.*, 1997b). While other BMPs such as waste utilization, pasture and hay land management, dead poultry composting, and waste storage structure construction were also implemented, the authors cited the NMP as the key BMP, which is in line with the results from plot and field scale studies. In a watershed like Lincoln Lake, with the unpredictable and intense nature of the January to April storm events, NMPs that reduce pollutants at the source are highly recommended.

4.2.3 Cannonsville Watershed, NY

While the Cannonsville Watershed (117,900 ha) in New York State (James *et al.*, 2007; Rao *et al.*, 2009; Flores-López *et al.*, 2010) is primarily forested, the agricultural area (17% of the watershed area), which is dominated by dairy operations, has a strong presence. The dairy operations have led to eutrophication problems in respective downstream reservoirs. Low intensity precipitation is predominant in the Cannonsville Watershed except for the occasional thunderstorm during summer (Figure 2.3; Walter *et al.*, 2001; Walter *et al.*, 2003). Near stream areas and areas with shallow soil over a slowly permeable glacial till soil or bedrock produce the majority of the overland flow via saturation excess overland flow. Due to mostly low precipitation intensity in the Cannonsville watershed, a patchwork of *B1* and *B2* land types exists with occasional seasonal shifts to the *A* land type during bursts of extremely high intensity precipitation (Table 2.5 and Figure 2.3) and frozen soils at the end of the winter.

SRP loading from dairy cattle manure was primarily mitigated through targeting HSAs. Exclusionary fencing and cattle crossings were placed in the near stream areas, and timing and placement of fertilizer and manure application were considered throughout the watershed (Rao *et al.*, 2009; Flores-Lopez *et al.*, 2010) by preventing manure spreading in HSAs. In order to make this possible, paved paths were constructed upslope and more powerful tractors were cost-shared that could pull the manure up the hill away from the saturated areas at the bottom of the watershed where farms were located because of the availability of drinking water for the cattle.

Using the Variable Source Loading Function model (VSLF) (Schneiderman *et al.*, 2007), Rao *et al.* (2009) analyzed the effectiveness of various BMPs. The authors concluded that total P losses decreased only after installing cattle crossings in the creek, protecting riparian areas, reducing the spreading of manure during hydrologically active periods, and excluding livestock from the stream. This suite of targeted NMPs resulted in the largest SRP reductions (Bishop *et al.*, 2005; Easton *et al.,* 2008) in the *B1* land type, which is the land type in this watershed that typically contributes the most to pollutant loss through surface flow paths.

4.2.4 Little River Experimental Watershed, GA

Because of extensive, long-term research in the Little River Experimental Watershed (LREW) (33,700 ha) in southern Georgia, climate and spatial distribution of soil types and depths are well known (Vellidis *et al.*, 2002) making application of the land-type based BMP recommendations relatively simple. The infiltration capacity of soils in the watershed is generally high, and thus the *B* land types are most prevalent (Table 2.5), although high intensity precipitation during summer thunderstorms also occurs (Figure 2.3). In riparian zones and toe slopes, the land type is *B2* because deep, well-draining soils are present. Shallower soils dominate upland areas which are mostly classified as *B1*. If farming were to occur in the riparian zones and toe slopes, controlled drainage, cover crops and crop rotations would need to be considered. However, because of the hydric properties of those soils, farming is not common and wetlands and forested buffers are highly recommended. Furthermore, the natural buffer strips close to the streams were indeed very effective because in this watershed, nutrient transport was very small (Cho *et al.*, 2010). In the upland areas, NMPs are recommended in order to control for the amount of fertilizers or pesticides that can be lost through surface transport pathways (e.g. runoff, erosion).

4.2.5 Consistency between Land Type Classification and Literature Recommendations at Watershed Scale

Implementation of BMPs at the watershed scale has the potential to achieve larger water quality impacts if the BMPs are targeted at the correct locations based on a thorough understanding of the local physical and climatic characteristics. At the watershed scale, it is especially important to understand the land types present at smaller spatial scales and how to correctly target BMPs within the watershed. With the proper information, land type based recommendations for targeting dominant flow paths within a watershed can be accomplished and lead to substantial improvements in water quality, as observed in the Paradise Creek watershed (Brooks *et al.*, 2010). In watersheds where less is known about the physical characteristics, land type identification, creation of BMP recommendations, and their subsequent implementation based on locally dominant flow paths may require an upfront time investment, but if the BMP implementation is designed correctly, water quality improvements may be achieved.

Temporal shifts in land type characteristics were not often evaluated in watershed studies. Baseline monitoring should occur before BMP implementation, and long-term, event-based monitoring should continue long after installation to determine effectiveness and account for the potential lag time of pollutant response. Our recommendations tie into those by Mulla *et al.* (2005) with regards to the ability of BMP effectiveness studies at the watershed scale to produce clear results. They need: 1) to be long enough to account for weather variability or lag time; 2) the study design to be scientifically rigorous; 3) the BMPs targeted at HSAs; and 4.) modeling efforts representative of actual physical processes.

Lag time of pollutant response when monitoring BMP effectiveness is often mentioned at the watershed scale. Many authors of watershed scale studies discussed the impact of lag time on results (e.g. Boesch *et al.*, 2001; Schilling and Spooner, 2006; Rao *et al.*, 2009; Brooks *et al.*, 2010; Gassman *et al.*, 2010). Study length may not be sufficient to measure the desired response (if any) or there may be too much variability in the results to meaningfully quantify the BMP impact. Studies and monitoring projects may also be abandoned before significant changes appear in monitoring data (Meals *et al.*, 2010; Hamilton *et al.*, 2011). In order to avoid that problem, study design needs to account for the response period by either lengthening monitoring time, decreasing scale by choosing a nested basin, or improving the statistical design (e.g. paired watersheds). Lag times are not constant across watersheds, varying based on watershed size, hydrology, pollutant type, BMP and stream characteristics (Meals *et al.*, 2010). However, estimations can be made in order to design more effective studies. Gregory *et al.* (2007) described the timing between conservation, restoration actions and ecological responses. They also recommended taking a more synergistic approach to watershed management beyond just the agricultural system, through incorporating ecological restoration and community collaboration to enhance biological responses in addition to improved water quality and decreasing lag time. Meals *et al.* (2010) compiled a list of reported lag times for different watersheds.

5. Conclusions and Recommendations

Our conceptual framework for analysis of NPS pollution in agriculturally dominated watersheds focuses on classification of land types (*A*, *B1*, *B2*) based on climate, soil type, land use, and topography. We sought to 1) develop a conceptual framework that relates BMP effectiveness with dominant hydrological flow paths that are a consequence of land and climate characteristics; 2) determine how BMP effectiveness reported in past plot and field scale studies fit within the framework; and 3) review BMP effectiveness in watershed scale studies and how cumulative effects of BMPs at this scale can be explained within the conceptual framework.

Our conceptual framework is centered on three hydrologic land types. Hydrologic land type *A,* where the restrictive layer is at the surface and land management practices that increase infiltration are effective. Hydrologic land type *B1*, where the surface soil has an infiltration rate greater than the prevailing rainfall intensity, but there is a restrictive layer at some depth causing lateral flow and saturated areas where water storage is limited and the profile cannot carry the flow from upslope. Few structural practices are effective for these soils. Hydrologic land type *B2,* where infiltration rate is greater than rainfall intensity (as with *B1*), but the profile lacks restrictive layers and the dominant flow path is percolation.

For each land type, effective BMPs were selected through a literature review analysis of plot and field scale studies based on the dominant hydrologic and pollutant flow paths, while taking into account the variability of land type characteristics in space and time. The key findings from the plot and field scale analysis showed that source BMPs such as NMPs and IPMs can be very effective at reducing pollutant delivery to surface and groundwater, independent of hydrologic land type. Conservation tillage (Blevins *et al.*, 1998) and CRP have been widely successful across landscapes at minimizing nonpoint source pollution (Hansen, 2007) and in converting *A* land types with sediment loading problems to *B1* and *B2* land types with increased infiltration capacity. Caution must be taken in buffer implementation because they may not be effective, especially in *B1* land types or in areas with concentrated flow.

We demonstrated that the conceptual framework could be applied at the watershed scale through an analysis of four data-rich watershed case studies. The inherent lag times within the social and physical system can disguise the actual effectiveness of a BMP or watershed scale suite of BMPs. Hydrologic land types can shift in space and time, and modifications from BMPs can alter the dominant flow path, triggering a new transport path for pollutants. Optimal NPS reduction at the watershed scale results when suites of BMPs address application of pollutants, transport of those pollutants based on the dominant flow paths, and delivery to the stream through effectively placed buffers or wetlands.

While there is an immense breadth of BMP effectiveness studies, it can be difficult to translate how a BMP will perform on a new landscape. Other limitations in current studies include limited monitoring period duration, lack of published studies that report when a BMP is not effective, and a large body of literature that includes empirical modeling as an analysis of effectiveness.

Transferring lessons learned from empirical modeling studies was particularly difficult because model results did not provide specific physical characteristics.

We recommend that future studies characterize climate, soil type, land use, topography, and the dominant flow paths through a process-based analysis. While there are many studies of BMP effectiveness at the watershed scale, they generally were only valid for the region in which they were tested. Our conceptual "land type" framework provides the opportunity to apply and transfer BMP effectiveness results to landscapes with similar characteristics.

In order to reduce NPS pollution, our conceptual framework and literature review emphasize the need to address both the application of pollutants (i.e. amount and timing) as well as the dominant flow path(s) that transport the pollutants. Our land type framework provides a qualitative understanding of when and where to apply agrichemicals and fertilizers and how BMPs interact with the dominant flow path. Understanding of quantitative effects of BMPs on pollutant transport for the different land types will require a process-based decision-support tool that utilizes readily available data. An example is the hydrologic characterization tool by Brooks *et al.* (*this issue),* which has simplified process-based modeling in a web-based environment. Such a tool advances the land type approach to include effects of site-specific spatial and temporal variability, such as slope configuration, seasonal patterns, and comparison of management scenarios, while also modeling adsorption and degradation traits of specific pollutants.

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Chapter 3: Understanding Success: A Case Study Analysis of Improved Water Quality and the Role of Collaborative Stakeholder Groups

1. Abstract

Stabilizing regulation of point sources in the U.S. and the subsequent water quality improvements during recent decades has shifted the focus for mitigating water pollution to nonpoint sources (NPS). The Total Maximum Daily Load (TMDL) is the section of the Clean Water Act that focuses on NPS. Within watershed management, NPS mitigation efforts require voluntary action and thus collaborative stakeholder groups play a key role. In this study, we conducted a multi-case analysis of collaborations in three watersheds in Oregon and Idaho where quantifiable water quality improvements were observed. The objective was to understand what they perceived as reasons for success, and how impacts of their conservation actions can be measured. While the findings reveal complexities involved in assessing whether best management practices add up to measurable success in a watershed, monitoring and data collection was seen as the ultimate measure of success. Participants reflected on how success was achieved and explained related factors and constraints such as availability of funding and resources, lag time of pollutant response, and the inability to attribute successes to a specific entity or action. Interviewees expressed that the TMDL process is controversial and needs to be more transparent. The results support previous findings that communication, trust, and respect are characteristics that facilitate collaboration.

2. Introduction

Surface water quality in the United States has drastically improved since the establishment of the Clean Water Act (CWA) in 1972 (Smith *et al.*, 1987; and Froemke, 2012). The regulation of point sources through the National Pollution Discharge Elimination System (NPDES) provided an opportunity for substantial early progress (Boyd, 2000). The focus has since shifted to tackling nonpoint source pollution which requires more coordination due to the diversity of activities and the absence of regulation (Cabrera-Stagno, 2007). Several lawsuits against the Environmental Protection Agency (EPA) in the 1980s and 1990s spurred widespread implementation of Section 303(d) of the CWA (33 U.S.C. §1251 et seq., 1972) (Boyd, 2000; Koontz and Johnson, 2004) which requires states to list impaired waters and implement the Total Maximum Daily Load (TMDL).

Because of the unregulated nature of nonpoint sources, voluntary efforts are needed to mitigate pollution (Boyd, 2000). The watershed movement took hold in the 1990s, arising from this need to encourage diverse stakeholder participation to improve water quality (Kenney, 1997), which has resulted in devolving some power to the local level (Leach and Pelkey, 2001). This includes a paradigm shift from a top-down to a bottom-up management approach (Griffin, 1999). Focusing water quality management efforts at the watershed scale encourages cooperation between individuals and entities at a scale that makes hydrologic sense because upstream actions impact downstream locations (Haith, 2003). As John Wesley Powell suggested in 1890, common interests and values are defined by hydrographic basins. This return to the watershed approach is a response to the failure of larger scale methods of governance and management (McKinney *et al.*, 2002), and political boundaries that do not always align with hydrologic boundaries.

The EPA defines the watershed approach as a coordination of public and private efforts to target the highest priority issues within a watershed (Haith, 2003). In order to do so, collaborative stakeholder partnerships of public and private interests have developed to define problems, set goals, and implement projects (Leach *et al.*, 2002; Koontz and Johnson, 2004). These partnerships can take on many different forms (Chaffin *et al.*, 2012). Moore and Koontz (2003) proposed a typology of stakeholder groups (agency-based, citizen-based, and mixed), each of which fills a different niche based on the unique water quality issues and composition of each watershed.

Analysis of the impacts of collaborative groups is often gauged by *perceived success* because of the difficulties associated with measuring improvements in watershed health. Perceived success is a proxy for measuring physical and ecological improvements in the watershed when there is a lack of reliable objective data (Leach *et al.*, 2002; Conley and Moote, 2003; Dakins *et al.*, 2005; Chaffin *et al.*, 2012). While perceived success can be useful, it is not a direct measure of water quality improvement. It is a measure of other factors that are assumed to positively influence water quality. In this work, we attempt to go beyond perceived success to identify watersheds in which water quality improvements have been scientifically and quantifiably shown then analyze those watersheds in depth to identify characteristics that may have led to water quality improvements. We define success as measured water quality improvements at any level, not strictly achievement of TMDL targets. For example, in the Bear Creek Watershed in southern Oregon, phosphorus levels have been reduced significantly, but not to a point of consistently attaining the TMDL target (USEPA, 2011b). We recognize that limitations to this definition of success are the inherent lag time in ambient water quality improvements (Kenney, 1997; Selin *et al.*, 2010; Koontz and Johnson, 2004), especially at the watershed scale; the uncertainty in attributing improvements to specific entities or actions; and the lack of resources or knowledge to perform proper monitoring (Leach *et al.*, 2002; Benham *et al.*, 2008).

While it is clear that some level of success in improving water quality has been achieved, we cannot assert causation between the watershed groups and the water quality improvements. Therefore, we seek to understand the dynamics and characteristics present in those watersheds and in the role of the collaborative groups through a multiple-case analysis of watersheds in Oregon and Idaho that have observed some level of improved water quality. Our specific objectives are to, (1) identify stakeholder motivation for participating in collaborative groups; (2) determine what "success" means to participants in watershed collaborations; and (3) evaluate stakeholder perceptions of the TMDL process.

3. Methodology

3.1 Selection of Case Study Watersheds

Our approach was an exploratory, embedded multiple case analysis. We triangulated multiple sources of evidence (Yin, 1994) through a systematic multi-level elimination process to identify case study watersheds with water quality improvements and an active watershed council or Watershed Advisory Group (WAG). This included an extensive review of primary and grey literature as well as direct communication with Idaho State Department of Environmental Quality (IDEQ) personnel. We reviewed documents from the Oregon Watershed Enhancement Board (OWEB 2009; 2011) that provided brief descriptions of each council's activities to eliminate watershed councils that were clearly outside the scope of the research; this reduced the number of potential watersheds from 75 to 50. Then we perused watershed council websites to find more in depth information, including council activity, stakeholder participation, and availability of data and reports, further reducing the number of watersheds to seven. Finally, we compared the watersheds remaining in our pool against the proven improvements noted in EPA Nonpoint Source Success Stories (USEPA, 2013).

Idaho watersheds required a less systematic method for identification of case study candidates because accessible information was not as readily organized and available as that provided by OWEB. For example, most watersheds in Idaho do not have their own website. The IDEQ website of WAGs in Idaho (IDEQ, 2013a) provided some information, however, not all were listed, and the details needed to perform the same elimination process as in Oregon were missing. We also found details of specific TMDLs on the IDEQ website (IDEQ, 2013b), but the level of information was inconsistent between each watershed and thus difficult to use. The EPA Nonpoint Source Success

Stories (USEPA, 2013) were helpful in identifying a few watersheds with known water quality improvements. To confirm the success of those watersheds and to find others, we directly contacted IDEQ.

We selected Bear Creek Watershed in the Rogue Basin of southern Oregon, North Fork Coeur d'Alene (CdA) Watershed in northern Idaho, and the Middle Snake River (Mid-Snake) Watershed in southern Idaho as case studies for this analysis (Fig. 3.1). We intended to hold constant the basic characteristics (i.e., land use, land ownership, watershed size) of selected watersheds to allow for more clear discernment of the factors that led to successful reductions. However, we were unable to identify watersheds with water quality improvements and characteristics that matched well enough to consider them comparatively as a scientific control. Therefore, the resulting analysis is of three very different watersheds.

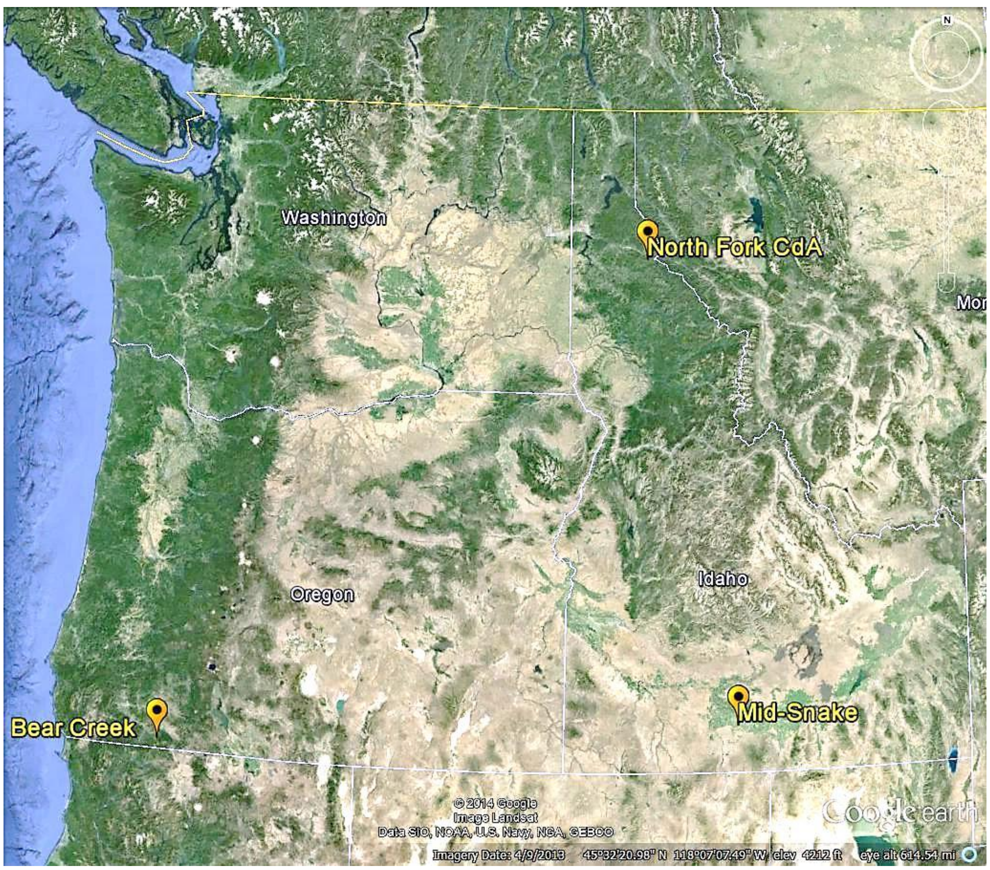

Fig. 3.1 Map of case study watersheds: Bear Creek, OR; Middle Snake River, ID; North Fork Coeur d'Alene River, ID.

3.2 Interviews and Analysis

We conducted one-on-one interviews with watershed councils, WAGs, regional and state DEQ personnel, and other stakeholders in the selected watersheds to provide insight and a diversity of perspectives to the analysis (Table 3.1). The University of Idaho Institutional Review Board certified this project as exempt (13-277) and participants agreed to disclose watersheds, groups and other entities while keeping individual identities confidential. We conducted one to two hour phone or in-person interviews with each interviewee using semi-structured interview questions (Morse and Richards, 2002) based on six themes: (1) watershed group formation; (2) leadership and collaboration in the watershed; (3) stakeholder participation; (4) water quality; (5) education and outreach; and (6) funding (see Appendix B for list of questions). The first point of contact in each watershed was the watershed coordinator, which in Idaho is a DEQ employee and in Oregon is part of the watershed council. At the end of each interview, we employed snowball surveying procedures and asked the interviewee which other stakeholders should be interviewed. With each additional interview, we followed the same general themes as the original interviews, but also sought elaboration on topics specific to that stakeholder. Our goal was four to five interviews in each watershed until we reached a saturation of themes (Morse and Richards, 2002) across a representation of several perspectives (Table 3.1). We also attended a North Fork CdA WAG meeting to observe the process and identify interview participants.

Table 3.1 Interviews conducted in study watersheds. Some interviewees spanned multiple stakeholder groups and/or watersheds, while some stakeholder groups included multiple interviewees. A total of 14 interviews were conducted. DEQ = Department of Environmental Quality.

Watershed (Region)	Stakeholder Group
Bear Creek (Rogue Basin, Southern OR)	Watershed Council Regional DEQ Irrigation Education/Outreach Municipal Water Utilities
Middle Snake River (Twin Falls, Southern, ID)	Regional DEQ State DEQ Environmental Group Aquaculture
North Fork Coeur d'Alene (Idaho Panhandle)	Regional DEQ State DEQ Resource Extraction Federal Agency Water-based Recreation Environmental Group

We recorded interviews with a digital voice recorder and personally transcribed each interview. Statements made by interviewees were triangulated with data, documentation, and other available sources of evidence (Yin, 1994). Following grounded theory, we coded data based on theoretical ideas and hypotheses from the literature and themes that emerged during the interviews (Schwandt, 2001; Charmaz, 2006). A list of codes used can be found in Appendix C. Coding and analysis was performed in the NVivo computer program.

4. Watershed Profiles and Role of Collaborative Groups

4.1 Bear Creek Watershed, Oregon

The Bear Creek Watershed (93,500 ha) in Jackson County, OR is part of the Rogue River Basin (ODEQ, 2007). Most of Jackson County's approximately 200,000 residents live within the Bear Creek Watershed. Land uses are mixed with 46% forest, 35% agriculture, and 18% urban. Public land ownership is at approximately 21%. Point sources in the watershed include the Ashland Wastewater Treatment Plant (WWTP) and other smaller NPDES permit holders. Nonpoint source Designated Management Authorities (DMAs) listed in the TMDL are the cities of Ashland, Talent, Phoenix, Medford, Central Point, and Jacksonville, Jackson County, Oregon Department of Agriculture, Oregon Department of Forestry, United States Forest Service (USFS), Bureau of Land Management (BLM), Oregon Department of Transportation, Emigrant Dam, and three irrigation districts. Bear Creek was placed on the 303(d) list in 1987; TMDL targets for total phosphorus, ammonia nitrogen, and biochemical oxygen demand were set in 1992, and for bacteria, temperature, and sedimentation in 2007 (ODEQ 2007; 2012). Several beneficial uses are present in the watershed, including cold water biota, domestic and industrial water supply, irrigation, livestock watering, primary and secondary contact recreation, wildlife and hunting, and hydropower. Bear Creek is not at the point of TMDL de-listing, but recognition of water quality improvements in the watershed have been made with an EPA Nonpoint Source Success Story for phosphorus reductions (USEPA, 2011b. Water quality monitoring in the Bear Creek watershed has occurred since the 1960s by the Rogue Valley Council of Governments, the extent of which increased after TMDL approval (RVCOG, 2005).

Three separate collaborative groups within Bear Creek work to improve watershed health: Bear Creek Watershed Council (BCWC); the Oregon Department of Environmental Quality (ODEQ) TMDL group; and the Water for Irrigation, Streams and Economy (WISE) Project. BCWC began as an ad-hoc group in 1993 (BCWC, 2011). In that same year, House Bill 2215 was passed by the Oregon legislature which called for the formation of voluntary, nonregulatory watershed councils which would receive some state funding (Dakins *et al.*, 2005). In 2007, BCWC gained nonprofit 501(c)3 status (BCWC, 2011). BCWC is a mixed group with both public and private interests, but with the board mainly composed of agency employees. According to participants and the BCWC website, its main functions are to bridge communications, provide a space for networking, perform education and outreach, and implement riparian restoration projects in order to reach its goals of fostering collaboration and protecting the watershed.

Bear Creek TMDL group meetings, facilitated by the ODEQ, are attended by DMAs and function within a regulatory environment with a goal of meeting TMDL targets. The TMDL group has been meeting since 1992, at which time meetings were contentious, according to participants, but in the last five years, the group "clicks" and the ODEQ has used it as a model for the Rogue Basin group. TMDL group meetings offer a space for communication, idea generation, and providing updates. The current focus is facilitating project implementation through collaboration. Some funding is also provided and can be used to incentivize participation. Additionally, there is information exchange between BCWC and the TMDL group.

The WISE Project is a regional collaboration focused on increasing stream flows and improving water quality and habitat through enhanced irrigation efficiency (WISE, n.d.). Once implemented, WISE will lead to large-scale conversion from flood to pressurized irrigation, which interviewees described as more efficient and stated that it reduces return flows thereby improving water quality. Additionally, irrigation pipes will be installed to eliminate delivery of irrigation water through tributaries and leave natural flows in the streams. All three groups in Bear Creek bring together a variety of partners and stakeholders. Group leaders coordinate as much as possible in order to not overwhelm stakeholders, but also see the value in overlapping participation.

4.2 Middle Snake River Watershed, Idaho

The Middle Snake (Mid-Snake) Watershed (651,000 ha, >100,000 residents) includes the cities of Twin Falls, Kimberly, Hansen, Filer, Buhl, Jerome and Wendell, ID, among other smaller towns (IDEQ, 2010). Agriculture and range land dominate the watershed, at 56% and 42%, respectively; the remaining land is urban, riparian/wetlands, and forest. Fifty-four percent of land is private, 46% public, and less than 1% is tribal. The TMDL for the watershed (Upper Snake Rock TMDL) was written in 2000 and modified in 2005 (IDEQ, 2005). It identified sediment, total phosphorus, pathogens, ammonia, pesticides, and oil and grease as the pollutants of concern. Beneficial uses present in the watershed are cold water biota, salmonid spawning, agricultural water supply, and primary and secondary contact recreation. Dozens of permitted point sources exist in the watershed, such as cities and aquaculture facilities. Nonpoint sources, such as agriculture, irrigation returns, ranching, dairies, and stormwater, are also present. According to IDEQ study participants, sediment loading in the watershed has decreased and meets TMDL targets 97% of the time. Phosphorus levels have declined in some locations and from some sources, but overall, little improvement has been made and nuisance aquatic macrophytes continue to plague the waterways (IDEQ, 2010). IDEQ employees stated that monthly monitoring has occurred on the river for 15 years, as well as smaller-scale monitoring focused on projects. However, funding reductions have cut back monitoring efforts.

The history of collaboration in the Mid-Snake watershed, as discussed by interview participants, began in 1983-84 with the Mid-Snake Study Group, an informal working group with public information meetings focused on water quality issues. The study group transitioned into the WAG after the "WAG/BAG law" passed in 1995 (Idaho Code 39-3615) created watershed advisory groups and basin advisory groups in all of Idaho. The Mid-Snake WAG is led by regional IDEQ employees and comprised of individuals representing relevant stakeholder groups in the watershed. The chairman is elected by the WAG. The function of the Mid-Snake WAG is to provide input and advice to IDEQ in an effort to attain TMDL targets. The list of participating stakeholders is extensive and spans the public and private sectors present in the watershed such as agencies, irrigation districts, aquaculture facilities, farmers, ranchers, canal companies, and cities. However, one participant observed that the earlier grassroots efforts were more diverse and more functional than the current top-down approach, which has been echoed in the literature (Agrawal and Gibson, 1999). One problem cited by participants in the Mid-Snake has been that some of the stakeholder groups or entities participating in the WAG send lawyers as their representatives rather than individuals that work more directly with water quality issues and projects.

4.3 North Fork Coeur d'Alene Watershed, Idaho

The North Fork Coeur d'Alene (North Fork CdA) Watershed (231,800 ha, <2,000 residents) is nearly all forested, with 94% of land managed by the USFS (IDEQ, 2013c). The remaining 6% is managed by private landowners, the state of Idaho, and the BLM, with land uses of agriculture, silviculture, recreation, residential use, and mining. No urban areas exist in the watershed; Murray, ID is the only population center and has fewer than 600 residents. Despite the small permanent population, recreation impacts are strong due to summer river activities (USFS, 2012). Beneficial uses in the watershed are cold water biota and salmonid spawning. Sediment, temperature, and heavy metals are the pollutants of concern. A sediment TMDL was written in 2001 (IDEQ, 2001) and a temperature TMDL in 2013 (IDEQ, 2013c). No major point sources are present in the North Fork CdA. Nonpoint sources include road encroachment, removal of riparian vegetation, recreation, and legacy sediments from mining and timber. Large reductions in sediment loads have been made in Yellowdog, Tepee, and Steamboat Creeks, which are nested watersheds of the North Fork CdA. Yellowdog Creek was recognized as an EPA Nonpoint Source Success Story (USEPA 2011a).

Interview participants explained that informal collaborative efforts began with the Coeur d'Alene River Preservation Committee, which was comprised mainly of private landowners that sought to target recreation issues. In 2000, a predominately agency-based partnership developed, led by the USFS, to reduce sediment loads in Yellowdog Creek through road decommissioning. In 2001, a de facto WAG formed with the sediment TMDL. A second WAG formed in 2007, coinciding with the temperature TMDL and the sediment TMDL five-year review. Current stakeholders include the USFS, Shoshone County, state agencies, mining, private landowners, water-based recreation, and environmental groups. The main function of the current WAG is to carry out administrative projects (e.g., writing reports), but it also writes grants, implements projects and seeks to conduct monitoring.

5. Factors Influencing Collaboration

Each study watershed is unique in its land use and land ownership, size, climate, population, pollutant sources, and stakeholder types. Despite those differences, commonalities across the collaborative groups are present. Each watershed has at least one group (Bear Creek has multiple) that formed early in the watershed movement (1980s – 1990s), either before it was encouraged or required by the state, or at the beginning of state involvement. Therefore, these collaborations have had time to overcome barriers and evolve into better functioning groups. Other common themes that emerged during this research included catalyzing events; funding, which connects to time and capacity; education and outreach efforts; communication, leadership, and respect; and participation of diverse stakeholders. When those factors were absent, as well as when politics got in the way or a general understanding of the importance of water quality efforts was not present, participants cited collaboration as being difficult.

5.1 Catalyzing Events

Some participants discussed catalysts that clearly led to collaborative efforts within each watershed. In Bear Creek, the WISE Project was partly catalyzed by a desire to avoid the problems that occurred in the Klamath Basin in 2001 when drought hit and water allocation became contentious (Levy, 2003). Founding members of WISE felt that if diverse stakeholders participated from the beginning, they would gain a sense of ownership and be more apt to collaborate. Participation in planning stages has also proved effective in other locations (Duram and Brown, 1999).

Several participants discussed two events that catalyzed the formation of the Mid-Snake WAG. The first was in June 1992 when then Governor Cecil Andrus' jet boat tour of the Snake was halted as algae clogged the boat (The Spokesman-Review, "Andrus' jet boat waylaid by algae plaguing Snake," 28 June 1992). The second was a lawsuit filed by the Idaho Sportsmen's Coalition and the Idaho Conservation League against the EPA for failing to uphold the CWA because not enough streams were listed as water quality limited on the 303(d) list (Idaho Sportsmen's Coalition v. Browner, 1996). According to IDEQ employees, "As a result of that lawsuit … and also because of that event that occurred on the Mid-Snake, the legislature in Idaho… created what is called the Idaho WAG/BAG law." Therefore, the catalysts for forming the Mid-Snake WAG were also catalysts across the state of Idaho. The lawsuit was also mentioned by participants in the North Fork WAG.

5.2 Collaboration Themes

Several themes, such as communication, trust, leadership, and diverse participation, were highlighted within each watershed as fostering collaboration. Nearly all participants discussed the role of communication in strengthening watershed groups. Through communication, stakeholders build trust in each other and in group leadership (Breetz *et al.*, 2005). In both Idaho watersheds, a lack of trust in the data and in the process has been a barrier to collaboration at some point in the groups' histories. Distrust in the data brings about doubt in the process and in those implementing the process. In that sense, trust can be built through a more transparent process in which stakeholders understand how targets and load and wasteload allocations (LAs and WLAs) are set and know that high quality data are being used. Associated with trust, interviewees in the Mid-Snake and the North Fork CdA noted that respect and equality are essential components of successfully working with others. When respect is present, people are more willing to compromise and find common ground, which was stated by participants in each watershed as being necessary. In a study of 30 collaborative groups in the U.S., Selin *et al.* (2010) also found that willingness to compromise was a characteristic of effective groups. One individual in the Mid-Snake said, "People who have interest in the watershed but don't necessarily get along come together in collaborative efforts and they seem to get more done because they end up checking their swords and cell phones at the door." Strong, effective leadership is essential to successful groups (Leach and Pelkey, 2001; McKinney *et al.*, 2002;

Benham *et al.*, 2008; Selin *et al.*, 2010; Chaffin *et al.*, 2012); it can help facilitate the respect and communication in the group and thus lead to trust and equality. Without leadership, collaboration may fail, as stated by one individual in Bear Creek: “The downfall of collaboration is, in my opinion, not having a central leadership – not having somebody or some entity that’s more or less independent that guides it.”

Participation by a diverse group of stakeholders was seen in each watershed as a benefit to collaboration and as an aid to reaching group goals, as found in other studies (Chess *et al.*, 2000; McKinney *et al.*, 2002; Selin *et al.*, 2010). One participant in Bear Creek stated, “Because of water, everyone’s a stakeholder, whether they know it or not.” In each watershed, others noted that collaboration is more effective than regulation. Each stakeholder brings in his or her own unique perspective that contributes to the whole. As group members buy into the goals of the group, they become “portals to stakeholders. They’re a way to get the word out.” It was recognized in one group in Bear Creek, however, that they are always talking to themselves. While the group may be diverse, it is still mainly comprised of people who already understand the issues. Furthermore, while having diverse perspectives at the table can strengthen the collaboration, it is also seen as a challenge because it adds more layers of bureaucracy, requires too much time, or shifts attention from important problems (Mullner *et al.*, 2001; Leach *et al.*, 2002). In Bear Creek, one interviewee said, “The different perspectives and the way people look at things – that’s what makes it hard to collaborate! I used to think collaboration was the greatest thing until I actually started trying to do it.” Despite such a strong view, that person also recognized achievements made through collaboration if the right conditions are present. Others recognized the logistical difficulties in trying to work with a diversity of individuals and groups. Scheduling conflicts were consistently mentioned. To include the general public at meetings, they should be held in the evening, but to include agency personnel and DMA representatives, they need to be held during the work day. Therefore, something as simple as meeting time can be a barrier to achieving broad stakeholder representation within a group, which was also observed by Bonnell and Koontz (2007). Additionally, when a watershed is made up of several small private landowners, such as in Bear Creek, it is difficult to carry out large scale projects because buy in is needed from several individuals.

5.3 Funding

Funding was the most frequently coded theme as it was identified by all participants as a barrier to project implementation. However, when present, it can provide motivation for collaboration.

Funding and the availability of resources in general have been cited as components of success (Leach and Pelkey, 2001; McKinney et al, 2002; Benham *et al.*, 2008). Many sources of funding were discussed, including federal agencies (EPA, Bureau of Reclamation, and United States Department of Agriculture), state and local grants, foundations, and projects themselves. 319 grants from the EPA are a common source of funding for watershed groups, but those funds have decreased in the past five years. Both DEQ and USFS budgets have been cut, limiting the number of projects and the extent of monitoring that can take place. Projects in the North Fork CdA have often been funded by timber sale receipts. Due to reduced logging and the controversy associated with it, this source of funding has also been reduced. However, a common view expressed between competing interests in the North Fork CdA was the need to generate some money off of the land in order to fund restoration projects.

In Oregon, OWEB provides grant funding which comes from Oregon Lottery revenue (constitutionally dedicated 7.5% of lottery proceeds) (Network of Oregon Watershed Councils, 2011). However, due to the increasing costs of sustaining councils and funding restoration, OWEB is encouraging watershed councils to consider scaling up and merging efforts at a larger geographic scale (OWEB, 2014). BCWC is analyzing the possibility of merging with other watershed councils in the Rogue Basin. While concern was expressed with respect to the potential loss of power and identity, increased organizational capacity could be an advantage (Bonnell and Koontz, 2007). Multiple interviewees felt the merger would be beneficial because it would increase capacity to do more projects, both restoration and monitoring, and receive larger grants; the scale of the group would be more realistic; and staffing could be diversified.

In Bear Creek, two projects were discussed as being able to generate their own revenue. First, thermal credit trading can fund riparian shading projects as a more cost efficient method to reduce stream temperatures than upgrades at wastewater treatment plants. Second, the pressurized irrigation system in the WISE Project will generate hydropower which can be sold. Participants expect that drastic improvements to water quality after WISE is implemented will help bring in more funding.

Partnerships are especially beneficial when funding is limited. They can be used to leverage resources between entities as well as to access funds from different sources. For example, in Bear Creek, the Medford Water Commission, rather than implementing their own projects, provides small grants to other groups to fund water quality projects (MWC, 2009). One participant explained that funding agencies are more likely to award grants to diverse partnerships because it shows “the other side" is already involved and disagreements are less likely, which was also discussed in Leach *et al.* (2002). Another method to leverage funds is partnering with a nonprofit group to apply for foundation grants, such as what the North Fork WAG and the North Idaho Flycasters have done. On its own, the WAG would not be able to access such grants.

Because of limited funding, which most participants believed was not enough to carry out all necessary projects, time and effort must be placed where it is most productive, according to one participant in Bear Creek. The watershed groups all have projects ready to implement, but lack the funds to do so. Interviewees in both Idaho watersheds discussed the need for the IDEQ to balance economics and water quality. One respondent said that because Idaho is a conservative state, they need to "stay in the black" to avoid a "bad feeling" in the citizenry. Across the three watersheds, participants agreed that there is not enough funding for monitoring, project implementation, and staffing.

5.4 Education and Outreach

Education and outreach with respect to watershed issues has multiple meanings. Participants discussed education and outreach in terms of various groups: K-12 students, general adult population, stakeholders/water users, and collaborative group members. It can then be conducted in a variety of ways. For example, in Bear Creek, the Oregon State University Extension Service is the primary entity working toward K-12 education. The Bear Creek Watershed Council reaches out to youth through field activities, but also targets the adult population through public education media campaigns focused mainly on storm drain issues and salmon habitat. The TMDL group, WISE, and BCWC educate their own members on more technical aspects through newsletters, meetings, presentations, and field trips; those members then go out and educate their peers. Similar education and outreach techniques were observed in the other study watersheds, although with less of an emphasis on K-12 education, especially in the North Fork CdA where the population is extremely small. In the Mid-Snake, informal education of stakeholders was preferred to formal education.

Increasing public understanding of water issues was discussed as important in improving water quality in each study watershed, and education and outreach were seen as the primary methods to achieve this. Duram and Brown (1999) found that two-way communication, such as education and outreach programs, was perceived as more effective than one-way communication, such as newsletters and advertisements. Additionally, through education and outreach, trust can be built (Breetz *et al.*, 2005) and people are encouraged to participate (Chess *et al.*, 2000). In the North

Fork CdA, one person stated, “Education is probably the number one thing that we need to be doing.” In Bear Creek, a participant observed that “it takes lots of education and change in people’s thought pattern” to implement projects and achieve positive outcomes. The result of such education and time can be seen in the Mid-Snake where “the same farmers, who 20-30 years ago were accepting dirty water, are now accepting only clean water. That’s a change in the way that the perception of water quality is to people that are utilizing that water.” A generational change is also happening. Through K-12 education, children grow up understanding water quality issues. When they are adults, they then consider what they have learned in their decision making.

While education and outreach was recognized as important by nearly all participants, they also emphasized that it is not easy, because it takes time to overcome the ignorance and misconceptions of water topics. In the Mid-Snake, two participants explained that education needs to be targeted not at technical aspects, but at getting people to “acknowledge that there’s a problem” and to “understand their role and what they can do to be a good steward.” Similarly, the North Fork WAG and the USFS work together to increase public understanding of the need for watershed stewardship through the Respect the River Program (USFS, n.d.). Despite a small resident population in the watershed, summer use is extremely high, which can have a large impact on water quality and riparian areas. One WAG member emphasized that people need to understand that “it doesn’t take a lot of money to destroy a watershed but, man, it takes a whole lot of money and time to get it back.”

5.5 Litigation and Environmental Groups

During the interviews, the role of litigation and the participation of environmental groups emerged as a strong theme. As discussed previously, there is a history of litigation related to the TMDL in Idaho (e.g., Idaho Sportsmen’s Coalition v. Browner, 1996). Additionally, in the North Fork CdA, lawsuits have been filed which halted logging projects (e.g., Lands Council v. Powell, 2004). As well, in Bear Creek, standoffs between environmentalists and irrigators in the Klamath Basin hit close to home (Levy, 2003). Some participants felt that litigation was a catalyst for progress and was necessary, while others felt that it was a barrier to collaboration. Several individuals representing each watershed connected environmental groups to the litigious process and discussed their lack of collaboration. One person said, “We get very little participation from environmental groups… Yeah they file lawsuits, but they really don’t want to participate in trying to solve problems.” Other terms often associated with environmental groups were, “sit back and sue,” and a “take-no-prisoners” attitude. While these statements may be true for some environmental groups, especially, for example, in the 1990s when litigation was a common tool used to spur TMDL action (Boyd, 2000), we found that these attitudes can represent misconceptions.

At the time of the interviews, environmental groups were not represented in the North Fork WAG or the Mid-Snake WAG. Interviewees presented three explanations for this absence: (1) they lack the capacity to attend meetings; (2) they do not feel welcome; and (3) they have specific goals they cannot deviate from. A WAG member stated explicitly that the environmental groups simply do not have the time, money, or capacity to attend all WAG meetings throughout the region. An environmental group member echoed that point: "There are a lot of groups and so if it's very far outside of our geographic area it's hard to get to meetings." Since the time of the interviews, an environmental group has joined the North Fork WAG.

Some interviewees thought that the participation of environmental groups in collaborations waned when there was hostility towards them. For example, many characterized the Mid-Snake as a "working river" and, therefore, many of the WAG members may have a different philosophy about natural resources than that of environmental groups. As stated by one participant, "I think [environmental groups do not participate] because the rest of these heavy hitter entities don't help make them feel welcome. They don't waste the time because they figure they'll go after them through litigation and get further." In the Mid-Snake, it was also seen as a capacity issue because many of the environmental groups are located in Boise.

Opposition to consensus often stems from its tendency toward the "lowest common denominator" (Dakins *et al.*, 2005). Environmental groups often have a specific mission which is difficult to uphold when consensus is used (Selin *et al.*, 2010). The unwillingness to compromise by environmental groups can be stated with both a positive and negative connotation:

> I think they also have an agenda in some cases that they can't really move off of too much. They will collaborate, but like on a letter, you sign off on this letter as [the environmental group], and the letter might say 'we support cutting trees in the riparian area.' They would come back and say, 'No, we can't sign off on that.'

Another said:

> I think maybe part of it is they don't like to go to meetings when there are people who don't believe what they believe. I can get on my soapbox that environmentalism is a religion to a large extent. Listening to heresy imperils your soul or something. I've ran into a few of them that I can talk to, but not very many of them.

Speaking with an environmental group representative brought another perspective to the analysis. It was clear that this particular environmental group sought compromise and collaboration. For example, in speaking of the variety of collaborative groups that they participate in, this person stated:

> In all of those we have a set of rules or bylaws and I think in all of those we strive for consensus. But if we don't reach consensus, we have some back up mechanisms. Let's say it was ten groups in favor and one opposed. The group opposed has to try to come up with a solution to resolve it. So what I find is that we generally almost always agree on things.

Furthermore, this participant stated that they prefer collaboration and would only utilize litigation when there are "egregious violations of the law," and that they "are thoughtful about it but not adverse." This person recognized, however, that their approach is not necessarily typical of all environmental groups: "There are some conservation groups who don't want to compromise. I mostly work with forestry conservation groups, and some of them don't agree with us working together with the timber industry. They think we're sell outs. There are some of those groups around still." These varying perspectives highlight the need for communication and respect in order to achieve the shared goal of improved water quality.

In Bear Creek, specifically the WISE Project, the view of environmental groups is different. They are considered a key part of the collaboration. Because of the lessons learned from the Klamath Basin, participation of environmental groups was sought from the beginning. While working with such diverse perspectives was initially difficult, it has proved successful for this group. This shows that when stakeholders with different perspectives are willing to communicate, they can find common ground for collaboration.

6. Stakeholder Motivation

Motivation for participating in collaborative groups is as diverse as the stakeholders themselves, ranging from altruism, personal benefit, and the influence of others (Leach *et al.*, 2002). When asked why stakeholders participate in the WAG, one person responded, "My view is we have a common interest to improve water quality conditions in the Snake River." Another said participation is "pretty much driven by a desire to make things better." That focus on valuing water quality and watershed health was shared by nearly all participants. Similarly, in a study of 44 stakeholder partnerships in California and Washington, Leach *et al.* (2002) found that the motive for participation of more than 80% of respondents across all stakeholder categories was "to improve the watershed." One North

Fork CdA participant stated, “One of my main focuses is fish habitat – cleaning up the sub-basin so we have good spawning and rearing.” In Bear Creek, salmon have returned due to barrier removal projects and improved water quality, which influences participation and care for the stream. But one individual saw the salmon return as a lesser motive than the desire to have creeks that are clean enough for children to play in. Even with common goals, stakeholders have to be willing to work together. A North Fork CdA participant explained that “people come and sit down at the table if they’re willing to sit at the table. They find they have common goals. They want restoration. They want clean water. They want healthy forests.”

In the North Fork CdA, de-listing streams so that more pressing issues can be focused on and problems can be solved was a shared motive. Along those lines, a WAG member explained that his goal was “to encourage the agencies to use better science and to work on the issues that really are a problem,” which has become a common call to agencies across the country (Kemmis, 2002). Additionally, providing leadership was cited as a motive for participation. IDEQ employees noted that some stakeholders are “forward thinking,” especially those in the irrigation sector.

While most participants discussed shared values, they also recognized that financial motives can either overshadow or enhance the motivation to participate in improving water quality. As stated by one interviewee, “Most people don’t mind doing something that’s beneficial as long as it doesn’t cost them a lot.” Others echoed that statement – when funds are available for projects, people are more inclined to participate. Some participants made the connection between water quality and the local economy through agriculture, recreation and tourism. Leach *et al.* (2002) observed that financial motives were stronger for stakeholders in the “resource user” category. DEQ employees have seen that participation increases when agenda items are pertinent to stakeholders: “We have more participation when the TMDL is up for renewal or when a document we are working on is going to impact them in some way. And then we have a full room for a WAG.” One participant cited “self-preservation” as motivation for collaborating. Time was often mentioned as a constraint to participation because stakeholders are “so busy with the things of just trying to live and exist,” as well as participate in other commitments.

Being part of a collaborative group is also motivation in itself. Some participants commented that they simply enjoy the people while another said people feel empowered when there is agreement and they have a voice. However, others found that if the environment is unwelcoming, participation declines. Information sharing and presentations at meetings also give people an opportunity to learn more. In order to further increase participation, understanding of the importance of collaborative watershed groups and water quality in general must increase.

Finally, the influence of others can inspire collaboration and participation. Specifically, peer pressure or social norms within the group can encourage involvement (Chen *et al.*, 2009). The threat of regulation also encourages participation in collaborative efforts. Most interviewees observed that stakeholders would rather work together to develop plans than be mandated to act.

7. Understanding Success

Success has many different meanings, especially when considering the diverse goals of watershed groups and the water quality issues that are unique to each watershed (Conley and Moote, 2003). Measures of success can vary between organizational parameters and stakeholder relationships, projects implemented (on-the-ground or administrative), perceived success, and quantifiable improvements in water quality (Leach *et al.*, 2002; Borisova *et al.*, 2012; Chaffin *et al.*, 2012). They also vary based on who is doing the measuring. In this work, we specifically wanted to understand success in terms of improved watershed health and water quality, as that is an identified need at this point in the watershed movement (Leach *et al.*, 2002; Conley and Moote, 2003; Sabatier *et al.*, 2005). Bear Creek, North Fork CdA, and Mid-Snake Watersheds have all observed some level of water quality improvements in recent decades. During interviews, we asked participants how they personally measure success and if they considered that efforts in their watershed had been successful at improving watershed health. A spectrum showing the results of how study participants measure success can be seen in Figure 3.2.

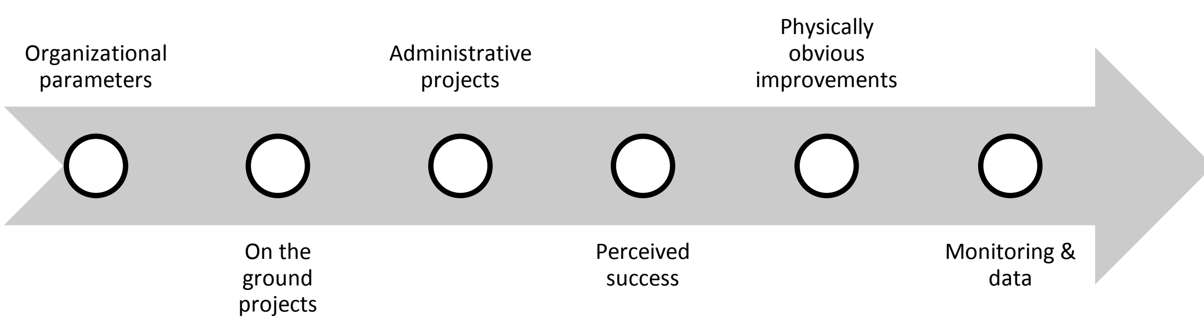

Fig. 3.2 Spectrum of participants' measures of success with frequency of coding increasing from left to right.

Monitoring and data were cited by most participants as the preferred method to measure success if the resources are present to do so. One participant stated, "the importance of monitoring is crucial in being able to understand what's going on and to look for improvement or not." Proper monitoring design is necessary to capture long term trends and show improvement (Benham *et al.*, 2008). Eight participants, representing different watersheds and stakeholder groups, discussed the analysis of long term data to be able to observe trends in water quality. In Bear Creek, one participant described unexpected decreases in temperature which would not have been detected without a properly designed monitoring program. Several others noted that long term monitoring is necessary in order to account for the lag time from project implementation to watershed response. For example, the lag time of different pollutants was discussed in the Mid-Snake:

> The lag time to achieve beneficial uses is maybe more than a lifetime. So that's the challenge there, for this particular kind of TMDL. If we were dealing with toxic things, you could measure that much quicker. But with a nutrient TMDL it's very difficult to measure improvements in a river that's a complex ecosystem.

Another interviewee explained that monitoring design should also include continuous sampling that captures events, not just grab samples that may not be representative of actual conditions. If designed properly, the results of monitoring programs can be used to target projects at locations or pollutants within the watershed that are most problematic in order to optimize success. Various authors suggest using targeting as a method to increase success (Haith *et al.*, 2003; Benham *et al.*, 2008; Nowak, 2011).

Not all participants agreed that monitoring and data analysis were important. One stated that "in the real world, people only care about data in the fact that if they're held to it for regulatory standards… But in terms of just analyzing data for trends and things like that, that's only for academics to do because none of the rest of us has time or money."

While utilization of comprehensive monitoring programs and data collection is a strong, quantifiable method for measuring success, it requires funding and staffing to carry it out. Without sufficient funding and capacity, monitoring may be inconsistent and thus the data may not be useful in measuring success. Lack of funding for monitoring programs was identified as a clear issue in all study watersheds. Many participants that emphasized the importance of monitoring water quality followed with statements of inadequate funding to actually carry it out. One participant said that those providing funds see project implementation as more important than monitoring. Therefore, with limited funds, that is how they are directed. However, this individual then noted the irony in funding priorities: "…everybody's saying, 'We want measureable results.' You can't have measurable results without measuring it… You could say, 'Hey, we have ten projects in and we did this.' But monitoring is crucial and it does need to have a higher priority in the scheme of things."

In Idaho, statewide funding shortfalls in 2009 – 10 halted monitoring. In the Mid-Snake, monitoring ended mid-season. In the North Fork CdA, the IDEQ could not carry out Beneficial Use Reconnaissance Program (BURP) monitoring on its own. Luckily, the EPA provided funding and the USFS provided staffing to collect BURP data. That collaboration allowed them to collect the necessary data for proposing the de-listing of several sub-watersheds. One North Fork CdA participant stated:

> There's no way this project would have been done without this partnership – this monitoring and assessment, de-listing, success stories, there's no way that would've happened without the partnership. All that good work could be done, it could be fully supporting no more sediment impairment, but it wouldn't be properly documented and we wouldn't be able to show that administratively without the partnership. So the fish would know but Congress wouldn't know!

ODEQ in Bear Creek discussed a potential solution to funding limitations for monitoring. In 2011, they hired a consultant to perform a statistical analysis of how much data are needed in order to show a certain level of water quality improvement. Such an analysis could help watershed groups focus monitoring on what is necessary, reduce unnecessary data collection, and thus require less funding.

Several participants in the North Fork CdA emphasized the importance of data quality to be able to properly measure success. At the same time, participants in both Idaho watersheds in various stakeholder groups recognized that there is a mistrust of the IDEQ's data because of a lack of transparency in the process as a whole; therefore, private entities collect their own data to protect their interests and provide what they deem adequate science. This phenomenon is common and has been termed "adversarial science" (Kemmis, 2002) and can hinder collaboration (Benham *et al.*, 2008). Along similar lines, data compatibility was cited as an issue in achieving measurable and recognizable success. According to participants, TMDL de-listing in Idaho is based solely off of BURP data, despite the presence of other water quality data. In the North Fork CdA, extensive data have been collected through the USFS PACFISH/INFISH Biological Opinion program, but they cannot be used because it does not fit into the "black box" for TMDL de-listing. In the Mid-Snake watershed, some nonpoint sources no longer submit data to the DEQ because it is not used.

Attaining WAG goals and TMDL targets, which are often one in the same, was a clear measure of success in the Mid-Snake watershed. Using TMDL targets to measure success allows for a common understanding of goals among stakeholders. A shared definition of objectives is a crucial piece of the watershed approach (Haith, 2003) because it has the potential to reduce disagreements and the use of adversarial science. However, using TMDL targets as the common goal is not always recommended. In all watersheds, statements were made that success is possible without meeting TMDL targets, largely due to the lag time inherent in observing improvements. For example, Bear Creek does not regularly meet targets, but has been recognized for its incremental improvements. Additionally, the disconnect between meeting TMDL water quality targets and attaining beneficial uses was discussed in both Idaho watersheds, and was also seen as a barrier in the literature (Maguire, 2003). In the Mid-Snake watershed, meeting phosphorus targets does not necessarily lead to the absence of nuisance aquatic macrophytes. In the North Fork CdA, one participant stated that TMDL targets must be attainable. This participant also discussed that measuring success is a process, starting with counting projects, then monitoring for water quality improvements, and finally, showing that beneficial uses are met. Following that process allows for stakeholders to feel that progress is being made before the impacts on beneficial uses can be seen. In the North Fork CdA, one participant has observed achievement of that ultimate goal: "Another way to measure water quality improvement is just the fishing. It's awesome… When I first moved up here … fishing was not good. And now the Coeur d'Alene River is probably one of the best blue ribbon streams in north Idaho. You can catch a lot of huge fish."

Administrative projects, specifically de-listing streams from the TMDL, are a measure of success that has been used in the North Fork CdA. Several streams in that watershed are expected to be de-listed in the near future. Many stakeholders feel that that will both encourage continued action as well as allow for the targeting of more problematic areas. De-listing streams is also important for getting the word out about project success and encouraging further involvement. However, two participants in the other watersheds felt that focusing on improvements would lead to satisfaction with being average, and that de-listing is only useful for meeting regulatory standards.

When perceived success was discussed, participants mainly focused on "physically obvious improvements" such that the perception was easy to make and would likely be made by most. These obvious improvements were observed in all study watersheds. In the North Fork CdA, less turbid streams and the presence of large fish marked the progress. In the Mid-Snake, tributaries and riparian habitats have clearly improved because of the efforts by canal companies, such that "even for the non-irrigators, we go to those sites and can appreciate the improvements made." In Bear

Creek, salmon have returned and “huge improvements” have been made because of how impacted the stream had been.

One participant summed up success as a combination of perceived and quantifiable success because “we need real information and some kind of data that shows it,” but at the same time,

> if nobody perceives that there’s success, there’s going to be limited support or poor decision-making, just a lack of understanding of what’s really happening. So there has to be a real understanding of whether there’s an improvement or not based on what, and then most of the time, it seems like those improvements aren’t going to happen without collaboration. I think you really have to have all of those things.

When asked what led to or hindered improvements in water quality, responses were unique for each watershed. In Bear Creek, initial reductions were achieved due to improvements at the Ashland Wastewater Treatment Plant, as well as reducing upland nonpoint source inputs. Municipalities have played an important role in water quality improvements, and irrigation is expected to be the next big step as conversions from flood to sprinkler are made. Streambank stabilization and riparian restoration, driven by a desire for quality fish habitat, have also made a difference in Bear Creek. Time has been important. The collaborative groups in Bear Creek have been working for approximately 20 years, which provides sufficient time to begin to see improvements. Although one participant clearly stated that despite having the time to account for the lag in water quality response, improvements have not been made in the Bear Creek Watershed.

In the Mid-Snake watershed, irrigation was also discussed by most participants as an important component in reducing sediment loads. The two canal companies have been “very forward thinking,” going beyond TMDL targets and requiring that irrigation return flows from their customers meet even stricter limits. To achieve this, sediment ponds and wetlands have been installed. For phosphorus, the aquaculture sector has met the TMDL WLA which requires a 40% reduction. However, overall improvement is slow in coming and nuisance aquatic macrophytes continue to plague the watershed; a few believe that the next step is to use flow manipulation to create scouring flows that will remove the macrophytes. All participants in the Mid-Snake also discussed the difficulty of making improvements for four reasons: (1) voluntary nature of nonpoint source pollution; (2) upstream sources outside the watershed; (3) dams and hydrologic modification; and (4) aquatic macrophytes.

In the North Fork CdA, regulation changes and project implementation led to large reductions in sediment loads generated from an extensive road network due to logging and recreation as well as legacy sediments from mining and logging. Intensive, large-scale, long-term restoration efforts targeted at the most sensitive areas have been implemented, including road de-commissioning and bank erosion projects. However, some see the permitting process as expensive and complicated, and thus a barrier to further improvements.

8. Perception of the TMDL Process

TMDLs guide much of the work done in the study watersheds, as well as in watersheds across the country dealing with water quality issues. While the role that TMDLs play in Oregon and Idaho is different due to state laws, many similar perspectives exist within the three study watersheds. Participants in Bear Creek focused on using the TMDL to put the "weight of the Clean Water Act" on DMAs, but from varying perspectives. On one end of the spectrum was the view that regulation is necessary in order to make people act with water quality in mind. On the other end, the "regulatory backstop" is something to keep in the background and use only when necessary, such as Nowak's (2011) suggestion to thoughtfully use regulation or sanctions on the few that are disproportionately contributing to the degradation of water quality rather than blanket regulations on all. Many participants in the Mid-Snake agreed that if regulation is overused or used improperly, it could lead to litigation rather than collaboration, which stalls the process and leads to conflict. Similar perspectives existed in the North Fork CdA in that regulation is secondary to collaboration but can be useful. Tied into the TMDL, Bear Creek participants discussed that ODEQ presence has made a difference in collaboration and water quality.

The TMDL process was identified by all types of stakeholders as being controversial. Overall, collaborative group leaders, agency employees, and environmental groups commented that TMDLs are effective, but recognized that it can be a contentious process. Individuals representing regulated entities across the study watersheds were opposed to the TMDL for two main reasons. First, they felt that regulation erodes trust, which is a common theme in other watersheds (Breetz *et al.*, 2005). Second, the TMDL process promotes inequality (Maguire, 2003). Specifically, the voluntary participation for nonpoint sources compared to regulation of point sources was viewed as unfair. Despite recognizing that inequality, the regulated entities did not want to force regulation on the nonpoint sources, but instead pointed to collaboration:

> They're just trying to get different groups of people to work together because they realize that the point source people represent less than 50% of what's going into the creek. It was a good start back in 1970 to regulate the point source people, but after they cleaned up their

> act and realized water would only get so clean, they had to figure out ways of going after the people that aren't regulated. You have to be careful if you're trying to regulate everyone, that's not going to go over too well.

Additionally, the determination of WLAs and setting targets was viewed as unequal by some and controversial by all who discussed it. Determining WLAs was likened to allocating a "limited resource" by one individual, which then breeds controversy and finger pointing. Participants questioned if targets are realistic, as well as called for a common understanding of goals in order to minimize disagreement and encourage participation. Individuals representing DEQ, point sources, and environmental groups agreed that collaboration and cooperation can reduce contention and bring entities together, and thus lead to a better TMDL. In the Mid-Snake, one individual explained that WAGs and BAGs were created to alleviate tension between interests by providing a space for discussion and collaboration.

Implementation plans are seen as the step in the TMDL process that encourages collaboration. It is more unifying to write a document that outlines doing projects to improve watershed health than to write one that outlines the problems in the watershed. However, taking the time to write the plan rather than simply implementing the projects was viewed by one individual in the North Fork CdA as a key problem in the process: "We have piles and piles of plans with recommendations of how to fix things and we just keep creating more and more and there's nobody to actually to do anything with any of it. That's my complaint and a lot of the locals' complaint." Contradicting that, another member of the North Fork WAG discussed a variety of projects that had been implemented and stated, "I've never seen an implementation plan in this area. We'll never write one." In synthesizing those and other opinions within the North Fork CdA, it appears that they have found a balance between required administrative projects and implementing on the ground projects with stakeholders.

Diverse participants from the Idaho watersheds agreed that the TMDL process needs to be more transparent and allow for more data sources. This connects back to the call for improved and increased monitoring and data analysis, as well as communication, collaboration, and understanding the process. If stakeholders are more involved in the process starting at the beginning, they will be more apt to buy into it and take the necessary actions to improve water quality and work together. But if they do not trust what is written in the TMDL, contention will be more prominent than collaboration.

9. Summary and Conclusions

The case studies of Bear Creek, OR; Middle Snake River, ID; and North Fork Coeur d'Alene, ID provide a glimpse into three watersheds with improving water quality. Phosphorus loads in Bear Creek have reduced significantly leading to recognition by the EPA, but with more progress needed to achieve TMDL targets (USEPA, 2011b). Mid-Snake has come close to meeting sediment targets but stakeholders must continue working toward achieving phosphorus targets and beneficial uses (IDEQ, 2010). Large reductions in sediment loads within some sub-catchments of the North Fork CdA have occurred (USEPA, 2011a), leading to proposed TMDL de-listings. Those lessons learned are now being transferred to other areas of the watershed.

We could not establish a causal relationship between collaborative stakeholder groups in the watersheds and water quality improvements because many confounding factors exist. Instead, we sought to simply examine the roles of the groups in the study watersheds and identify elements that fostered collaboration. While each watershed and group is unique, a few common conclusions can be drawn that could be extended to other locations. Well-established groups have had time to implement projects and observe impacts. They have also had time to build communication, trust, and respect, which are important characteristics to allow for participation of diverse stakeholders. Strong leadership is important in facilitating positive group dynamics and maintaining the focus of the group. Adequate funding is absolutely essential to provide the capacity necessary to carry out group functions. While individuals in the collaborative groups understand and respect the importance of watershed health, public understanding of these concepts beyond the groups is needed to make further progress toward goals. Various types of education and outreach efforts can be used to communicate with the public at both the K-12 and adult levels. Targeting efforts for education and outreach, project implementation, and monitoring can help distribute limited resources (e.g., time, capacity, funds) to where they will have the largest impact.

We have established that diverse participation in groups and public understanding of water quality issues are factors that can foster collaboration. Identifying the motivation for participation, could then be utilized to increase public understanding and participation of others. Within the three watersheds, the motives for participation that could be extended to a broader population were (1) valuing water quality; (2) financial impacts; (3) peer pressure; and (4) threat of regulation. Ideally, using education and outreach efforts to increase public understanding of the value of good water quality would be effective. However, when it is not sufficient, groups can focus on the economics of water quality, the work that other watershed residents are doing, or the potential for regulation.

Throughout the literature, criteria for success vary. Therefore, we sought to understand how study participants view and measure success. Utilizing monitoring and data to measure success was preferred by most, if the study design accounted for lag time and targeting on both spatial and temporal scales. Observing physically obvious improvements, implementing on-the-ground projects, and completing administrative projects were also seen as measures of success. Merging quantifiable water quality improvements and perceived success was suggested as a comprehensive method to measure success because results are clearly measured and are recognized by the people. In order to attain success, goals need to be agreed upon within the group and with the regulating agencies. Goals should be attainable and should lead to restoring beneficial uses, not just meeting numerical standards.

Finally, because the TMDL guides much of the water quality work across the U.S., it is important to understand stakeholder perceptions of the process and generate discussion of improvements that can be made. The TMDL process was seen as contentious by most participants. Oftentimes the data and methods used to create the TMDL are questioned, and the process of setting targets and determining wasteload and load allocations generates inequality and mistrust. Therefore, greater transparency of the process, including stakeholder involvement from the beginning and allowing for the use of more data sources, would increase trust and support of the TMDL, and could lead to better TMDLs. They would also have broader appeal if there were a balance between administrative projects and implementation of on-the-ground projects. Additionally, regulation should be used as a tool rather than a weapon in that it should be used only after other methods to motivate action prove unsuccessful.

To build off of this work, future research could identify other watersheds with improved water quality and attempt to establish a causal link between collaborative groups and water quality improvement. If causation cannot be found, simply understanding the characteristics present in those watersheds and groups could help to understand how success can be achieved, especially for those watersheds still struggling to meet targets. Within this work, further analysis may include continuing interviews in the study watersheds to broaden the perspectives provided and follow up on emerging themes, as well as incorporating results from a fourth watershed that has not documented improved water quality.

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Appendix A

Supplemental Data for Chapter 1: Correlation Matrices for All Numerical Variables in Statistical Analysis

Table A.1 Correlation Matrix for Watershed Outlet Events. See Table 1.1 for definition of variables.

	Event Load	t_d	t_r	t_r/t_d	Q_{max}	Q_{ave}	Q_{max}/Q_{ave}	B	CP	SC	V	D	t_p	P	i	P_7	A	S	M
Event Load	1																		
t_d	0.512	1																	
t_r	0.540	0.756	1																
t_r/t_d	0.197	0.157	0.633	1															
Q_{max}	0.807	0.511	0.460	0.124	1														
Q_{ave}	0.749	0.556	0.525	0.201	0.869	1													
Q_{max}/Q_{ave}	0.042	-0.112	-0.128	-0.181	0.266	-0.182	1												
B	0.184	0.326	0.165	-0.050	0.270	0.504	-0.371	1											
CP	-0.031	0.044	-0.054	-0.187	-0.018	0.092	-0.145	0.352	1										
SC	0.085	0.226	0.169	0.018	0.071	0.121	-0.107	0.174	0.052	1									
V	0.777	0.802	0.713	0.191	0.787	0.839	-0.072	0.350	0.047	0.152	1								
D	0.777	0.802	0.712	0.191	0.787	0.839	-0.072	0.350	0.047	0.152	1.000	1							
t_p	0.308	0.537	0.709	0.397	0.280	0.271	0.005	0.040	-0.066	0.066	0.418	0.418	1						
P	0.536	0.496	0.438	0.212	0.595	0.550	0.105	0.077	-0.090	-0.102	0.668	0.668	0.288	1					
i	0.141	0.113	0.042	-0.123	0.307	0.233	0.243	0.104	0.144	-0.140	0.201	0.201	0.048	0.411	1				
P_7	0.144	0.209	0.050	0.009	0.157	0.240	-0.153	0.408	0.045	0.032	0.198	0.198	-0.041	0.222	0.079	1			
A	0.459	0.577	0.530	0.243	0.531	0.520	0.016	0.022	-0.129	-0.041	0.669	0.669	0.366	0.833	0.418	0.176	1		
S	0.268	0.382	0.301	0.131	0.259	0.351	-0.160	0.084	-0.110	0.005	0.432	0.432	0.039	0.459	0.250	0.174	0.649	1	
M	0.194	0.440	0.424	0.183	0.252	0.283	-0.088	-0.047	-0.122	0.042	0.408	0.408	0.308	0.323	0.263	0.057	0.793	0.604	1

Table A.2 Correlation Matrix for Rural Events. See Table 1.1 for definition of variables.

	Event Load	t_d	t_r	t_r/t_d	Q_{max}	Q_{ave}	Q_{max}/Q_{ave}	B	CP	SC	V	D	t_p	P	i	P_7	A	S	M
Event Load	1																		
t_d	0.436	1																	
t_r	0.414	0.752	1																
t_r/t_d	0.123	0.113	0.651	1															
Q_{max}	0.809	0.506	0.405	0.054	1														
Q_{ave}	0.658	0.399	0.337	0.077	0.859	1													
Q_{max}/Q_{ave}	0.298	0.303	0.230	-0.072	0.418	0.022	1												
B	0.121	-0.005	-0.026	-0.074	0.195	0.556	-0.399	1											
CP	-0.110	-0.273	-0.199	-0.151	-0.141	0.034	-0.313	0.382	1										
SC	0.033	0.167	0.069	-0.097	0.055	0.129	-0.102	0.165	-0.025	1									
V	0.778	0.719	0.610	0.128	0.809	0.798	0.180	0.267	-0.082	0.130	1								
D	0.778	0.719	0.610	0.128	0.809	0.799	0.179	0.268	-0.082	0.130	1.000	1							
t_p	0.194	0.566	0.645	0.359	0.232	0.161	0.170	-0.081	-0.158	0.052	0.339	0.338	1						
P	0.562	0.524	0.498	0.177	0.624	0.477	0.313	-0.005	-0.156	-0.173	0.687	0.687	0.297	1					
i	0.135	0.157	0.176	0.015	0.262	0.194	0.228	0.083	0.059	-0.251	0.208	0.208	0.146	0.540	1				
P_7	0.039	0.079	0.066	0.078	0.159	0.182	-0.007	0.236	0.054	-0.048	0.147	0.147	-0.065	0.294	0.262	1			
A	0.485	0.653	0.616	0.206	0.574	0.403	0.357	-0.095	-0.248	-0.077	0.663	0.663	0.424	0.873	0.455	0.267	1		
S	0.380	0.538	0.438	0.121	0.385	0.293	0.177	-0.008	-0.180	0.020	0.472	0.472	0.308	0.473	0.196	0.218	0.711	1	
M	0.189	0.561	0.525	0.162	0.279	0.143	0.272	-0.175	-0.268	0.080	0.365	0.365	0.420	0.355	0.160	0.125	0.766	0.739	1

Appendix B

General Guiding Questions Used in Watershed Stakeholder Interviews for Chapter 3

Theme 1: Watershed Council/WAG/Other Collaboration Formation

1. What individuals or groups led to the formation of the watershed-scale collaboration?
2. Why did the collaboration develop?
    a. Did it have roots in legislation, a TMDL, a local need?
    b. *Oregon specific:* Was the formation of the council spurred by the watershed council movement in Oregon or was it earlier and/or separate?
    c. *Idaho specific:* Did the WAG form in response to a TMDL?
3. Was the formation of the group formal or informal?
4. Did local government, state or federal agencies play roles in forming the group?
5. Did any other groups play an active role in forming this collaboration?

Theme 2: Leadership and Collaboration in the Watershed

1. Can you characterize the extent of collaboration in the watershed?
    a. How many years did it take to achieve that? i.e., was there a social lag time in the watershed?
2. Is the watershed council or WAG the leader for water quality/watershed projects, collaborative efforts, etc. in the watershed?
    a. If not, what entity is?
3. Is there collaboration with other stakeholder groups and agencies?
4. Are there any political or other barriers to this collaboration?

Theme 3: Stakeholders

1. Who do you define as a stakeholder in the watershed?
2. What stakeholders were involved in the group (watershed council, WAG or other) at the onset?
    a. How have they evolved and changed over time?
3. What stakeholders are currently involved directly with the group or with the watershed improvement projects?
    a. Does that include all relevant stakeholders?
4. Is trust an issue for stakeholders?
5. Have there been/are there any barriers to overcome?
    a. If so, what did it take for stakeholders to get on board with water quality and habitat protection?

Theme 4: Water Quality

1. What are the major land uses in the watershed and how have they changed over time?
2. Is there a TMDL written for any part of the watershed?
    a. What are the target pollutants in the watershed?
    b. What are the main sources of pollutants?
3. Are agricultural pesticides an issue in your watershed?
    a. If so, are they routinely monitored?
4. Is mining/heavy metal pollution present in your watershed?

5. Beyond water quality, what other issues are present in the watershed, such as water quantity, water rights, etc.?
6. What efforts have been undertaken to solve water quality and other watershed health issues?
    a. Is there diverse stakeholder participation in these projects?
    b. At what scales do these projects function? (e.g., whole watershed, stream reach, etc.)
7. To what extent does monitoring happen in the watershed?
    a. What is the physical lag time associated with observing improvements?
    b. Are there monitoring results that clearly indicate numerical improvements in water quality?
8. Would you consider your watershed to have successfully improved watershed health (water quality, aquatic/riparian habitat)?
9. How do you measure success? e.g., perceived success, extent of collaboration/volunteer participation, quantifiable water quality improvements, fish passage, etc.

Theme 5: Education

1. Does your group or any other in the watershed work toward youth and/or community education about the issues affecting the watershed?
2. Do you see education as an important role in improving water quality?

Theme 6: Funding

1. What was the original source and amount of funding?
2. What is the current source of funding?
    a. Is it enough to maintain current projects? To start new ones?

Is there anyone else I should interview that can provide further insight to the collaboration in your watershed?

Appendix C

Codes Used in Analysis of Stakeholder Interviews for Chapter 3

- Barriers and Aids
    - Time, Capacity
    - Leadership
    - Communication
    - Trust
    - Equality
    - Diversity of Representation
    - Politics
    - Land Ownership
    - Compromise
    - Consistency
    - Transparency
- Funding
- Collaboration
    - Stakeholder Participation
        - Motivation
        - Values
    - Group Formation
        - Catalyst
        - Legislation
        - Grassroots
    - Projects
        - Restoration
        - Thermal Credit Trading
        - Fish Passage
    - Goals
    - Decision Making Process
- Education and Outreach
    - Public Understanding
    - Adults
    - K-12
- Measuring Success
    - Monitoring and Data
    - Lag Time
- Science
    - Technical Expertise
    - Local Knowledge
- Environmental Groups
- Litigation
- Targeting
- Government
    - Regulation
    - TMDL
        - Attainability of Targets
- Water Quantity
- Nonpoint Sources
    - Irrigation and Agriculture
    - Timber and Roads
    - Recreation
    - Development
    - Legacy
    - Stormwater
    - Dairies
- Habitat
- Point Sources
    - WWTP
    - Mining
    - Aquaculture
- Dams
- Adaptive Management