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MS ENVIRONMENTAL BIOLOGY
CAPSTONE PROJECT

by

Alex A. Stacy

A Project Presented in Partial Fulfillment
of the Requirements for the Degree
Masters of Science
in Environmental Biology

REGIS UNIVERSITY
May, 2020

MS ENVIRONMENTAL BIOLOGY
CAPSTONE PROJECT

by

Alex A. Stacy

has been approved

May, 2020

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CHAPTER 1. LITERATURE REVIEW

A More General World Under Climate Change

Anthropogenic climate change, via increased greenhouse gas emissions and other pollutants, is shifting global climate patterns. Increased temperatures and alterations to precipitation are the most well-known effects (Karl, Melillo, & Hassol, 2009). Climate change is expected to impact the biosphere in many ways, one of the most important being species diversity. Climate change affects species interactions, which adds to the already decreasing biodiversity (Cahill et al., 2013). Biodiversity is important because of the accompanying ecosystem services that benefit the species in an area, including humans (Christie et al 2012). A single species loss not only disrupts a single function of the ecosystem but can disrupt the functioning of the whole ecosystem (Gamfeldt et al., 2008). The diversity of these components help to maintain a healthy ecosystem. When one species disappears, it is likely that the whole community will suffer (Gamfeldt et al., 2008). Mutualistic species, like bees and flowering plants, are especially important to study because of their role in ecosystem services (Ramirez, & Kallarackal, 2018). Mutualistic species such as plants and bees are increasingly being studied to determine their outcome under climate change.

Climate change negatively affects species-dependent relationships (Cahill et al., 2013). For example, climate change is expected to impact bees and the plants that they pollinate because of their phenology. Phenology in organisms most often refers to life cycles in response to cyclical patterns of climate (Bartomeus et al., 2011). The relationship between flowers and bees under climate is a positive feedback loop in which flowering time is affected which alters bee pollination timing, which in turn influences the plants. As climate change progresses, bees and

flowers likely will have to change their phenology to keep up with the climate, and each other. Bees and the plants that they pollinate are likely to adjust to climate shifts at different rates and are thus likely to be affected by phenologic mismatch (Memmot et al., 2007). Adjustments in phenology could potentially help or hinder a plant under climate change. The success of mutualistic interactions is unequally dependent upon different strengths in relationships between plants and pollinators. Generalist bee populations are more likely to adapt to climate change because they are not limited to the phenology of one plant.

Bees can be classified in to two types of pollinators, generalists and specialists. Generalists pollinate a wide range of plant species, whereas specialists are likely to only pollinate one or two species of flowering plants (“Specialists vs. Generalists - UC Berkeley Urban Bee Lab,” n.d.). Specialist bees may have a harder time adapting to climate change than more generalist species due to their more closely coordinated relationship with a plant (Forrest, 2015; Olliaff-Yang, & Messler, 2018; Schenk, Krauss, & Holzschuh, 2018; Slominski & Burkle, 2019). Generalist bees do not have to perfectly line up their timing with their mutualist partner and are thus more likely to overcome phenological mismatch. Generalist bees have a greater range of available food sources and thus, a potential resilience to climate change (Bartomeus et al., 2011; Bartomeus et al., 2013; Memmot, Craze, Waser, & Price, 2007). The resilience of bees to climate change is closely related to how the plants they receive pollen from respond to environmental variability.

For flowering plants, phenology is the timing of development in reproductive structures (Cleland et al., 2007). Timing of plant phenology is important in making sure the plant can maximize fitness by assuring the optimal seed production at the correct time (Cleland et al., 2007). Theoretically, if a plant can adjust its flowering timing, it will better adapt to changes in

year to year climate variation. This is beneficial to the plant's allocation and timing of resource use. It will also better avoid impairment to its flower structures. The opening of a plant's flowers signals to pollinators that there is pollen available at that time (Ramirez & Kallarackal 2018). Many pollinators are most active when pollen count is at its highest (Minckley, Wcislo, Yanega, & Buchmann, 1994). The relationship between bees and flowering plants is mutualistic because the pollinators get food resources from the flower and the plant can optimize its reproductive strategy from pollination.

Bee phenology is mainly associated with wintering and reproduction. Many longer-lived bee species hibernate in the winter (Doke, Frazier, & Grozinger, 2015). These winter bees use adaptive phenology to determine when they emerge from hibernation (Doke, Frazier, & Grozinger, 2015; Kudo, 2014). If a bee were to wake up from hibernation too late or too early, it could miss the blooming period of their preferred plant. In addition, reproductive capacity has been positively correlated to pollen availability for female bees (Schenk, Krauss, & Holzschuh, 2018). Flowering plant pollen production is thus directly related to reproductive success in bees (Schenk, Krauss, & Holzschuh, 2018).

Temperature is the most important driver of both plant and insect phenology (Slominski, & Burkle, 2019). As anthropogenic climate change increases, so does temperature (Memmot et al., 2007). Therefore, many scientists are worried about the effects of climate change on these species (e.g. Memmot et al., 2007; Sherry et al., 2007; Bartomeous et al., 2011; Bartomeous et al., 2013). Flowering plants adjust to experimental warming in different ways. In some early-summer flowering plants, warming advances reproductive phenology, whereas in some late-summer flowering plants, warming delays reproductive phenology (Sherry et al., 2007). This decreases the overlap between early-flowering and late-flowering plants (Sherry et al., 2007). A

limited overlap could lead to a gap in pollen resources on which bees depend. This is, however, species dependent and determined by whether the flowering responds more to temperature or photoperiod. In plants where photoperiod is the main driver of flowering, warming could negatively affect the plant via temperature stress (Weis & Berry, 1988). Life span also affects timing mismatches in plant-pollinator interactions. If a plant or pollinator is short lived, they are more susceptible to mismatches because they have less time to meet reproductive requirements (Fagan et al., 2014). In addition to intraspecies variability, there will be variability between the species in phenologic adjustments to climate change. Not every bee species will react to climate change in the same way, neither will every plant.

Historically, specialist bees have adapted their phenology to wake up from hibernation at the optimal time to match the flowering of their mutualistic plant species (Minckley, Roulston, & Willams, 2013). In specialist bees, temperature was found to alter timing and development of several biological fitness processes including wintering emergence and weight (Slominski & Burkle, 2019). Mason bees (*Osmia spp.*), who overwinter longer than leafcutter bees (*Megachile spp.*), were more sensitive to losses in fitness after temperature changes (Slominski & Burkle, 2019). This indicates that overwintering may decrease bee fitness as climate change progresses (Slominski & Burkle, 2019).

In the last 130 years, generalist wild bees in North America have been keeping pace with temperature increases and their host plant flowering times (Bartomeus et al., 2011; Forrest, 2015). In a historical assessment of 10 generalist bee species, models found that generalist bee phenology is advancing with climate change and coordinated with the phenology of plant species used for foraging (Bartomeus et al., 2011). This does not, however, mean that as the rate of

temperature change increases, bees will continue to be able to keep up, and further studies are needed to determine the full extent of this relationship.

Climate change increases the potential for mismatched pollination timing. If there is a high diversity of generalist bee species, plants are less likely to be affected by a mismatch in pollination timing, because higher bee diversity increases the range of time available for pollination (Bartomeus et al., 2013). Different bee species adjust to changes in phenology at different rates. A high diversity of pollinator species can mitigate the effects of mismatched timing by stabilizing pollination rates (Bartomeus et al., 2013). Memmott et al. (2007) use a model to calculate that between 17% and 50% of pollinators are expected to have deficiencies in food from pollen and nectar under climate change scenarios. Bees that visited fewer species of plants (specialists) were more likely to face deficits than more generalist bees (Memmott et al. 2007). However, extinction of specialist bees does not necessarily mean extinction of the plants they pollinated because they may still be visited by generalist pollinators (Memmott et al., 2007, Forrest, 2015).

Some of these specialist bee species are ground-nesting bees which creates unique challenges for their overwintering under climate change. Ground-nesting bee emergence time from overwintering is closely linked to maximum temperature, whereas their mutualist plant species' flowering time is more responsive to average temperature (Olliaff-Yang & Messler, 2018). Increased soil moisture advances emergence time in ground-nesting bees but delays flowering times which further separates their pollination overlap (Olliaff-Yang, & Messler, 2018). Currently, their phenological overlap is sufficient, but may become less stable in the future. These species also show lower fitness due to staggered emergence of different sexes of bees which negatively affects the ability of the bees to time reproduction (Olliaff-Yang, &

Messler, 2018). Ground-nesting bees are more likely to be affected by climate change than other bee nesting types because of their relationship with temperature and soil moisture (Olliaff-Yang, & Messler, 2018).

Studies show that interactions of plant specialist bees are completely disrupted, leading to a loss of both the plant and pollinator (Forrest, 2015). In an experimental study, different specialist bee species were followed under 3 scenarios in which temporal mismatch was up to 6 days. These bee species were unable to meet their resource requirements when timing was modified to 6 days, and one species-specific bee pollinator was unable to meet the resource requirements when timing was modified to as little as 3 days (Schenk, Krauss, & Holzschuh, 2018). These species were unable to mitigate their fitness loss, specifically reproduction, under mismatched flower pollen scenarios (Schenk, Krauss, & Holzschuh, 2018). More experiments like this need to be done to fully understand how each bee species will respond to climate change.

As climate change progresses, it is important to understand what kinds of species relationships could be affected. Specialist bees are likely to be more hindered by climate change than their generalist relatives because specialist plant-bee interactions have a higher potential for timing mismatches. Some may view the loss of specialist bees as mainly negligible, but that loss decreases overall biodiversity. In addition, there may also be unforeseen consequences to other organisms that interact with these bee species that have yet to be studied. Further work needs to be done to assess the full impact of climate warming on the flower and bee positive feedback loop. It is critical to determine whether the potential loss of specialist bee species can be outweighed by the persistence of generalist bee species.

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CHAPTER 2. GRANT PROPOSAL

Determining the Effectiveness of an Unmanned Aerial Vehicle (UAV) Versus Traditional Ground Surveys to Map the Distribution of Bell's Twinpod (*Physaria bellii*) in Boulder County

Section 1. Abstract

Establishing maps of plant species populations is important for monitoring and conservation, but traditional ground surveys are time consuming. As technology advances, UAVs (unmanned aerial vehicle) are increasingly being used to create maps. Bell's twinpod (*Physaria bellii*), a threatened plant species endemic to Colorado, lacks sufficient monitoring on BCPOS (Boulder County Parks and Open Space) lands. I propose to determine whether UAV technology can be used to map the species distribution of *P. bellii* more efficiently than traditional ground surveys. To do this I will compare GPS points of *P. bellii* individuals from ground surveys to those located by UAV processed images. If the UAV can successfully detect *P. bellii*, this technology will provide species map distributions to BCPOS for monitoring, as well as help to close the knowledge gap in using UAV technology for creating other plant species distribution maps.

Section 2. Literature Review, Objectives, Hypotheses, Anticipated Values

Literature Review

Bell's twinpod (*Physaria bellii*) is a perennial herb in the Brassicaceae family with obovate basal leaves that form a rosette, yellow flowers, and paired fruits (Mulligan, 2013). This species is endemic to the Front Range of Colorado and is found on exposed rock formations of limestone or shale associated with shrubland or grassland ecosystems (Fig.1; Carpenter, 1997).

P. bellii faces extinction from anthropogenic-induced habitat loss due to limestone mining, housing developments, and construction in the Front Range, as well as invasive-species encroachment (Mulligan, 2013; Peterson & Harmon, 1981). In light of Boulder County Parks and Open Space's (BCPOS) goal to conserve rare plant species and sustain a viable population, two studies of *P. bellii* have been conducted on BCPOS lands to provide more information about this rare plant's biology (Carpenter, 1997; Kothera, Ward & Carney, 2007). Prior monitoring, though valuable, has not been conducted on a full scale since 1997 and thus needs updating. In order to conserve this imperiled species, a reliable form of monitoring and management is critical. Traditionally, ground surveys have been used to monitor plant populations, but new technology has opened up faster and more efficient techniques.

Unmanned aerial vehicles (UAV) have been increasingly useful in creating spatial maps in an affordable, fast, and detailed way (Baena et al., 2017). This technology can be used to produce orthostatic maps (geometrically accurate to the earth's surface) and three-dimensional digital surface models of a landscape (Baena et al., 2017; Cruzan et al., 2016). UAV data collected over different periods of time can assess growth differences across seasons of a plant community (Cruzan et al., 2016). While using UAVs to collect this kind of vegetative data requires less human labor, processing all these data in a time-efficient manner still remains a challenge (Baena et al., 2017; Gonçalves et al., 2016).

Much of the current literature on using UAVs in plant ecology determines that they can be used to map species occurrences in a variety of landscape types (Baena et al., 2017; Cruzan et al., 2016; Sun, Yi, & Hou, 2018). UAVs have been successfully used to identify larger plants such as trees in a heterogenous landscape (Baena et al., 2017). This information is useful in monitoring communities and providing a basis for conservation and restoration projects (Baena

et al., 2017). Minimal data is available, however, on UAV projects that have focused on a specific plant the same size as *P. bellii*, which can be between 5 and 13cm in width (Mulligan, 2013). One such study focusing on a single species of poppy found that the orthostatic data were not able to detect an individual plant <10 cm (Rominger & Meyer, 2019). Thus, it is important to understand if UAVs are able to detect populations of clustered plants as well as single individuals of both large and small size to better monitor *P. bellii*.

Objectives

I propose to determine the feasibility of using an Unmanned Aerial Vehicle (UAV) to map the spatial distribution of *P. bellii*, a rare and endemic plant species at BCPOS. Mapping the spatial distribution is critical in assessing the geographic range of the species. The goal of this research is to better understand the capabilities of using UAV on a single plant species, *P. bellii*, to help inform conservation efforts as well as establish a more efficient monitoring technique for single species.

Hypotheses

Question: Can UAV technology successfully determine the spatial distribution of *P. bellii*?

H1: UAV technology will be as accurate as ground surveys in detecting *P. bellii* individuals.

H2: UAV survey data processing is more time efficient than ground survey data processing in creating spatial maps of *P. bellii* distribution.

Anticipated Value

Endemic to the Front Range of Colorado, *P. bellii* is found on BCPOS land. Therefore, it is imperative that the BCPOS conserve this rare species to preserve local biodiversity. An assessment of the capability of UAV technology to detect this species can show whether this technique is able to produce maps of the current species distribution. These maps are critical for

the conservation of this species because establishing an efficient monitoring technique helps land managers create more comprehensive management plans. The success of UAV technology for detecting and mapping a single species allows for that species to be monitored over time with more ease than ground surveys. This research can also be applied to mapping other species of concern in a quick and efficient manner. If UAVs are successful on the test plots, this technique can be applied across the whole geographic range of *P. bellii*.

Section 3. Methods

Detailed Sampling and Analysis Plan

Specific Aim 1: UAV vs ground survey detection accuracy assessment

To assess the accuracy of using a UAV to detect and capture spatial data on *P. bellii* on BCPOS land over that of ground surveys, I will do surveys using both techniques and then compare them. I will use the 10 macroplots that have been used to monitor *P. bellii* populations in the past which are located along highway 36 in Boulder, CO (Fig.2; Carpenter, 1997). Both survey types will be completed during the flowering period of *P. bellii*, which is from May-July. At each of the 10m x 10m plots, I will conduct ground surveys when the plants are in bloom by using an RTK GPS unit to gather the coordinates of the location of every *P. bellii* individual in each plot (Ehsani, Upadhyaya, & Mattson, 2004). Once the coordinates are recorded, I will measure the width of each individual of *P. bellii*. To conduct the UAV surveys, I will program the UAV to take high-resolution (15m resolution) imagery from a 30m altitude within each plot (Rominger & Meyer, 2019). I will then process the UAV data using Pix4D software to create an orthomosaic of the data. The orthomosaic will then be put into ArcMap 10.5 and overlaid with the coordinates of the plants from the ground survey. I will then create a protocol to locate *P. bellii* individuals based on their width (5-13cm), color (yellow), and shape (round). Areas fitting

these characteristics will be marked in a separate data layer. Once I have the data points from both survey types, I will test how accurate the UAV points are in comparison to the ground survey points using inter-observer reliability of the two techniques (Salvatore et al., 2008). If the UAV is 90% accurate in comparison to the ground survey points, I will determine that the UAV method is successful. Additionally, I will run a linear regression to determine whether there is a relationship between recognized *P. bellii* individuals and diameter (Arc Maps 10.5).

Specific Aim 2: UAV vs ground field study efficiency assessment

If the UAV detects *P. bellii* as accurately as the field survey, I will assess the efficiency of using UAV technology. During the process of specific aim 1, I will record how long it takes to collect and process the data from each survey type. I will conduct a t-test on the amount of time (in minutes) it took for each type of data collection and processing (R studio). If the UAV technology is significantly faster, I will recommend that it be used to continue monitoring the *P. bellii* population on BCPOS land.

Project Requirements, Logistics, and Negative Impacts

I will communicate with BCPOS to get permission to carry out in-person ground surveys as well as UAV surveys on BCPOS lands. BCPOS will be provided with a detailed map of the study plot locations as well as dates and times of surveys. UAV use will comply with all FAA regulations for research use. Because both UAV and ground surveys are minimally invasive, I do not foresee any negative impacts to the study species or habitat.

Timeline

Dates	Activities	Deliverables
June 2020-July 2020	<ul style="list-style-type: none"> Establish coordinates of past plots Collect ground survey data on <i>P. bellii</i> presence 	<ul style="list-style-type: none"> Raw data from ground surveys
August 2020-September 2020	<ul style="list-style-type: none"> Program UAV and run flight survey 	<ul style="list-style-type: none"> Raw data from UAV survey
October 2020-November 2020	<ul style="list-style-type: none"> Data analysis of <i>P. bellii</i> presence in different survey types Start writing report 	<ul style="list-style-type: none"> Results of processing time and accuracy of the two methods Draft report
December 2020	<ul style="list-style-type: none"> Synthesize and revise report 	<ul style="list-style-type: none"> Final report

Section 4. Budget

Item	Justification	Cost, unit	Quantity	Total cost
UAV with camera	Needed for UAV surveys	\$919.99	1	\$919.99
External hard drive (2 Tb)	For large amounts of data storage	\$39.99	1	\$39.99
GPS	Necessary for ground surveys	\$149.59	1	\$149.59
Pix4D	Software for image processing	\$495	1	\$495
Gas	Transportation to and from site	\$2.60 per gallon	4	\$10.40

Field technician	Student work incentive for carrying out ground surveys	\$1000	1	\$1000
Measuring tape (25')	For plant measurements	\$10	1	\$10

Total Cost: \$2,624.97

Section 5. Appendix and References

Appendix

A. Maps of Study Area

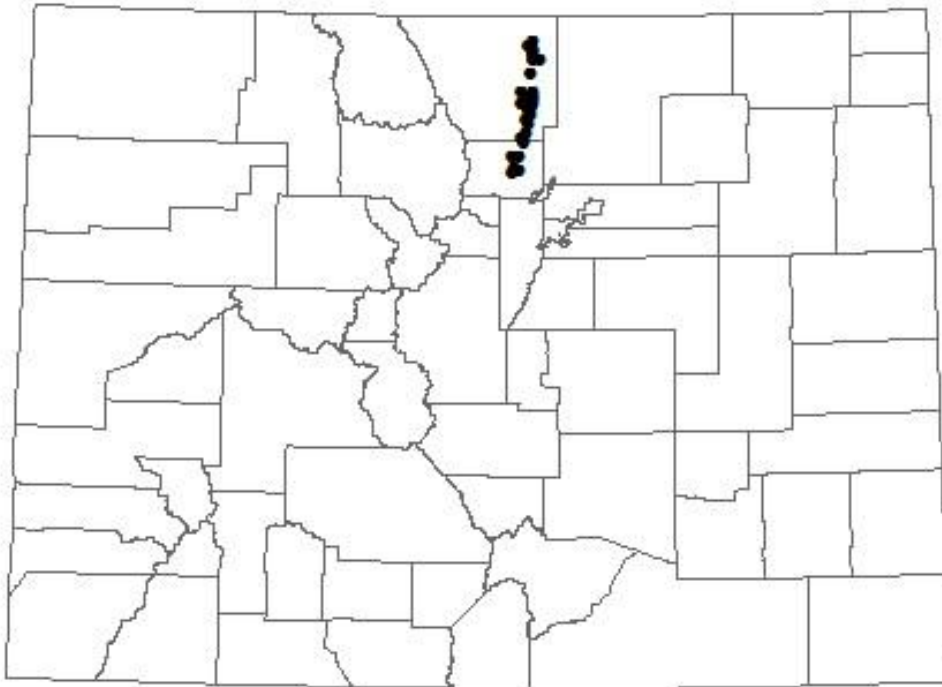


Figure 1. A map of Colorado showing *P. bellii*'s known geographic distribution (Mulligan, 2013).

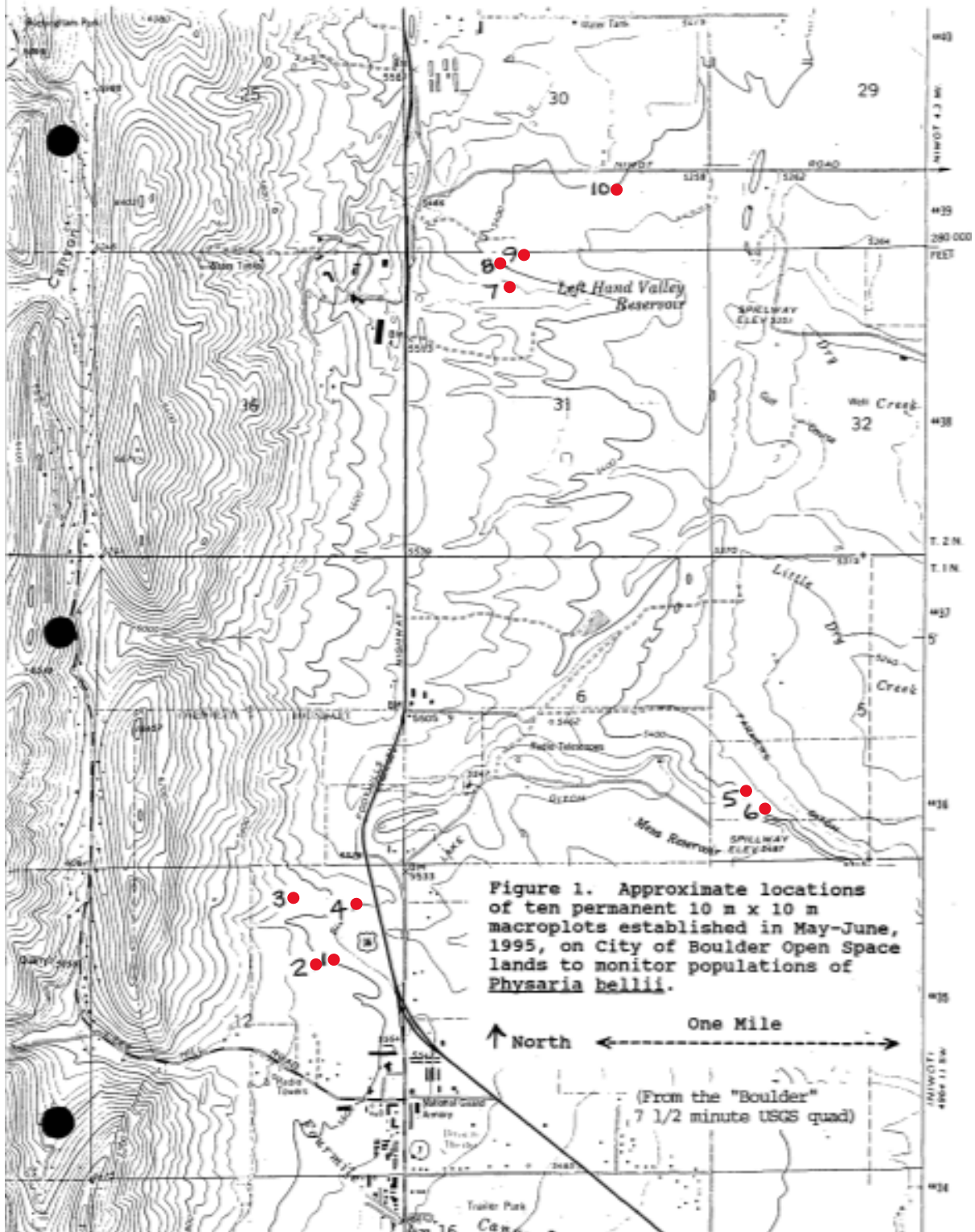


Figure 2. Approximate locations of ten permanent 10m x 10m macroplots established in May-June, 1995, on City of Boulder Open Space lands to monitor populations of *Physaria bellii* (Carpenter, 1997). Macroplots are indicated with red dots.

*B. Qualifications of Researcher***Alex Stacy**

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EDUCATION**Bachelor of Arts in Environmental and Earth Science**

Willamette University, Salem, OR

May 2019

Minor - Ecology/Evolutionary Biology

Master of Science in Environmental Biology

Regis University, Denver, CO

May 2020

FIELDWORK EXPERIENCE**Two Ponds National Wildlife Refuge** Arvada, CO Fall 2019

- Assessed invasive species cover
- Optimized research design
- Identified native and invasive plants
- Analyzed data through R

Rio Mora National Wildlife Refuge Watrous, NM 2019-2020

- Assessed grassland productivity
- Optimized research design
- Identified native plants
- Analyzed data through R

Zena Forest Eola Hills, OR Seasonal: October 2016-October 2018

- Set up and monitored tick traps for a professor's research project
- Maintained trails by thinning and eradicating invasive species

Archeology Field School Ness of Brodgar, Orkney Islands, Scotland July-August 2018

- Used different excavation techniques to uncover layers and structures in soil
- Utilized surveying tools such as total station and GPS for planning and mapping
- Applied theory on neolithic structures and developments to field notes

EMPLOYMENT EXPERIENCE**Digital Observer Inc., Fish Technician** Bristol Bay, AK June-July 2019

- Monitored salmon quality using a variety of techniques
- Managed customers in an efficient and friendly manner
- Practiced proper data collection

Capra Environmental Services Corp., Intern Rancho Cordova, CA Seasonal: June 2017-Jan 2018

- Researched environmental advantages of goat grazing
- Developed a business plan and set up corporation documents
- Communicated with shareholders to ensure operations progressed

Ikeda's Country Market Cashier, Auburn, CA May-August 2016

- Responsible for up-to-date memory of product codes and information
- Facilitated positive customer experience
- Handled and counted money when opening or closing tills for the day

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CHAPTER 3. JOURNAL MANUSCRIPT

Macroinvertebrate Assemblages Indicate Declining Ecosystem Diversity for In-Stream Restoration Sites at Deer Creek, CO

Abstract

As more stream restoration projects are put in place, it is important to understand how biotic communities can be expected to respond. Beaver dam analogs installed in Deer Creek in Littleton, CO, have been monitored over time to track biotic and abiotic responses. To assess the progress of restoration in Deer Creek, I analyzed aquatic macroinvertebrates as ecosystem surrogates for changes over a four-year period post-restoration. I hypothesized that in-stream restoration structures would increase the connectivity of the stream and allow for more niche space for macroinvertebrates. I also suggested that macroinvertebrates at in-stream restoration sites should show signs of recovering from installation disturbance. However, faster declines in macroinvertebrates at restoration sites than control sites indicate disturbance from in-stream restoration may be further impairing macroinvertebrates. Additionally, declines in both restoration and control sites indicate that regional forces such as drought and urbanization are causing negative impacts on macroinvertebrates in Deer Creek.

Introduction

Watersheds across the globe have been greatly disturbed by human use and modification (Poff et al., 1997; Bernhardt et al., 2005). Land cover change, channelization, damming, and other stream modifications by humans have impacted the natural flow regimes in most US waterways, and this has in turn damaged the biological processes and functions of these streams (Poff et al., 1997; Allan, 2004). For example, when forested watersheds are converted to urban

land, peak flows and hydrograph flashiness both increase (Paul & Meyer, 2001; Walsh et al., 2005). Because riverine organisms have evolved life history strategies to match specific flow characteristics (Poff et al., 1997), when flow regimes are altered organisms whose traits are maladapted to the new flow regime may be extirpated from the system (Poff et al., 1997). Consequently, anthropogenic stressors that disrupt the physiochemical structure of freshwaters have resulted in steep declines in freshwater biodiversity worldwide (Dudgeon et al., 2006; Ricciardi & Rasmussen, 1999). In order to stem these biodiversity losses, watershed managers have implemented restoration projects that aim to recover natural flow regimes (Stranko, Hilderbrand, & Palmer, 2012).

In the last 30 years, stream restoration projects have become increasingly common, but our knowledge of biotic responses to restoration is still inadequate (Palmer, Menninger, & Bernhardt, 2010). Many stream restoration projects are costly and time consuming, so it is important to understand how freshwater biota respond to these projects (Poff et al., 1997; Jähnig et al., 2011). Bioindicator species are often used in conjunction with abiotic variables, such as pH, temperature, and dissolved oxygen, because together these measures comprehensively communicate the overall health of stream systems (Stranko, Hilderbrand, & Palmer, 2012; Yarsan & Yipel, 2013). Macroinvertebrates are frequently the bioindicator of choice for many stream monitoring programs due to their high diversity, abundance, and varied responses to water quality and habitat degradation (Resh & Rosenberg, 1993; Resh, 2008; Merritt & Cummins, 2008). Because individual macroinvertebrate species vary in their tolerance of pollutants and degraded environmental conditions (Merritt & Cummins, 2008), the composition of macroinvertebrate communities is frequently used to indicate the biotic integrity of stream ecosystems (Resh & Rosenberg, 1993). Additionally, macroinvertebrates are ideal for indicating

environmental stresses because they bioaccumulate toxins and are resilient to natural disturbances (Resh & Rosenberg, 1993; Grimm & Fisher, 1989; Resh, 2008). For these reasons, monitoring macroinvertebrate compositional changes can indicate whether restoration techniques successfully improve ecological stream condition. For example, if restored sites witness greater increases in macroinvertebrate biodiversity than unrestored sites, restoration practices likely improve the biotic integrity of the system (Miller, Budy, & Schmidt, 2010). However, because macroinvertebrate assemblages may have delayed responses to restoration efforts, especially when restoration activities themselves disturb stream biota, continued long term monitoring may be necessary to assess whether communities have improved (Orr et al., 2020).

Restoration strategies that aim to improve natural flow regimes include in-stream structures like temporary sod plugs (TSPs) that mimic the ecosystem engineer work of beavers (Orr et al., 2020). Natural beaver dams slow the incision of channels and force water into the floodplain creating a nutrient rich vegetation zone that helps maintain river water quality (Orr et al., 2020, Poff et al., 1997). Beaver dams increase physical habitat heterogeneity within streams, and in the process enhance biodiversity by creating more niche space for freshwater organisms (Fuller & Peckarsky, 2011; Polluck et al., 2014). In areas where native populations of beavers have been reduced, TSP installations should mimic natural beaver dams with the goal of restoring both ecosystem structure (e.g., biodiversity) and functions (e.g., stream flow patterns) (Polluck et al., 2014). Demonstrating whether in-stream structures like TSPs improve in-stream biota is important because in-stream restoration projects are a common tool used by watershed managers.

Although many restoration studies commonly use macroinvertebrate biodiversity measures to assess restoration success, far fewer interpret the community structure through the

lens of functional feeding groups analysis (Merritt & Cummins, 2008). Functional feeding groups are defined by the different community roles fulfilled by an organism's utilization of a specific food source type (Merritt & Cummins, 2008). Macroinvertebrate functional feeding groups include predators, shredders, gathering collectors, filtering collectors, parasites, and herbivores (Merritt & Cummins, 2008). The presence and relative abundance of different combinations of these functional feeding groups can indicate how well an ecosystem is functioning (Merritt & Cummins 2008). For example, higher proportions of shredders in forested watersheds indicate efficient processing of incoming leaf litter (Boyero, 2005), a key resource for many small shaded streams (Wallace et al., 1997). Restoration techniques such as TSPs are likely to increase the accumulation of coarse particulate organic matter by trapping leaf litter, therefore supporting a diverse community of shredding invertebrates (Merritt et al., 2002). Additionally, higher values of the ratio of predator to prey indicate higher turnover in prey macroinvertebrate taxa, indicating a well-functioning food chain (Merritt et al., 2002). Though much research has elucidated the functional roles of these feeding groups in stream ecosystems (Cummins & Klug 1979), few studies have investigated their utility in restoration assessment

Managers of Chatfield Farms in Littleton, Colorado have utilized in-stream restoration practices for a stream (Deer Creek) that runs through their property. Deer Creek is a stream that has long faced human influence, currently it runs through partly urbanized landscapes including Chatfield Farms, a property managed by the Denver Botanic Gardens (DBG). In 2015, DBG installed TSPs and restored vegetation along corridors where beavers would have naturally built dams. To assess restoration progress, managers conducted annual monitoring of physiochemical measures and macroinvertebrates in the stream. Analyzing macroinvertebrate functional feeding groups will complement this monitoring by indicating food chain health and the accumulation of

particulate matter that mimics streams with naturally occurring beaver dams. By comparing restored and unrestored stream reaches, I will answer the following: 1) Has Chatfield Farms use of TSPs successfully restored macroinvertebrate diversity in Deer Creek? 2) Has the restoration altered ecosystem function as indicated by macroinvertebrate functional feeding groups?

Because TSP structures mimic the ponding function provided by beaver dams, these structures should increase the likelihood of finding pools upstream of the TSP. With more pools and fewer riffles near TSPs, I predict that restored sites will exhibit lower dissolved oxygen and temperature than unrestored sites. By increasing water residence time, floodplain inundation, and organic matter retention within restored sites niche space for macroinvertebrate taxa should increase in TSP sites. Consequently, I predict that macroinvertebrate diversity in restored sites should increase over the course of the study while diversity in unrestored sites should remain flat. Lastly, TSPs should accumulate leaf litter behind the dam-like structures, resulting in macroinvertebrate communities with more shredders to process the leaf litter.

Methods

Site description

Deer Creek in Littleton (Arapahoe County), Colorado flows through Chatfield Farms, a restoration site managed by the Denver Botanic Gardens (Fig.1). Chatfield Farms is at an elevation of approximately 1690m surrounded by arid grasslands. Deer Creek is situated on the west foot of the Rocky Mountains accumulating approximately 6.7 cm of precipitation annually, and throughout its length in Chatfield Farms is surrounded by a dense riparian canopy of mainly invasive species. Deer Creek is a tributary of



Figure 1. Chatfield Farms site in Littleton, CO.

the South Platte River that has been continually degraded by urbanization, mining, and agriculture since the 1860s (Paces, 2019).

Study Design

DBG managers have attempted to restore the riparian zone around Deer Creek by increasing connectivity of floodplains. In 2015, they introduced vegetation as well as in-stream TSP structures. At the Chatfield site, DBG managers have attempted to restore historic beaver function by installing three TSP structures that mimic natural beaver dams. Installations of wooden stakes held down by biodegradable weighted bags make up the TSP structures. The TSP structures were destroyed by flood conditions in 2016 and replaced in March of 2017. To monitor the progress of their stream restoration 12 transects were set up, 3 at TSP sites and 9 at control sites (Fig. 2). Six more transects were added in 2018 but are not included in this study for consistency across all 4 years of data.

Monitoring

DBG managers set up monitoring of the 12 transects to determine the how macroinvertebrate communities respond to TSP structures in Deer Creek. All monitoring followed the methods of *Aquatic Monitoring Protocol for measuring and collecting data* (Hufft, Paces, McGill, & Levy, 2016). From 2016-2019 in the summer, managers sampled each of the 12 transects for aquatic macroinvertebrates and physiochemical measurements.

Aquatic macroinvertebrate samples were collected using the kicknetting method (Walk et al., 1997). Kicknetting with a 500 μm net was performed for 1 minute every 5 meters starting at 20m downstream of the transect origin. At each sampling bout, the stream was physically disturbed 1m² upstream of the net. Net contents were flipped into a 500 m bucket at each sampling bout. Then, no more than 50% of sampled material was scooped into 1L jars and

labeled. Compiled samples were preserved in 70% ethanol and sent to GEI Consultant Inc. for processing. In the laboratory, GEI identified individuals to the lowest practical taxonomic level and compiled composition counts by total taxon richness, EPT (sensitive taxa orders Ephemeroptera, Plecoptera, and Trichoptera) richness and abundance, Ephemeroptera abundance, Shannon diversity index, and the Colorado multimetric index (CO MMI).

In-stream physiochemical measurements were taken at the origin of the transect and included thalweg depth or deepest wadable point (m), wetted channel width (m), microhabitat (percentage of riffle, run, pool, undercut, or other), dissolved oxygen (DO, mg/L) or percent concentration of oxygen, total dissolved solids (TDS, ppm), pH, temperature (degrees Celsius), electrical conductivity (EC, microsiemens/cm). In years 2017-2019 percent concentration of oxygen was converted to DO using the University of Minnesota Natural Resources Research Institute DO- %saturation calculator (UMN, 2015). The elevation used in the calculation was taken at the midpoint of the transects. Two water samples (250ml, 150ml) were taken in-situ for Nitrate over nitrite and total Kjeldahl nitrogen (mg/L) and processed by the Colorado Department of Public Health and Environment.



Figure 2. Map of the transect sites along Deer Creek in Littleton, CO. Points labeled “New Transects” are not included in this study. (R. Hufft, personal communication, Jan. 23, 2020)

Statistical Analysis

I used generalized linear mixed models to compare temporal trends of abiotic and biotic variables between restored and unrestored sites. These models included fixed effects for year, restoration status, and their interaction as well as a random effect for site to account for non-independence.

I performed a principal component analysis (PCA) to quantify the major environmental gradients distinguished by the correlated physical and chemical habitat measures using the *ecodist* R package (Goslee and Urban 2007). Missing data points were corrected for using the *pcaMethods* package in R (Stacklies et al., 2007). Log transformations were done to correct for non-normal distribution on width, depth, percent riffle, percent runs, percent undercut, and percent pools.

To examine how species composition changed over time in restored and unrestored sites, I used a non-metric multidimensional scaling (NMDS) ordination of the Bray-Curti distance matrix calculated from $\log(x+1)$ transformed densities (McCune & Grace, 2002). Following the

methods of Voss & Bernhardt (2017) only species that met the 0.5% density threshold were used in the analysis. I used a two-dimensional ordination based on the result of a stepdown procedure that showed minimal reduction in stress after two dimensions. The final ordination was run from the lowest stress value from 1000 random start values and then rotated with principal components analysis to put the largest explained variation on the first axis. To find the strength of the relationship between community ordination scores and physiochemical variables, I fit environmental variables post hoc using the methods from the vegan R package (Oksanen et al., 2015). To determine how community structure varied by year and transect type, I conducted a permutational multivariate analysis of variance.

To determine if functional feeding groups can be used to assess restoration progress, I assigned primary functional feeding group based upon the *Freshwater Biological Traits Database* (Environmental Protection Agency, 2010). Species-specific functional feeding groups were used when available, and when not, I assigned them by family. I then used this information to calculate different proportions for ecosystem parameter surrogates including shredder, trophic, filtering collector, predator-prey, and channel stability indices (Merritt, & Cummins, 2008). I used generalized linear mixed models to compare temporal trends of ecosystem surrogate indices between restored and unrestored sites. These models included fixed effects for year, restoration status, and their interaction as well as a random effect for site to account for non-independence.

I performed all analyses in R version 3.6.3 (R Core Team, 2019).

Results

Abiotic Effects of Restoration

Transects in this study differed in both physical and chemical habitat characteristics (Fig 2). 37.8% of the variation in abiotic variables was explained by the first (21.8%) and second

(16.0%) axes generated from a principal component analysis (Table 1). Physical habitat characteristics including velocity-depth regimes (percent of stream habit in runs, pools, and other), and channel morphology (log(width), and log(depth)) loaded more strongly onto the first principal component (Table 1). On the other hand, chemical measures including, pH, TDS, EC, DO, and nitrate over nitrite tended to load more strongly onto the second principal component (Table 1).

Table 1. Loadings of environmental variables on the first two principal components in the PCA. The higher the value, the stronger the influence of the variable on the principal component. Values over +/- 0.3 are bolded. The sign of the value indicates the direction of the correlation (+/-).

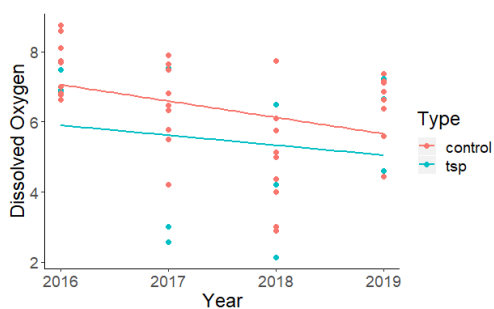
Abiotic Variable	Principal Component 1	Principal Component 2
Temperature	0.1225	0.1394
pH	-0.0500	0.2928
Total Dissolved Solids	0.2638	0.4304
Electrical Conductivity	0.3420	0.4173
Dissolved Oxygen	-0.2417	0.2439
Nitrate over Nitrite	-0.0119	0.3456
Total Kjeldahl Nitrogen	0.3119	-0.1986
Log(Width)	0.2875	0.0492
Log(Depth)	-0.0292	-0.0684
Log(Riffles)	-0.1662	0.4134
Log(Undercut)	0.0428	-0.0754
Log(Runs)	-0.5006	0.0421
Log(Other)	0.3684	0.1604
Log(Pools)	0.3751	-0.3011

Baseline measurements of several physical and chemical habitat variables differed between TSP and control sites and in some cases showed clear trends over the course of the study. However, when there were clear trends over time, these trends were similar for TSP and control sites.

Important chemical habitat variables either declined or stayed the same over the course of the study (Fig. 2, a-d). pH became significantly more acidic ($p = 2.6 \times 10^{-6}$, 4.3×10^{-5}) in both restored and unrestored sites. Average pH in control sites declined by 0.31 units (95%CI: 0.20-0.42) and 0.037 units (95%CI: 0.18-0.57) annually. Dissolved oxygen only declined significantly ($p = 0.045$) in control sites, with average dissolved oxygen decreasing by 0.47 mg/L per year (95%CI: 0.01-0.92). Both electrical conductivity and temperature were stable over time ($p = 0.1833$, 0.8050).

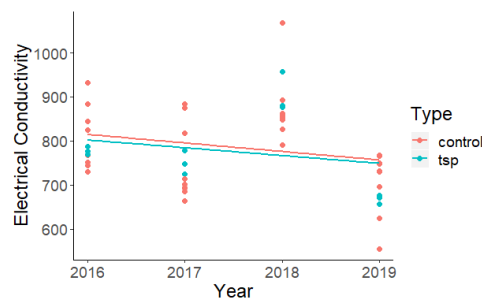
Important physical habitat variables showed mixed trends but marginal decreases over the course of the study dominated (Fig. 2, e-h). Percentage of riffles declined significantly ($p = 0.0017$) at a rate of 8.5% every year (95%CI: 3.42-13.54) in control sites and 5.4% every year (95%CI: 3.33-14.14) in TSP sites. Percentage of pools showed marginal increases ($p = 0.0540$) at a rate of 8.6% annually (95%CI: -17.43-.015). Channel morphology measurements of width and depth did not change significantly over time ($p = 0.1538$, 0.0885).

a.



c.

b.



d.

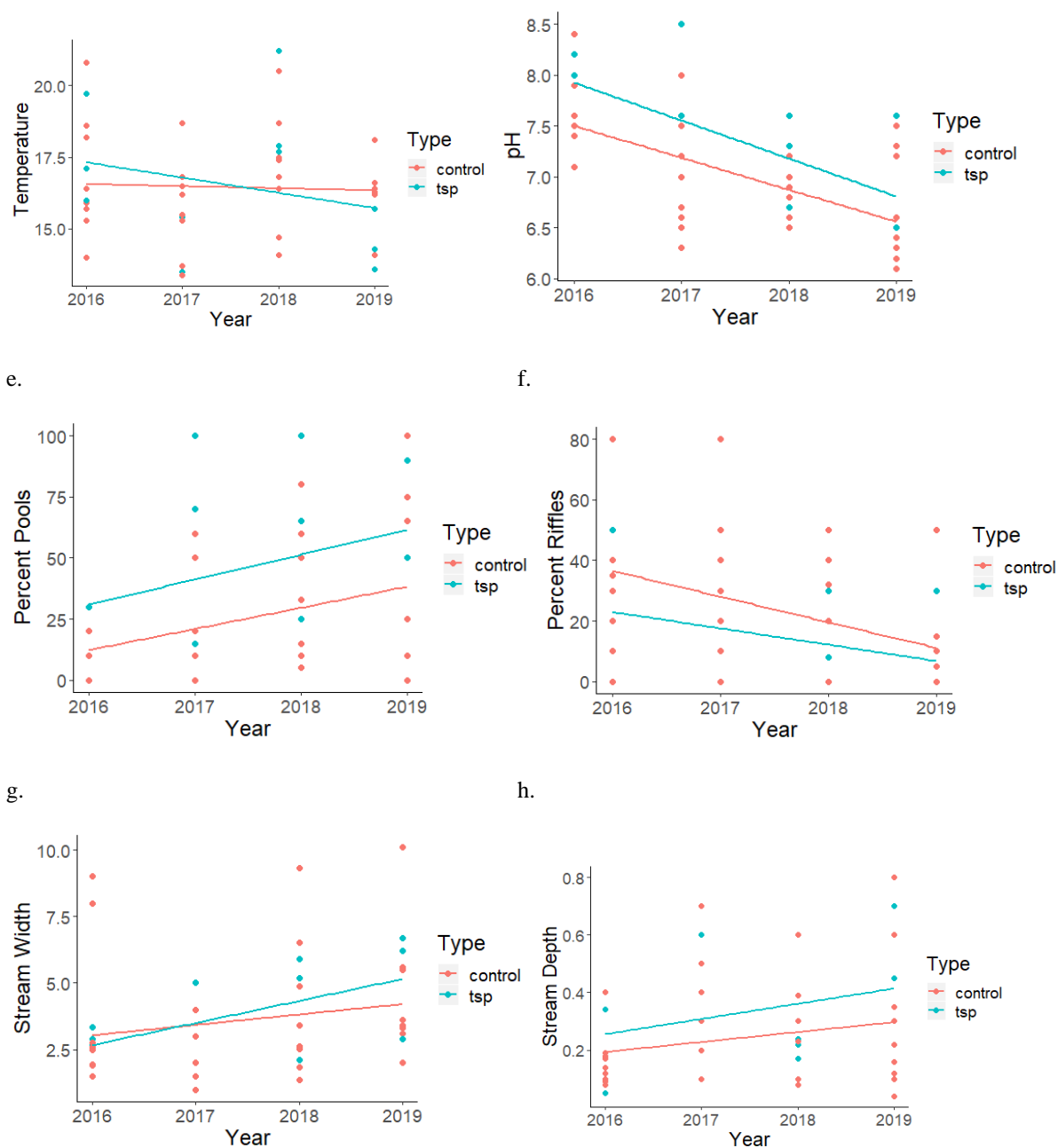


Figure 3. a-g. The relationship between environmental variables and time for control and TSP sites. Chemical habitat variables (a-d) showed some declines, while physical habitat variables (e-h) showed mostly non-significant trends.

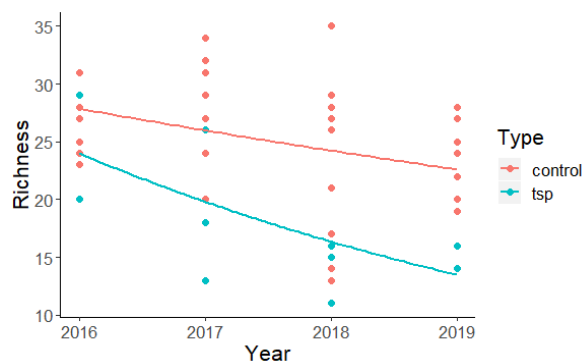
Biodiversity Effects of Restoration

Most macroinvertebrate biodiversity metrics decreased over the course of the study in both TSP and control sites, but several metrics declined more sharply in TSP sites (Fig. 3).

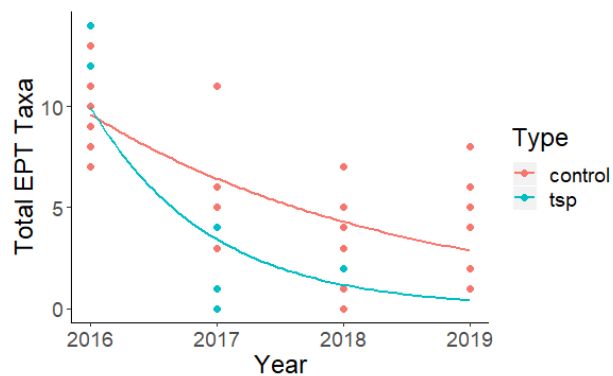
Average macroinvertebrate richness declined by 6.7% (95% CI: 1.1-12.0%) each year in control

sites, whereas TSP sites showed marginally faster ($p = 0.0723$) declines of 17.5% per year (95%CI: 7.0-26.8%). Average richness of sensitive EPT taxa declined even more precipitously in both control and TSP sites. While these sensitive taxa declined by 33.1% (95% CI: 23.8-41.2%) each year in control sites, their declines in TSP sites were significantly ($p = 0.0013$) faster at 65.6% per year (95%CI: 49.5-76.7%). Average Shannon Diversity surprisingly increased over time in control sites, but TSP sites showed no strong change over time. The Shannon diversity at control sites increased by 0.314 each year (95%CI: 0.13-0.50) ($p = 0.0014$), while TSP sites did not change significantly over time ($p = 0.3629$). These changes manifested themselves in the CO MMI, which is combination of biodiversity measures. Average CO MMI showed similar declines ($p = 2.12 \times 10^{-6}$) in both site types with a decline of 7.9 points per year (95%CI: 5.12-10.71) in control sites and 8.6 points per year (95%CI: 3.72-13.40) in TSP sites.

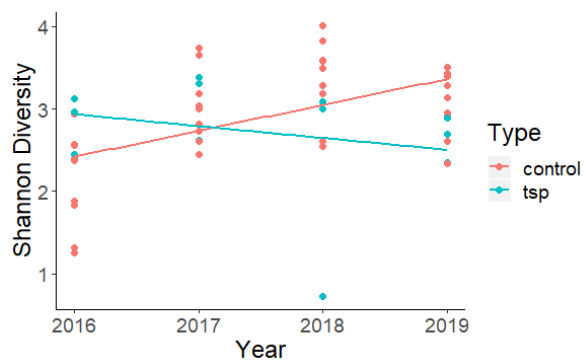
a.



b.



c.



d.

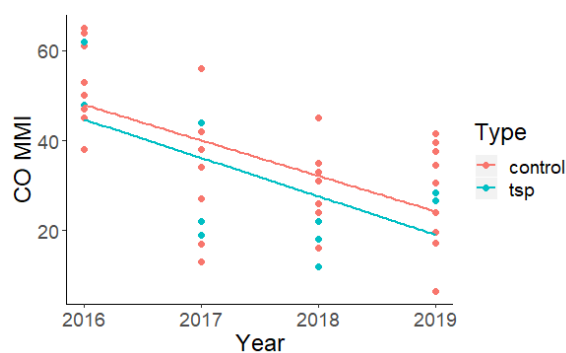


Figure 4. a-d. Macroinvertebrate diversity indices decreased in control and TSP sites over time in all indices except for Shannon diversity. Further, richness and sensitive EPT taxa showed stronger declines over time in tsp sites.

Community Structure Effects of Restoration

Macroinvertebrate community structure differed in TSP and control sites, and changed over time in both site types, but those differences were modest (Figs. 4&5). The NMDS ordination explained 54.6% of the variation in community structure roughly evenly split between two axes (Axis 1: 28.2%, Axis 2: 26.2%). Temporal changes in community structure correlated with the first axis, whereas restoration status aligned more so with the second axis (Figs. 4&5). While 12.5% of the variation in community structure across sites was explained by annual changes ($p = 0.001$), a smaller proportion (9.1%) was explained by restoration status ($p = 0.001$). Temporal change in community structure differed in TSP and control sites ($p = 0.023$), but only accounted for a scant 3.6% in variation. Curiously, none of the biodiversity measures in this study significantly correlated ($p > 0.05$) with community structure. However, four abiotic variables, including EC, total Kjeldahl nitrogen, TDS and width, correlated ($p < 0.05$) with community structure with higher values more likely in unrestored sites than in TSP sites. Physical habitat ($PC1 R^2 = 0.048$, $p = 0.017$) rather than water quality ($PC2 R^2 = 0.031$, $p = 0.138$) tended to dominate these correlations.

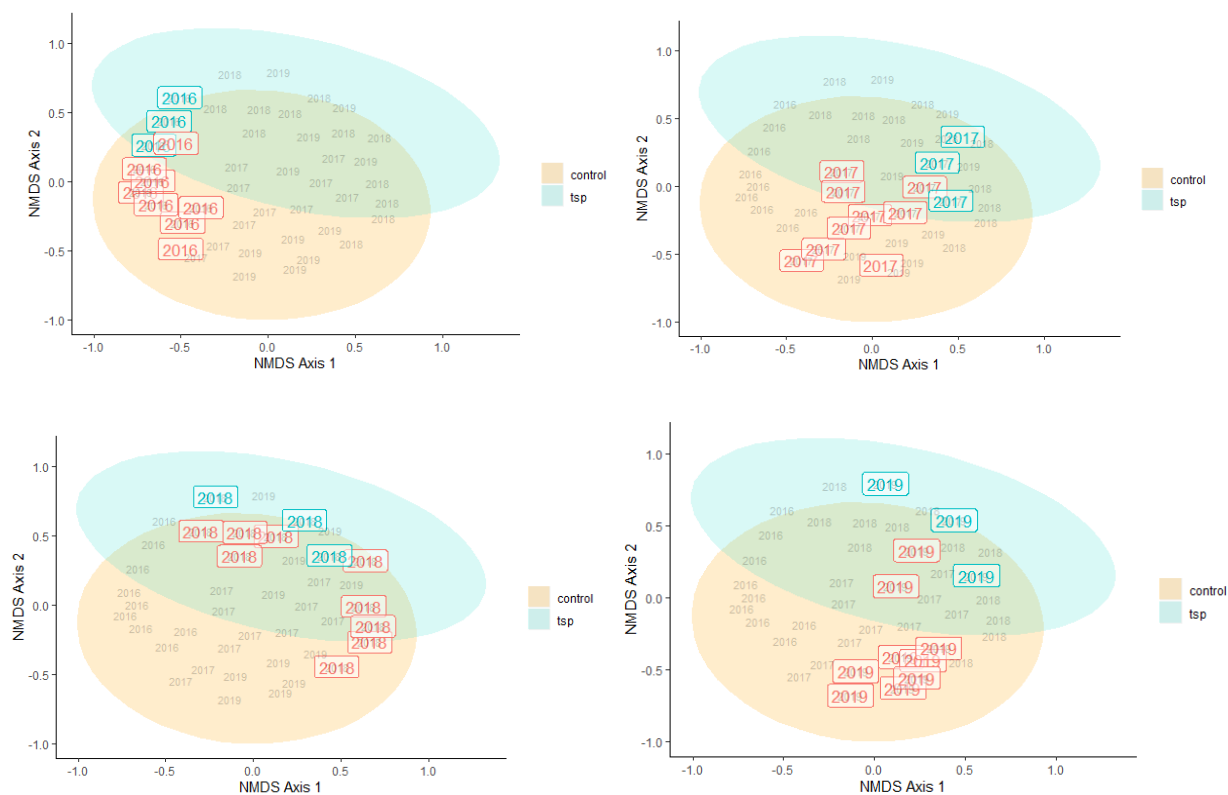


Figure 5. NMDS ordinations showing the change in ordination space over time of sites.

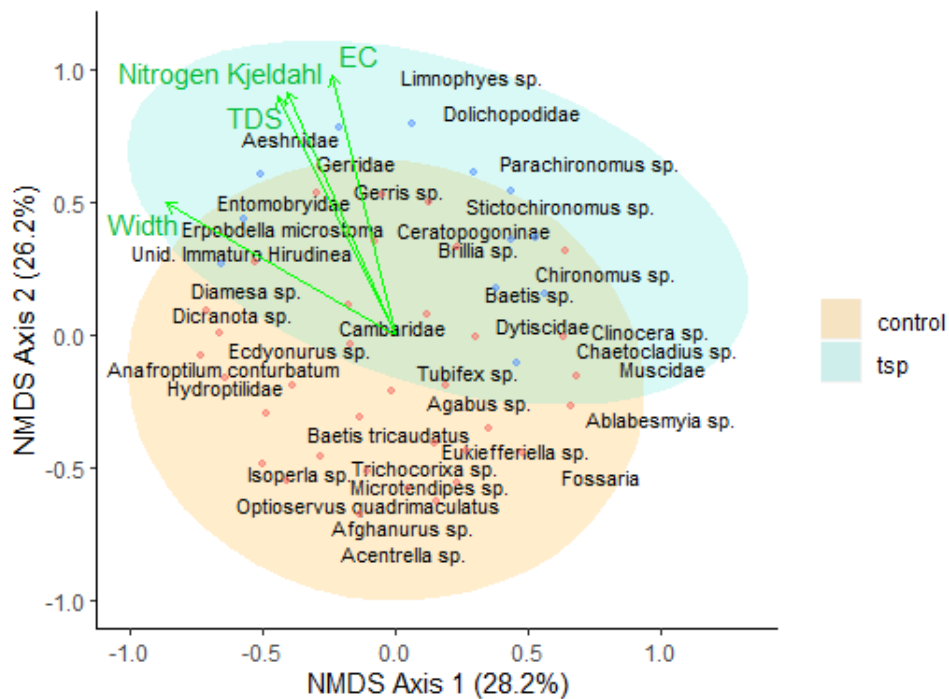
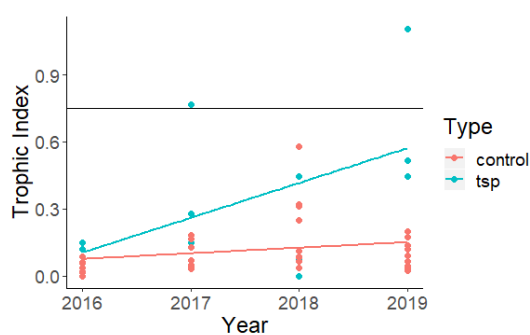


Figure 6. The NMDS ordination results showing macroinvertebrate taxa positions calculated as weighted averages of site scores. Taxa that occur in TSP sites tend to be a subset of the taxa that occur in control sites.

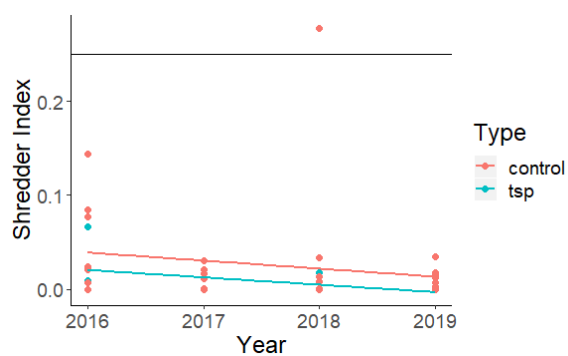
Ecosystem Surrogates

Ecosystem surrogates at control and TSP sites followed similar trends over time, with some exceptions (Fig. 6). The trophic index was stable ($p = 0.3349$) in control sites but increased significantly ($p = 0.0154$) by 0.15 annually (95%CI: 0.07-0.24) in TSP sites, approaching the autotrophic threshold of 0.75. The filtering collector index only had significant changes ($p = 0.0426$) in control sites with an increase of 0.09 per year (95%CI: 0.04-0.14), whereas TSP sites remained flat ($p = 0.9937$). The channel stability index changed significantly over time for both control and TSP sites changes (control: $p = 0.0003$, TSP: $p = 0.0049$). In control sites, the average channel stability index increased by 0.12 points per year (95%CI: -0.04-0.29), whereas TSP sites increased by 0.16 per year (95%CI: 0.05-0.26). The shredder index, however, showed no clear trends over time in either control or TSP (control: $p = 0.2182$, TSP: $p = 0.5096$), but all values but one outlier were below the threshold for abundant coarse particulate matter. The predator-prey index did not change significantly over time (control: $p = 0.7361$, TSP: $p = 0.4166$) remaining within consistent predator-prey levels.

a.

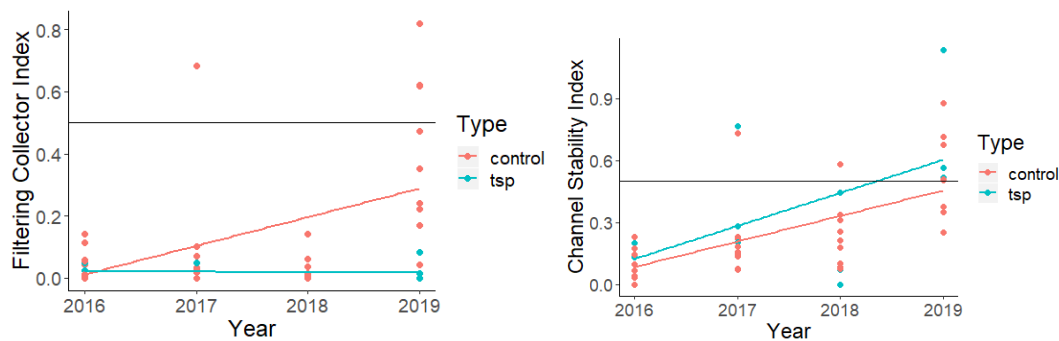


b.



c.

d.



e.

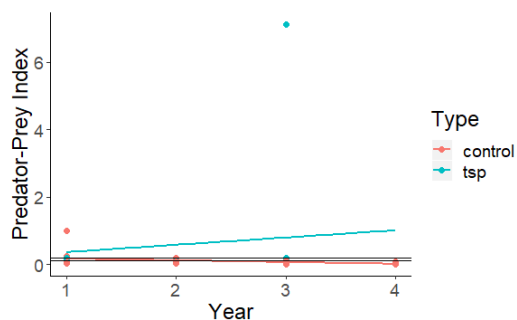


Figure 7. a-e. The relationship between ecosystem surrogates and time between tsp and control sites. Black lines indicate cutoffs for surrogate levels (a. autotrophic $>.75$, heterotrophic $<.75$. b. lack of coarse particulate matter $<.2$, abundant coarse particulate matter $>.2$. c. lack of fine particulate matter $>.5$, abundant fine particulate matter $<.5$. d. stable channel characteristics $>.5$, unstable channel characteristics $<.5$. e. normal predator-prey range $.1-.2$).

Discussion

In-stream restoration activities in Deer Creek did not affect the sites in a predictable way. The changes in physiochemical measures I observed between restored and unrestored sites did not conform to my predictions. While increased ponding did occur in restored sites, ponding also increased in control sites. These changes did result in decreased dissolved oxygen in control sites, but contrary to my predictions, the temperature regime was unaffected in both sites types. Although I expected macroinvertebrate biodiversity to increase at restored sites and stay flat in unrestored sites, I observed declines at both restored and unrestored sites. Declines in both restoration and control sites indicate that regional forces such as drought or urbanization may be stronger drivers of the macroinvertebrate community in Deer Creek. In fact, stronger declines over time and a general lack of sensitive taxa at restored sites indicate that the disturbance caused by the restoration activity itself may exacerbate these regional influences. Though I expected

shredders to increase at restored sites due to increased availability of leaf litter, shredder populations didn't change over time in either restored or unrestored sites. In Deer Creek, in-stream restoration does not appear to be benefiting macroinvertebrate communities.

While some studies show that stream restoration activities improve macroinvertebrate biodiversity my results in Deer Creek are more in-line with studies that show minimal or negative effects of in-stream restoration on macroinvertebrate biodiversity. Restoration activities that improve habitat heterogeneity by removing concrete channeling in streams allows macroinvertebrates to recover. In the Han river in South Korea, the biodiversity of degraded macroinvertebrate communities increased four years after the removal of cement used for channelization (Bae, Kil, & Bae, 2005). On the other hand, in Finland, after 20 years of monitoring an in-stream restoration project in which boulder ridges and flow deflectors were installed to create enhanced in-stream structural habitat diversity, macroinvertebrate communities failed to improve because macroinvertebrates were limited more by water availability than habitat heterogeneity, and because the communities were disrupted by restoration activities (Louhi et al., 2011). Macroinvertebrate biodiversity in Deer Creek not only failed to improve, but instead actually declined after TSP installation. Re-establishment of macroinvertebrates following in-stream disturbance is limited by recruitment and recolonization from nearby stream reaches, making diversity improvement dependent on whether immature macroinvertebrates survive to adulthood (Spänhoff & Arle, 2007). Consequently, Spänhoff & Arle (2007) argue that when large flow obstacles like stones or deflectors are added to streams, macroinvertebrates in young life stages are unable to avoid the disturbances limiting survivorship to adult stages. In Deer Creek, TSPs were installed in 2015 and they had to be reinstalled a year later. Because I observed stronger macroinvertebrate diversity declines in restored sites than in

unrestored sites, the multiple disturbances resulting from TSP installation in Deer Creek actually impaired macroinvertebrate communities.

Because macroinvertebrates declined in both restored and unrestored sites, the macroinvertebrate communities in Deer Creek are likely more strongly controlled by regional factors than the changes to local, in-stream habitat afforded by TSPs. Expecting small physical changes in small parts of the system to have wide-scale impacts is known as the “field of dreams” myth of restoration (Hilderbrand, Watts, & Randle, 2005). Deer Creek is only a small stream in the Upper South Platte watershed that is surrounded by urbanization and agriculture. Urbanized streams have been consistently linked to stream degradation from the urbanization of surrounding areas (Chadwick et al., 2005; Walsh et al., 2005). Habitat restoration on the other hand, is a small-scale resolution to a large-scale problem (Bond & Lake, 2002). In a study of urban stream restoration in North Carolina Piedmont, Violin et al. (2011) found that in-stream restoration practices in degraded reaches, when compared to pristine reaches, did not improve environmental variables like water quality and hydrology that are controlled for at the watershed level. Consequently, biodiversity in restored reaches was similar to that in degraded reaches (Violin et al., 2011). Similarly, Deer Creek flows through degraded, suburban area, so watershed-level impacts may not be improved by the local habitat changes the in-stream restoration structures are providing. Deer Creek further exemplifies the notion that improving only the local physical environment does not mean that community assembly will follow in a predictable pattern.

Similar to constraints on recovery set by the broader watershed context, interannual variability in climate may limit the recovery potential of macroinvertebrates from in-stream restoration activities. Climate drives variation in flow from year to year, and the idiosyncratic

variability of flooding and drought may affect restoration outcomes (Bond & Lake, 2002). After strong rainfall and flooding that destroyed TSP structures in 2016, Deer Creek experienced drought in subsequent years. Nearby USGS stream gauges show that flows in 2016 were much higher than in the following years. Furthermore, in the summer of 2018, newly established transects upstream of the twelve investigated in this study had no water in them at the time of monitoring. Not only do these extreme water events cause shifts in stream biodiversity, they also disrupt links between streams and catchments. This can make monitoring stream restoration challenging due to biotic response delays caused by these events (Reich & Lake 2015; Gunn et al., 2010). For example, although many macroinvertebrates recolonized Ontario streams where acid mine drainage was treated, impairment persisted because sensitive taxa could not thrive during the drought that occurred during the eight years of the study (Gunn et al., 2010). Declines in water availability attributed to climatic variation may be part of the reason that we observe decreases in macroinvertebrate diversity in Deer Creek whether or not sites have TSPs installed.

Adding ecosystem surrogates to the analysis of restoration in Deer Creek adds some insight to how installing TSPs may influence ecosystem function in Deer Creek. Beaver dam analogs such as the TSPs used in this site should trap sediment transported downstream and temporarily increase organic matter (e.g. leaves) retention (Wohl & Beckman, 2014), but lack of improvement in the shredder population does not support this conclusion. Despite the general lack of shredders in the study, both restored and unrestored sites are heterotrophic, meaning they rely more so on allochthonous carbon from the riparian zone. However, in sites where TSPs were installed, the trophic index has shifted towards autotrophy, indicating that there could be less reliance of macroinvertebrates on leaf litter and more reliance on algal resources (Merritt & Cummins, 2006; Whiting et al., 2011). Additionally, restored sites in Deer Creek showed fewer

filterers over time than in control sites. This might be due to limited fine particulate matter escaping downstream as retention increases behind the TSP structures (Merritt & Cummins, 2006, Wohl & Beckman, 2014). Continued interpretation of ecosystem surrogates through functional feeding group analysis in Deer Creek can help us understand how these in-stream structures influence material and energy processing in the stream.

Although in-stream restoration structures have negatively impacted the macroinvertebrate community in Deer Creek, the other assemblages may show conflicting effects of TSP installation. Goals for in-stream restoration are often focused on reconnecting floodplains and creating more heterogeneous habitats for stream species. Watershed-scale experiments have found that structures similar to TSPs increase ecosystem function by reconnecting floodplains and improving habitats for fish species like steelhead that rely on stream complexity (Bouwes et al., 2016). The main goal of the Deer Creek restoration project was to increase connectivity of the river with its floodplain and the vegetative communities in the riparian zone. Vegetative monitoring and analysis at the same transect sites as macroinvertebrates have shown promise for increased floodplain connectivity on vegetation in Deer Creek (R. Hufft, personal communication, Jan. 23, 2020). Similarly, Orr et al. (2020) found that beaver dam analogs encouraged the restoration trajectory for riparian vegetation by increasing willow growth. Though macroinvertebrates aren't benefiting from restoration in Deer Creek, the TSP structures are still meeting the goals of managers. Further, using an assemblage matched to the intended restoration goal is critical for assessing stream restoration success, but using data from multiple assemblages may reveal when restoration strategies have unintended effects on non-target taxa.

Contrary to my predictions, I found that macroinvertebrate diversity decreased across Deer Creek, with even stronger declines at in-stream restoration sites. The overall decline in

macroinvertebrate diversity is attributed to large-scale watershed stressors such as urbanization and climate variability that cannot be overcome by small in-stream structures. The act of installing and reinstalling the TSP structures likely caused greater declines at restoration sites. Thus, it is possible that an expectation of universal biotic improvement after restoration is unwarranted. I recommend that restoration practitioners maintain clear goals on what factors in the ecosystem they want to improve. For macroinvertebrate biodiversity to improve in Deer Creek, other restoration techniques may need to be employed. Specifically, I would suggest a benthic diversity focused restoration plan. Creating stable riverbed substrate is effective at increasing abundance and diversity of macroinvertebrates by increasing the available habitat (Nakano et al., 2008). Additionally, I suggest that managers outline the types of macroinvertebrates they want to increase, and how those increases correlate with ecosystem functions. For example, by focusing on certain functional feeding groups and the ecosystem surrogates they indicate, managers they want to implement or increase in the stream. In-stream restoration projects that do not focus specifically on increasing macroinvertebrate biodiversity and the functions they provide may be overestimating the improvements that in-stream restoration can provide.

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CHAPTER 4. ENVIRONMENTAL STAKEHOLDER ANALYSIS

Pebble Mine Proposal: A Stakeholder Tale of Two Industries

The State of Alaska has a long history in mining that started as early as the 1800s. Alaska was granted statehood in 1959 on the condition that it remain economically self-sufficient by exploiting the land's natural resources including mineral reserves (Reeves, Jungreis, Sellers, & Wilkson, 2009). During the next several decades, surface mining declined because oil drilling had become more profitable. By 1980, after passage of the Alaska National Interest Lands of Conservation Act, interests had moved away from exploiting natural resources to protecting natural resources, so mining activity reached record lows. Mining eventually resurged when it became more economically feasible after deregulations that allowed Americans to freely own and trade gold ultimately increased gold's profitability (Spengler, 2013). Mining deregulation has come at a cost to Alaska's environmental resources: 100% of Alaska's largest mines have had at least one major hazardous waste spill since the early 2000s (Gestring, 2020). Indeed, mining is one of the most environmentally destructive industries because of toxic waste it produces pollutes approximately 40% of watersheds in the western US, including Alaska (Dudka & Adriano, 1997; National Wildlife Federation, n.d). Mine construction and sediment runoff severely degrades downstream aquatic habitats including those of the economically and ecologically important salmon (Hauser, 2007).

Pebble Limited Partnership (PLP), now exclusively owned by Northern Dynasty, proposes to build a mine on the Pebble Deposit in Southeast Alaska. The Pebble Deposit is the world's second largest deposit of gold, copper, and molybdenum that may be worth as much as 500 billion dollars (Bluemink, 2016). The proposed mine would be the largest in North America and would be located in the headwaters of the Kvichak and Nushagak rivers that

discharge into Bristol Bay (Fig. 1). Bristol Bay has one of the

world's largest and most productive salmon runs that supports an economically and ecologically valuable fishery. Therefore, pollution from the mine threatens not only the fishery in Bristol Bay, but also a landscape rich in ecological resources (Hauser, 2007). Throughout the last decade, stakeholders have debated whether the economic shift from fishing to mining is worth the environmental costs. The devastation the mine could cause to both the fishing industry and the area's environmental integrity will likely result in economic losses for nearby residents. The economic losses resulting from environmental degradation outweigh those of the economic gains brought by a new mine because the resources of the mine are finite, and only a small portion of the profits would remain in the local economy. Consequently, I advise against the development of Pebble Mine because any modest gain experienced by the economy would come with large environmental costs that impair the long-term sustainability of the fishing industry.

Very fewer stakeholders want the mine to be constructed, and those that do justify their position on economic arguments whose claims are faulty. Pro-mine stakeholders include Northern Dynasty Mining company and individuals seeking employment in the mining industry.

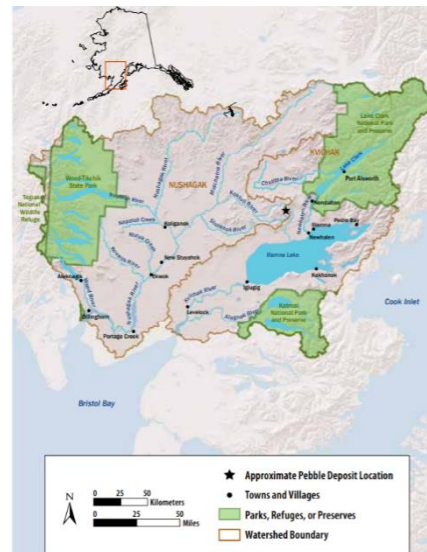


Figure 3. A map of the proposed Pebble Mine and the affected watershed area.

Northern Dynasty expects profits from the Pebble Mine to be around 45-55 million dollars over the next five decades. Those seeking work from the mine can also expect to earn strong wages, but most of these individuals are not local residents. Individuals employed by copper mines earn on average \$67,000 annually (“Copper Mining Salaries”, n.d.); however, of the 1000 workers who participated in early phases of geologic exploration for the Pebble project, only 140 were from the Bristol Bay area (Reeves, Jungreis, Sellers, & Wilkson, 2009). 2000 new jobs will be created during the mine’s initial four-year construction, but this is not as promising as it sounds. If employment follows these trends, only 14% of the jobs will benefit local residents, and the jobs the mine does create would last only as long as the ores remain profitable to mine. The lifetime of copper mines ranges from 5-70 years, but Northern Dynasty proposes a 20-year timeline (NCES, 2013) for the Pebble Mine in Alaska. Thus, the argument that Pebble Mine increases economic prosperity for Bristol Bay is misleading when you follow the money trail.

Mining is economically driven, but environmentally challenging. Mining is problematic because tailings left behind from mineral extraction result in runoff that pollutes downstream ecosystems. Additionally, the deposits at Pebble Mine are low-grade, which means that large amounts of waste material will be produced to get a small amount of ore (Environmental Protection Agency, 2014). Leached acid and heavy metals from improper storage of mine tailings negatively affect ecosystems at all levels, from reduced growth in individual fish and macroinvertebrates to decreased nutrient cycling and energy flow within food webs (Peplow & Edmonds, 2005). Northern Dynasty has proposed to solve the problem of tailings pollution by installing containment pools for wet slurry (Northern Dynasty Minerals Ltd., n.d.).

Unfortunately, containment pools like the ones proposed by Northern Dynasty do not adequately protect downstream reaches because they are prone to structural failure (“Mount Polley Mine

tailings,” 2014). In fact, the engineering firm Northern Dynasty has contracted to create the containment pools at the Pebble Mine is responsible for an environmental disaster that occurred at the Mount Polley Mine in 2014. When the containment pools at Mount Polley Mine breached, tailings were released into the nearby water supply resulting in flooding that caused dramatic changes to channel morphology and increased sediment metal concentrations that may last millennia (Byrne et al., 2018). This environmental disaster outraged local citizens who had previously trusted these tailing storage practices would be safe (“Mount Polley Mine tailings,” 2014). Containment ponds at Pebble Mine may be designed similarly, making it more likely that a parallel breach will occur. Not only are there problems with surface water contamination, but groundwater faces similar threats from contamination. Northern Dynasty claims it will properly mitigate for groundwater contamination using a pit lake to treat groundwater, but 76% of mines that have made similar mitigation promises ultimately fail to meet water quality standards (Hauser, 2007). These contamination issues that result from improper waste handling will likely take a toll on neighboring ecosystems.

Mining construction is detrimental to local ecosystems because of its large and highly destructive footprint. Once built, the mine will cover 77 km² of land currently occupied by pristine terrestrial and aquatic habitat. In addition, water will be extracted from 97 km of headwater streams to supporting mining operations (Hauser, 2007). When headwater flow to downstream estuaries is reduced by tributary dewatering, decreased nutrient loading and increased salinity may decrease estuary productivity (Montagna, Palmer, & Pollack 2013). Not only will mining construction reduce water flows onto the Bay, but it will also accelerate the transport of mining-derived pollutants downstream into the ecologically productive Bristol Bay.

Once these contaminants pollute receiving waters, they threaten the ecologically and economically important salmon fishery (Hauser, 2007).

Construction and mine operations are likely to have large impacts on the local ecosystem through their effects on salmon. Decreases in juvenile salmon growth have been linked to bioaccumulation of copper from mining pollution (Moran et al., 2018). In addition, increased silt transportation downstream from mining can negatively affect recruitment by smothering salmon eggs (Smith, 1940). Because salmon are both ecosystem engineers and keystone species, they are critical for maintaining ecosystem structure. For instance, salmon runs also play a large role in mediating nitrogen and phosphorus cycling (Helfield & Naiman, 2006). They are also an incredibly important food source for other charismatic fauna such as bears, orcas, seals, and birds of prey (Helfield & Naiman, 2006; National Wildlife Federation, n.d.). Thus, if salmon populations are impacted by the mine, there will be likely be large-scale effects on communities and ecosystems that would decrease both ecological function and economic value. If the Pebble Mine is developed, the ecosystem is likely to be disrupted by mining contamination which in turn will cause economic losses to the fishery.

The Pebble Mine project has been met with substantial pushback from stakeholder groups including Alaskan Natives partly because of the projected impacts to the salmon fishery. Beyond providing physical sustenance, salmon are revered in many native Alaskan traditions for their life-sustaining properties, and the threat of harming this important species without regard to these values promotes cultural ignorance (Rahr, 2016). Long before Alaska became a part of the United States, Alaskan Natives have lived alongside the area's rich natural resources including salmon (National Wildlife Federation, n.d.). To this day, Alaskan Natives rely on the area's plentiful salmon harvests for sustenance and economic support (National Wildlife Federation,

n.d.). They fear that the proposed mine will pollute and degrade water resources that are used by humans and animals in the area. Anna Hurley, executive director of the United Tribes of Bristol Bay has stated, “If the environment is harmed, therefore our people are harmed.” Local Alaskans will face many of the negative effects mining has on salmon.

Because of the potential negative effects that mining has on salmon, commercial fishers and processors also strongly opposes the mine. Organizations such as Defend Bristol Bay, a local environmental non-profit that spreads awareness about Pebble Mine’s potential to harm the salmon, has rallied support from local fishers (Defend Bristol Bay, n.d.). These supporters canvas local businesses to gain support by expressing their disapproval of the mine on the grounds that it will diminish the economic value of the salmon industry. In Bristol Bay, the salmon industry brings in 1.5 billion dollars to the economy and 500 million dollars in labor income (Knapp, Guettabi, & Goldsmith, 2013). Working as a third party in the salmon industry in Bristol Bay, I have witnessed the widespread disapproval of Pebble Mine from fishers; not one stakeholder has expressed support for the Pebble Mine project. These fishers fear that their livelihoods will be lost, and that the salmon as an iconic symbol of Alaska will disappear.

Open pit mining in this area can do more than just harm salmon; it can also detract from the environmental integrity of the area. Stakeholders such as tourists, hikers, birdwatchers, sports fishers, and other nature enthusiasts are all likely to be against the mine due to the 77km² footprint of the open pit mine and other required mining infrastructure (Save Bristol Bay, n.d.; Fig. 1). This footprint ultimately detracts from these stakeholders’ goal of connecting with nature, preserving natural resources, and protecting wildlife. Negatively impacted environments are less likely to be visited by these stakeholders who prefer pristine nature. Decreased visitation of local natural areas by recreational enthusiasts may also harm the local economy. The Bristol

Bay Chamber of Commerce has publicly expressed its disapproval of the Pebble Mine for the harm it could cause to local businesses from depressed tourism (Defend Bristol Bay, n.d.).

Mining in the largely untouched Bristol Bay region is a loss for those who enjoy nature.

I recommend against the development of Pebble Mine on the grounds that economic benefits of mining this area pale in comparison to the environmental degradation and economic losses it will likely cause. The loss in sustainable revenue from the salmon industry is not outweighed by the short-term growth that a new mine will bring. The mining industry still has a long way to go before people can trust that mining pollution will not taint pristine local environments. Many stakeholders oppose the mine because the contamination it will produce may irreversibly damage the salmon populations upon which local residents' livelihoods rely. Mining the Pebble Deposit may put money into the economy, but not enough to make up for the economic and ecological loss of one of the last pristine fisheries. Salmon are not just another fish in Bristol Bay, they are a source of food, income, and identity. With so many stakeholders against development, those who are pro-mine are likely few in the tight knit communities of the region. Those against the mine are constantly working to stop Pebble mining development in its tracks, and I have hope that they will succeed (Defend Bristol Bay, n.d., Save Bristol Bay, n.d.).

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