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MS ENVIRONMENTAL BIOLOGY
CAPSTONE PROJECT

by

Kelsey R. F. Morrison

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

REGIS UNIVERSITY
May, 2018

MS ENVIRONMENTAL BIOLOGY
CAPSTONE PROJECT

by

Kelsey R. F. Morrison

has been approved

May, 2018

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CHAPTER 1. LITERATURE REVIEW: SUSTAINING SALMON-DEPENDENT ECOSYSTEMS

The collapse of fish stocks was forecasted decades ago (Gordon, 1954) but has only recently started to enter the awareness of the general public. Pacific salmonids are among the numerous stocks showing dramatic declines in the past century (Nehlsen, Williams & Lichatowich, 1991). Salmonid stock declines are thought to be due to habitat destruction that occurred with the construction of multiple dams in the 1800-1900s that blocked the migration of adult salmon inland from oceans to their freshwater spawning grounds (Groot & Margolis, 1991), as well as poor harvest management during spawning season (Carney & Adkison, 2014). Decline in Pacific salmonid stocks is one of the best known and researched declines because the consequences of salmonid demise are wide ranging: impacting economies, cultures, and ecosystems (Gende, Edwards, Willlson & Wipfli, 2002).

Salmonids are keystone species in the terrestrial ecosystems drained by their spawning streams since they bring many marine nutrients into landlocked habitats, causing changes to riparian community structure and serving as a vital nutrient source to many organisms (Gende et al., 2002). Salmon are also a vital economic resource for the many communities in the Pacific Northwest who have harvested and depended on wild salmon for centuries, and who hold salmon as a symbol of their culture and community (Gresh, Lichatowich & Schoonmaker, 2000; Carney & Adkison, 2014). Between 2000 and 2004, an average 88,000 metric tons of wild-caught Pacific salmon were consumed each year by people in the United States alone (Knapp, America, Roheim & Anderson, 2007), a number dwarfed by the 350,000 tons harvested each year in Alaska alone, much of which is exported (Knapp, 2007) . The economic importance of salmon cannot be ignored, but it also should not be the only consideration when developing policies and strategies for ideal management. There are several management strategies that have proven effective at re-introducing salmon to their historic ranges without reducing harvest rates. These methods should be implemented widely in combination with current strategies to yield the highest economic and ecological returns to both communities and ecosystems.

Most populations of fish recognized as salmon are anadromous, spending different phases of their life in fresh and salt water habitats (Gende et al., 2002). They are born in fresh water streams but migrate to the ocean for most of their adult life (roughly 2-4 years) (Gende et al., 2002). The adult salmon then return to freshwater streams to mate, an event widely known as a salmon run (Gende et al., 2002). Salmon runs, which occur annually, comprise tens of millions of individual fish of various salmonid species; different stocks of each species will return to the same river they were born in, ranging from Alaska down to Northern California (Gende et al., 2002). When adult salmon enter freshwater streams to begin migration to spawning sites, they are full of lipids to give them the energy to swim upstream, sometimes thousands of miles inland (Hendry & Berg, 1999; Gende et al., 2002). Adults that do not die on the journey inland stay in spawning sites for weeks, relying on their stores of lipids and proteins to supplement feeding (Gende et al., 2002). As salmon use these stored nutrients and expel their waste into the stream, they change the surrounding water chemistry - increasing concentrations of nitrogen and phosphate (Donaldson, 1967). This results in augmented algal biomass and riparian plant growth (Gende et al., 2002). Additionally, several terrestrial animals such as predatory mammals and birds gather at spawning grounds to feast on the abundant influx of protein, fat, calcium, and other nutrients brought inland by the salmon (Reimchen et al., 2000; Gende et al., 2002). These terrestrial animals are dependent on this important food source for energy and nutrients to help their over-winter survival, growth, and general health (Reimchen et al., 2000; Gende et al., 2002). As these mammals and birds migrate away from the stream after spawning is complete, they continue the flow of marine nutrients inland (Hilderbrand, Hanley, Robbins & Schwartz, 1999; Gende et al., 2002). Many of these animals are important seed distributers, and without this ecological service the composition of the surrounding plant habitats may be dramatically altered (Hilderbrand et al., 1999; Gende et al., 2002).

The ecological benefits of salmon runs are thwarted if management strategies aim only to increase salmon availability to consumers. Fish farming, often thought to be a solution to declining wild fish stocks (Meffe, 1992; Goldburg & Naylor, 2005), concentrates on this metric exclusively. While farming salmon may reduce the demand for wild-caught stocks, it still requires great oceanic input since salmon are predatory and feeding them requires the harvest of smaller oceanic fish in amounts equal or greater to the weight of the resulting farmed fish itself (Goldburg & Naylor, 2005). Locally, farming salmon can be used to supplement some demand for the fish, but such practices will not allow for the same level of economic benefit to local communities since farming facilities are generally more centralized and owned by companies rather than families (Knapp, 2007). This concentration of the fish farming industry makes it an unviable solution to global stock declines as it would not be as accessible to most of the 3 billion people globally who rely on aquatic sources for 15% of their diet, many of whom are within the poorest regions of the world (Godfray et al., 2010). Salmon fishing communities throughout the Pacific Northwest are among these groups reliant on the natural abundance and accessibility of wild salmon stocks (Carney & Adkison, 2014). Lastly, farmed salmon also do not travel inland or transfer marine nutrients upstream, so while this practice may be beneficial on a small scale, management should aim more towards sustainable wild harvest and increased natural salmon habitats.

Currently there are several different management strategies that are implemented for the harvest of wild salmon, most of which surround escapement goals. Escapement is the number of fish that make it to spawning grounds without being caught by fishers and is usually estimated by comparing the number of fish caught (reported by fishers) to the estimated run size (Carney & Adkison, 2014). There is significant uncertainty surrounding escapement goals, and just as much uncertainty in knowing how many escaped salmon are sufficient to generate a strong generation that will return in numbers large enough for future harvest.

Since salmonids are such an important economic resource, there has been abundant research investigating how many spawning salmon are needed to generate the minimum number of smolt that can be supported in a given area of river habitat (Bradford et al., 2000). A smolt is a juvenile salmon that has undergone the transformations necessary to migrate into the sea, and is usually attained after about a year of living in the freshwater stream in which hatched (Bradford et al., 2000). Typically, these estimates are based on index streams (streams with enough historic run data) and are assumed to be the same for all salmon stocks regardless of species or habitat (Bradford, Myers & Irvine, 2000). The use of index streams makes multiple assumptions, but the simplicity of this method has made it widespread (Bradford et al., 2000). In attempts to better understand the productivity of coho salmon (*Oncorhynchus kisutch*) researchers observed adult female abundance and resultant smolt production (Bradford et al., 2000). By looking at these relationships across multiple coho stocks, they were able to fit a simple model of how many females are needed to seed a 1 km^2 area of river with sufficient eggs to produce the minimum smolts needed for healthy run returns in the future. They found the minimum number of females needed per 1 km^2 of suitable habitat was around 19 (Bradford et al., 2000). Their methods were simple enough that similar models could be developed for each species of salmon, and this would make fewer assumptions about minimum escapement, which may be lower or higher than currently thought depending on the fecundity of each species (Bradford et al., 2000). Locally refined models like this are important, as assumptions can easily lead to over-harvest. However, this only answers half of the escapement question, as the amount of suitable habitat must also be known to determine escapement goals.

Often in estimating escapement, a small segment of river is analyzed to obtain the proportion of habitat suitable for building redds (Groves et al, 2013). Redds are nests built by female salmon during spawning and require certain water depths and sandy bottoms (Groves et al., 2013). This small segment of river is then assumed to represent the entire upstream habitat and these estimates of suitable habitat are then used to estimate how many salmon can be supported (Groves et al., 2013). However, this estimate makes too many assumptions. Another proposed method would be to take aerial photographs that can be used to physically count redds (Groves et al., 2013). When this was done in the Snake river, an important river for chinook salmon (Oncorhynchus tshawytscha), they found that the amount of suitable habitat is lower than what was estimated initially, indicating that those assumptions may not be the best estimate of suitable habitat (Groves et al., 2013). They also found that not all suitable habitats are used every year, indicating that more habitat would be needed to sustain the amount of escaped salmon (Groves et al, 2013). Counting redds is initially labor intensive, but yields better insight into salmon capacities, which will allow for more accurate escapement goals that provide the largest economic gains without compromising the stock size.

Bristol Bay in Southern Alaska is home to some of the largest salmon runs, and the regional economy is built largely off of this resource (Knapp, 2007; Carney & Adkison, 2014). Here they use two methods for limiting catches to meet escapement goals: emergency orders and fixed fishing seasons. When evaluating both of these methods it is clear that both have benefits depending on the size of the fishery. An emergency order will permit or prohibit any fishing based on run-size estimates, giving the fishery managers the power to call off any fishing for a single day, or the entire season if the salmon run is too small (Carney & Adkison, 2014). This method is most beneficial for areas with large economic dependence and where overfishing is likely (Carney & Adkison, 2014). Thus it should be implemented for large fisheries since it won't give unfair consideration to economic over ecological gains. For smaller fisheries, however, the best method is a fixed season. A fixed season allows fishing within set dates regardless of run size (Carney & Adkison, 2014), which is beneficial because there is less urgency to fish on any one date as it is known that fishing will be available the next day. Male and female salmon usually migrate separately, as do different species of salmonids, so spacing out the intensity of fishing can be used to keep natural sex ratios as well as maintain sufficient escapement (Carney & Adkison, 2014).

While both of these management strategies offer benefits to salmon runs by limiting catch, these systems are easily cheated. Top-down control of systems is often less effective because it causes people to feel that they have no responsibility in the success of salmon runs. Another suggested method that could be implemented in areas where control is difficult, would be to implement rights-based catch shares (Costello, Gaines & Lynham, 2008). This method gives individuals incentive to protect salmon since they now are the ones in control over harvest (Costello et al., 2008). Researchers compiled a global database of fishery catch statistics from 1950-2003 from over ten thousand fisheries, and what they found was that implementation of catch shares had the capability to either stop the stock decline entirely, or even reverse it (Costello et al., 2008). This method, however, is unlikely to reverse salmon stock decline, since many salmonid species are habitat limited by dams built across much of their historic range.

In order to regain the ecological benefits brought to terrestrial habitats by salmon runs, we must make it possible for salmon to migrate upstream to their former habitats. Dams were built in many former salmon habitats as a way to hold water in dry areas and generate power via hydroelectric systems. In multiple areas, such as California, 80% of former salmon habitat has been made inaccessible by these barriers (Quinones et al., 2015). Recent public interest in river reconstruction and fisheries has helped push for the removal of dams that are no longer providing substantial electricity or the water storage capacity they once had due to sediment build up (Robbins & Lewis, 2008). However, in the many studies done surrounding dam removal, there are contradicting results in the ability to better support wild salmon runs in undammed rivers. In California a model was built to investigate the effects of removing 8 proposed dams on multiple salmonid species (Quinones et al., 2015). In their model, researchers found that the benefits to multiple salmonid species would greatly outweigh the short-term harm of sediment disturbance, so long as enough time was given to test for salmon recovery (Quinones et al, 2015). Another study regarding dam removal effects on Atlantic salmon found that not only did salmon return, but the economic benefits of recreational fisheries increased with dam removal and outweighed any economic benefit of keeping the dam present (Robbins & Lewis, 2008).

However, not all studies have shown dams to be a hindrance to wild salmon stocks, in one study, researchers found that once dams were modified with fish ladders (small segments of flowing water that allow anadromous to swim up them) survival of smolts migrating back to sea was actually greater in rivers with dams present compared to rivers without dams (Welch et al., 2008). While the results of this study are surprising, it is important to remember that they only looked at half the equation: smolts returning to the sea. There is a paucity of research regarding the success of adult salmon migrating upstream to spawn in modified-dammed habitat, but it has been shown to be lower in studies where dam removal has been modeled or observed (Quinones et al., 2015; Robbins & Lewis, 2008). However, without research specifically studying migration success in streams with modified dams, it is hard to make conclusions on their efficacy. Research in this area is critical, as it is unlikely that every dam will be removed, but modification may allow for some habitat reconstruction.

With dams that are removed, there will need to be additional effort to reintroduce species to habitats that have been excluded from salmonid spawning for centuries. Because salmon return to the same spawning site that they hatched in, their rates of reintroduction to newly available habitat may be slow, and will vary depending on species. New habitat is only colonized when individual species stray from spawning in their birthplace, an event that occurs in 2-10% of salmon depending on the species (Pess, Quinn, Gephard & Sanders, 2014). Species with high straying rates such as steelhead salmon (*Oncorhynchus mykiss*) will be most benefitted by dam removal since they will likely use new habitat long before any other salmon species encounter it (Pess et al., 2014). But in species with low straying rates, such as sockeye salmon (*Oncorhynchus nerka*), the benefits of dam removal will only be seen in the long term if they are unaided in finding the new habitat (Pess et al., 2014). For salmon with low rates of straying, spawning adults may need to be manually transported upstream to new habitats to see any short-term benefit to barrier removal (Pess et al., 2014).

Dam removal could also cause new problems, since the historic habitats they block may no longer be suitable for salmonid species. A lot can change in a habitat when a keystone species such as salmonids are removed for an extended period of time. Without salmon runs the water chemistry and surrounding riparian habitats may be substantially changed (Gende et al, 2002), possibly enough to make the habitats unsuitable for salmon. Additionally, new species such as brook trout (*Salvelinus fontinalis*) have been introduced and it is unknown if they will out-compete young salmon (Warren, Harvey, McClure & Sanderson 2014). In one study researchers modeled three different management practices and their effects on juvenile chinook salmon survival: (1) placing salmon carcasses in the rivers/along shores to simulate nutrient output of historically large salmon runs, (2) eliminating non-native competitors such as brook trout, and (3) stocking rivers with hatchery chinook to supplement historic production (Warren et al., 2014). They found that placing salmon carcasses yielded the most positive response in juveniles, which also correlated with increased production of periphyton (Warren et al., 2014). Eliminating brook trout had no effect on juveniles, and stocking rivers with hatchery juveniles had a negative correlation with survival of wild juveniles (Warren et al., 2014).

The results of the salmon carcass method may seem impractical and almost wasteful, but it is important to remember that this method would only need to be used until runs naturally become larger. This method has been widely tested, and in multiple cases has been shown to have at least moderate effects on the surrounding ecosystem and periphyton growth (Claeson, Compton & Bisson, 2006; Kohler, Rugenski & Taki, 2008). There has also been significant research to indicate that the use of hatchery juveniles is detrimental to wild salmon stocks (Meffe, 1992), likely because of increased competition.

There is a wealth of methods available to manage salmon stocks, but few are implemented on a large scale. Strategies that have proven effective at increasing stock abundance such as fishing rights, dam removal, and habitat restoration are often underutilized and unfunded; but with so many global fish stocks on the brink of collapse, innovative methods must be used. Methods that increase salmon habitat will be both ecologically and economically beneficial, as they will allow for the greatest harvest without depriving ecosystems of a keystone species.

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CHAPTER 2. GRANT PROPOSAL: IMPACTS OF SALMON REINTRODUCTION ON HERBACEIOUS VEGETATION COMPOSITION AND BIOMASS

Abstract

Historically, salmon have provided essential marine nutrients to land-locked habitats during their spawning migration. Many plant and animal species are dependent on this source of nutrients for survival and growth, and their absence has been shown to dramatically alter these terrestrial communities. Dams have been built throughout much of the historic range of salmon, cutting terrestrial ecosystems off from the flow of marine nutrients they depend on, leading to degradation of wetland habitats and reduced predator abundance. With time, however, concern for the decreased salmon abundance has resulted in organized efforts to remove dams in historically important salmon spawning areas, but the effects of removal on stream-associated ecosystems remain unknown.

The Rogue River in southern Oregon offers a unique opportunity to investigate the impact of salmon reintroduction on surrounding herbaceous vegetation since it has had eight dams removed since 2003, but also has remaining dammed portions that can serve as controls. I predict that herbaceous plant species richness and above-ground biomass will increase in areas where salmon are reintroduced compared to areas where the rivers remain dammed. By comparing herbaceous vegetation richness and biomass along two tributary streams where salmon have been reintroduced to two tributary streams where dams remain, I will be able to investigate differences in community structure and determine if salmon reintroduction has had positive impacts on formally degraded habitats. Wetlands, such as those surrounding tributary streams, provide

essential habitat for numerous plant, bird, and insect species, and their health is imperative to maintaining surrounding ecosystems that are functional and productive.

Background/Rationale/Significance

Salmon are keystone species in the terrestrial ecosystems drained by their spawning streams since they bring marine nutrients into landlocked habitats, causing changes to riparian community structure and serving as a vital nutrient source to various organisms (Gende, Edwards, Willlson & Wipfli, 2002). Riparian communities are habitats that border wetlands, such as river banks or lake shores. Most populations of fish recognized as salmon are anadromous, meaning they spend distinct phases of their life in fresh and salt water habitats (Gende et al., 2002). They hatch in freshwater streams but migrate to the ocean for most of their adult life (roughly 2-4 years) (Gende et al., 2002). The adult salmon return to freshwater streams to mate, an event known as a salmon run (Gende et al., 2002). Salmon runs, which occur annually, comprise tens of millions of individual fish of various salmonid species; different stocks of each species will return to the same river they hatched in, ranging from Northern California to Alaska (Gende et al., 2002).

When adult salmon enter freshwater streams to begin migration to spawning sites, they are full of lipids (fats) to give them the energy to swim upstream, sometimes thousands of miles inland (Hendry & Berg, 1999; Gende et al., 2002). Adults that survive the journey inland stay in spawning sites for weeks, relying on their stores of lipids and proteins to supplement feeding (Gende et al., 2002). As salmon use these stored nutrients and expel their waste into the stream, they change the surrounding water chemistry, increasing concentrations of nitrogen and phosphate (Donaldson, 1967). This results in augmented algal biomass and riparian plant growth (Gende et al., 2002).

Several terrestrial animals such as predatory mammals and birds gather at spawning grounds to feast on the abundant influx of protein, fat, calcium, and other nutrients brought inland by the salmon (Reimchen et al., 2000; Gende et al., 2002). These animals are dependent on this food source for energy and nutrients to help their over-winter survival, growth, and general health (Reimchen et al., 2000; Gende et al., 2002). As these mammals and birds migrate away from the stream after spawning is complete, they continue the flow of marine nutrients inland (Hilderbrand, Hanley, Robbins & Schwartz, 1999; Gende et al., 2002). Many of these animals are seed dispersers, meaning they eat seeds of plants and later expel them in waste in another location. Without this ecological service the composition of the surrounding plant habitats may be dramatically altered (Hilderbrand et al., 1999; Gende et al., 2002).

The importance of salmonids in terrestrial ecosystems was not fully realized until after the construction of dams in the 1800-1900s, which blocked seasonal migration of salmonids (Groot & Margolis, 1991). In multiple areas along the Pacific coast, up to 80% of former salmon habitat has been made inaccessible by these barriers (Quinones et al., 2015). In the absence of these annual migrations, the chemistry of spawning streams stopped receiving an influx of nitrogen and phosphorus, resulting in decreased algal and plant biomass (Gende et al., 2002; Bilby et al., 2003). Predatory mammal and bird populations may have also declined in these areas, as they depend on this important food source (Gende et al., 2002). With fewer predators it is likely that changes occurred to plant community structure beyond riparian habitat since many plants rely on these animals to act as seed dispersers (Hilderbrand et al., 1999; Gende et al., 2002).

Findings on the role of salmonids on the greater terrestrial habitats of the Pacific Northwest and the rapid decline of these economically important fish stocks has sparked public interest in river reconstruction and has helped push for the removal of dams that are no longer generating substantial electricity, or those that lack the water storage capacity they once had due to sediment accumulation (Robbins & Lewis, 2008). However, it remains unknown whether the reintroduction of salmonids has the potential to reverse habitat degradation caused by dam construction that blocks these keystone species from their annual migration. Research on the changes to surrounding habitat and implications of reintroducing salmonids will provide insight into the ability of salmonids to bring the same ecosystem services they are known to provide in river systems that have remained undammed. This allows for more careful consideration and understanding of any benefits of dam removal for upstream habitats and whether such action is worth the effort.

I aim to investigate this topic by comparing upstream riparian habitats in rivers that have been recently undammed to those remaining dammed to determine if there is a significant change in the plant density and plant species richness in the habitat gradient surrounding each river after reintroduction of salmonid species. Since salmonids are known to boost concentrations of nitrogen and phosphorus, two nutrients that often limit the growth and abundance of plants (Gende et al., 2002), it is likely that above-ground density of riparian plants will increase after reintroduction. Additionally, reintroducing salmonids will result in a surge of food for seed-dispersing predators which is expected to escalate their abundance in terrestrial habitats surrounding spawning streams. With increased seed-disperser presence it is possible that new species of plants will spread across and outside of riparian habitats, ultimately raising plant species richness throughout these areas. By measuring above-ground plant mass and species richness at incremental distances from the water's edge, I can determine how far-reaching the impacts of salmon reintroduction are on plant communities.

With the decline of natural habitats around the world, understanding ways we can reverse human-induced change is vital. As salmonids are keystone species, they offer critical benefits to terrestrial habitats that surround their spawning streams (Gende et al., 2002) and their reintroduction may be key to revitalizing important riparian habitats that have degraded in their absence. Riparian habitats provide important ecosystem services ranging from filtering pollutants out of the water and stabilizing banks, to sheltering various insect and bird species (Gregory, Swanson, McKee & Cummins, 1991). Increasing vegetation density and diversity allows these ecosystems to better perform these services (Gregory et al., 1991) which should be a top priority as habitats face greater change from human-influenced degradation.

Regis University challenges students to "learn proficiently, think logically and critically, identify and choose personal standards of value, and be socially responsible" (Regis Mission Statement). Ecosystems across the globe are in peril, and research on how to better restore these habitats is essential so that we can take on the responsibility to reverse the damage we have caused. Reintroducing salmon to their historic range has the potential to revitalize upstream riparian habitats, but better research is needed to understand whether these changes are likely, or if other methods will be needed. My proposed research aims to investigate this topic further to provide insight into how we can best enhance upstream riparian habitats.

Purpose and Specific Aims

Dams built throughout the Pacific Northwest have made many areas of historic salmonid habitat unavailable. Some of these upstream areas have not been visited by salmon for over 100 years, which has led to changes in riparian plant composition (Quinones et al., 2015). The objective of this study is to determine how reintroduction of salmon to previously-dammed habitat will change surrounding plant community structure. Salmonids change the water chemistry in spawning streams, which is linked to increased algal biomass and riparian plant growth (Gende et al., 2002), but the effects of salmonid reintroduction to up-stream habitats are not fully understood. I have chosen to focus on herbaceous vegetation because it is unlikely that changes have occurred in slower-growing woody plants during the brief period since the dams were removed. I predict that herbaceous plant species richness will increase in areas where salmon are reintroduced compared to areas where the rivers remain dammed, since salmon provide food for many important seed-dispersing animals that will bring with them new plant species (Gende et al., 2002). I also hypothesize that the reintroduction of salmon to their up-river habitats will result in increased above-ground herbaceous riparian plant mass compared to rivers that remain inaccessible, as the salmon will increase the nitrogen and phosphorous input in the stream (Gende et al., 2002).

Methods

Study Site

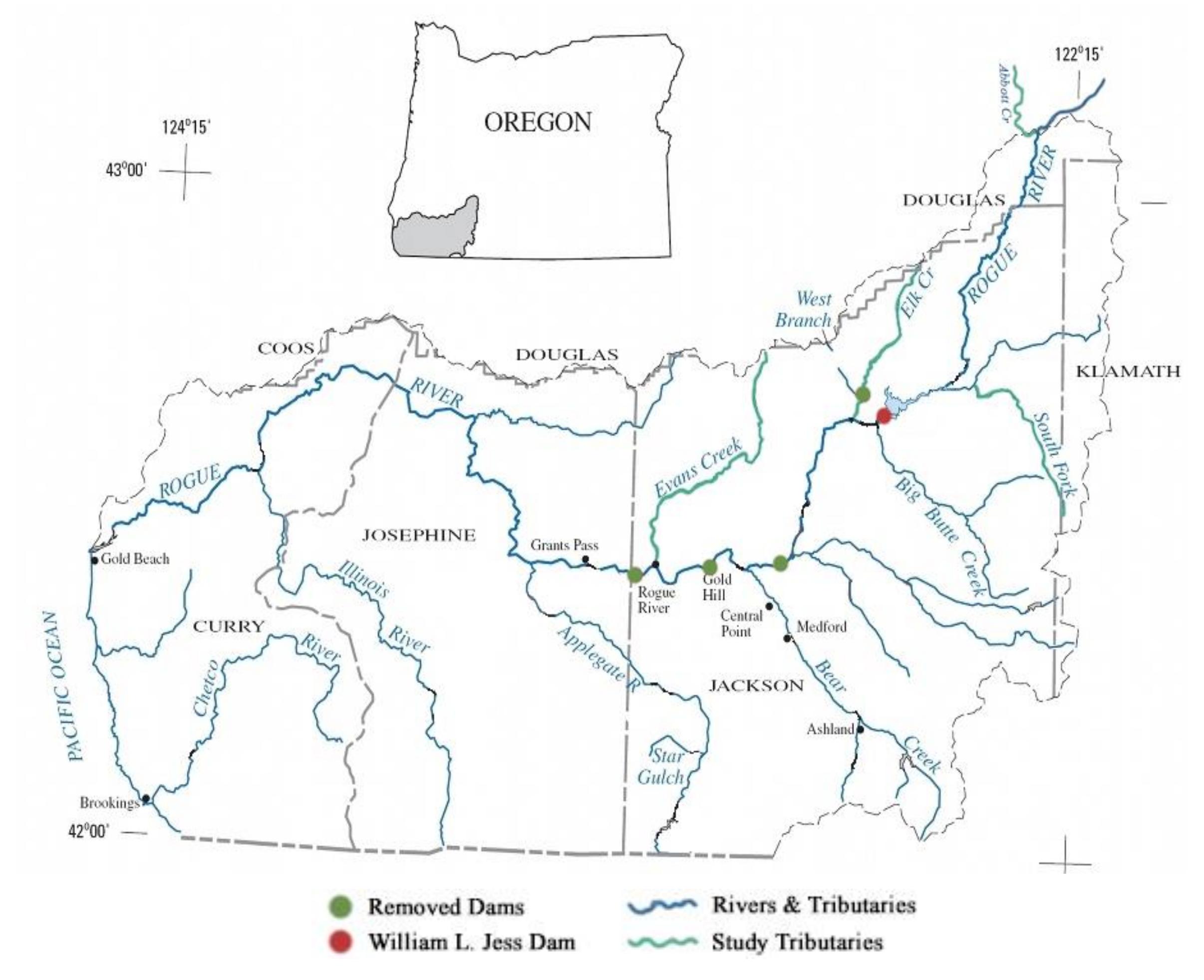

Figure 1: Map of Rogue River Basin with relevant dams & study sites marked (adapted from Water Watch, 2015).

The Rogue River in southern Oregon is home to fall chinook salmon (*Oncorhynchus tshawytscha*) and coho salmon (*Oncorhynchus kisutch*) and historically had one of the largest salmon runs in the state, second only to the Columbia River (Water Watch, 2015; American Rivers, 2016). Throughout the 1800s-1900s multiple dams were built within this river system to save water for agricultural use, but public dismay over the reduced salmon runs resulted in the removal of eight of these dams between the Lower and Upper Rogue Basins since 2003 (McDermott, 2016). Demolition of these dams has opened over 330 miles of free-flowing river habitat between the Rogue River and its tributary streams (American Rivers, 2016). The lowest dam to remain intact within this river system is the William L. Jess Dam, situated at the base of Lost Creek Lake (McDermott, 2016). This 345ft tall dam, constructed between 1970-1974, is unsurpassable to anadromous fish because no fish ladders were built to allow for their passage (McDermott, 2016). This river system provides the perfect opportunity to investigate the effects of salmon reintroduction on surrounding riparian habitat since it provides dammed and undammed sites within the same ecosystem which can be easily compared.

To keep sites as similar as possible, I will sample from four tributary streams with comparable discharge: Evans Creek, Elk Creek, South Fork Rogue River, and Abbott Creek (Figure 1). Both Evans Creek and Elk Creek are free of dams all the way through to the Pacific Ocean, and have been since 2015 and 2008 respectively. Both creeks have had returns of salmon runs following dam removal (McDermott, 2016). South Fork Rogue River and Abbott Creek are upstream of the William L. Jess Dam, and have not had salmonid spawning migrations in over 30 years. These creeks range in elevation from 991ft (Evans Creek) to 3100ft (Abbott Creek), and are all located within the Cascade Mountains (Rogue River Keeper, n.d.).

To determine how far from the river bank changes in vegetation can be observed, I will implement a method similar to that outlined in the Hegazy, El-Demerdash & Hosni (1998) paper, wherein ten 200m transects will be set up around each of the four tributaries. The first transect will be placed 100m from the mouth of each creek, then every 500m upstream from the previous transect. The start of each transect will be placed at the current water's edge, and extend 100m in a random direction away from the shore. The second half of the transect will start on the opposite river edge, and extend 100m in the other direction. One-square-meter quadrats will be placed every 10m along each transect, starting at 0m. I will collect all data in the month of July when snow has melted and most plants will be established for the season (Rogue River Keeper, n.d.).

Plant Species Richness & Diversity

Within each quadrat, all plant species will be identified to the lowest taxonomic ranking possible without risking uncertainty (Hegazy et al., 1998). Species richness will be determined, and diversity will be statistically evaluated using the Shannon-Wiener Diversity Index since it is more sensitive to species compositional changes compared to other diversity indices such as Simpson diversity (Ke et al., 2017). To compare species diversity and richness between dammed and undammed tributaries, a Student's T-test will be run between the overall Shannon-Wiener Diversity indices (among all quadrats of either river category) as well as species richness counts to determine if there is a significant difference between river categories. To determine how far from the water's edge differences in community composition extend, quadrats of the same distance from the water's edge (for example, all quadrats 10m from the edge) will be compared between both river categories using a Student's t-tests. Generalized linear models will also be used to determine the relationship of richness or diversity as a function of distance; the slopes will be compared to detect significant differences in change of plant composition along this gradient between site categories.

Plant Biomass

Once all species richness measurements are recorded at each quadrat, all above-ground herbaceous vegetation will be cut down to the soil, collected, and dried before being weighed to determine the above-ground plant biomass. Drying the plant material will provide a better estimate of the true biomass, because water levels in plant tissues can drastically change throughout the day (Franks & Goings, 2016). These measurements will be compared between site types in the same way outlined for the species richness and diversity above.

Work Plan

June 15-20: Purchase all needed supplies

June 28-30: Driving from Denver to Joseph H. Stewart State Recreation Area

July 1-28: Data collection will happen each day (completing 20 quadrats/day). I will work 5 days, then have 2 days off (in total I will work on data collection 20 days, and have 8 days off).

July 28-31: Drive back to Denver

August-September: Data analysis

October-November: Write up Project

Budget

Items **(please itemize amounts below)**	**Funds Requested from URSC**	**Funds Requested from Other Sources**	**Source of Other Funds**
Equipment (Non-consumables)			
1m^2 Quadrat	$0		
100m Measuring Tape (Keson OTR Fiberglass Tape (100m) from Tiger Supplies)	$0		
Tarps (20x Brown Poly Tarp 10' x 10' from Tarps Plus)	$160 ($8/each x20)		
Scale (Crane Scale Hanging Scale Digital Professional 660 Lb 300 Kg With Accurate Sensor For Farm Fishing Hunting from Amazon.com)	$39.99		
Garden shears (Felco 300 Bypass Picking and Trimming Snip F300 from Felco)	$15.99		
Plant Identification Book (Plants of the Pacific Northwest Coast from Amazon.com)	$20.80		
Tent (REI Co-op Half Dome 2 Tent from REI)	$199.00		
Other			
Gas to Drive to/around Rogue River from Denver (assuming $2.057/gallon, and 30mpg. 1,269 miles between Denver and Rogue River (x2) + 600 miles of driving up and down river)		$215.16	NIH
Mileage on Car (assuming 3,138 miles total, at $0.51/mile)		$1,600.38	NIH
Camping Site (Joseph H. Stewart State Recreation Area for 30 nights)		$510	NIH

Budget Justification

Quadrats and tape measures can be borrowed from Regis University, and thus do not need to be purchased. I will need garden shears to cut down the vegetation for measuring plant biomass. I need 20 tarps because the cut plants from each quadrat will have to be dried separately, and I will be doing 20 quadrats per day. I need a scale to weigh biomass on-site. The Biology Department at Regis does not have shears, tarps, or scales that I would be able to take with me, thus they need to be purchased. To identify plants correctly, it is important to have a plant id book for the specific region where I will be. I will not have access to WIFI or cellular service in many field areas, thus I need a physical book to bring. Since I will be camping, I need a tent, but I own all the other camping gear needed.

Relevance to Current Coursework

I am currently working towards my MS in Environmental Biology with a strong interest in fishery management. This project would help me gain insight into the benefits of fish on terrestrial ecosystems, which is important to consider when deciding on management strategies. Undertaking this project would give me valuable field and data analysis experience, as well as practice writing experimental papers, which are all essential skills for scientific careers.

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CHAPTER 3. JOURNAL MANUSCRIPT: DESIGNING URBAN PARKS AS HABITAT ISLANDS FOR NATIVE BIRDS OF DENVER, CO

Abstract

Habitat loss is the main cause of bird declines across the globe, and much of this habitat loss is a result of urbanization. Urban parks can serve as habitat islands within city sprawl, effectively reducing the distance between less disturbed areas. However, most parks are designed with the intent of human recreation and few studies have evaluated which environmental features are preferred by birds of different types. Using publicly available data, I modeled bird communities as a function of greenspace, water, connectivity and park area. I found that each group was influenced by different habitat variables, and that any one single variable did not have the same influence across all bird groups. Passerine species were positively affected by percent greenspace and drainage density, while these same predictors had negative impacts on waterfowl species. In line with prior literature, connectivity variables were found to positively influence total bird species richness as well as proportion of passerine birds. These findings highlight the importance of having diverse habitat types within urban park systems and emphasizes the need of connectivity between parks.

Introduction

Global populations are rapidly increasing, resulting in accelerated development and urbanization within cities (Keßler & Marcotullio, 2017). Widespread urbanization fragments and disturbs wildlife habitat (Grimm et al., 2008) and amplifies the disturbances wildlife experience (Rebele, 1994; Alberti, 2015). Urbanization has negatively impacted birds and, in many cities, reduced species richness and abundance (Batáry, Kurucz, Suarez-Rubio & Chamberlain, 2017).

Urban parks are often thought of as habitat patches within developed landscapes, but studies of their effects on urban bird abundance and diversity have shown a range of effects, some of which are negative (Batáry et al., 2017). Urban parks are primarily designed for recreational use by people, and thus are often underutilized by wildlife (Batáry et al., 2017), but as humanity expands across once-wild habitats, it is vital to find ways to design urban spaces to accommodate displaced species and effectively connect undeveloped land.

Birds play crucial roles in pest control, seed dispersal, and nutrient cycling (Şekerciuğlu, Daily & Ehrlich, 2004), but their populations are declining faster than most other assemblages. While birds are among the most studied taxa in the world, few studies have explored how urban park design can affect native bird populations. Humanity depends on ecological services provided by birds, and keeping their populations healthy will, in turn, benefit us. Birds aid our agricultural production by controlling insect and rodent pests, and pollinating important crops (Şekercioğlu et al., 2004; Whelan, Wenny & Marquis, 2008). Birds also shape plant community composition by dispersing seeds into appropriate habitats. Consequently, declines in populations of several bird species have resulted in the demise of several co-evolved plant species (Şekercioğlu et al., 2004; Whelan et al., 2008). Birds also transport nutrients, sometimes long distances, to nutrient-poor habitats (Whelan et al., 2008). For example, raptors along the Northwestern United States consume marine fishes and later deposit nutrients into land-locked habitats where these resources are otherwise not abundantly available (Gende, Edwards, Willson & Wipfli, 2002). These long-distance nutrient-delivery pathways are disrupted by urban expansion that degrades and fragments habitat for resident and migrant birds.

Careful urban planning and park design can ameliorate some of the negative consequences of habitat loss on bird species. Planning for parks and open spaces within the urban matrix creates links which bird species can use to connect their territories in undeveloped habitats (Savard, Clergeau & Mennechez, 1999; Fernández-Juricic & Jokimäki, 2000; Cabeza & Moilanen, 2001; Clergeau, Jokimäki & Savard, 2001). Links between habitat are not only important for genetic exchange between bird populations that might otherwise be blocked by development or sprawl, but also yield greater alpha diversity both within parks and gamma diversity across the city (Savard et al., 1999; Clergeau et al., 2001). Habitat islands also increase the accessibility of certain bird species and provide essential nesting and resting grounds for migratory birds (Savard et al., 1999; Cabeza & Moilanen, 2001).

Some park attributes, such as size, are inherently likely to increase bird abundance and richness (Fernández-Juricic & Jokimäki, 2000), but building large parks is not always feasible, especially in already-developed areas. For this reason, other park attributes that result in greater bird abundance and diversity need to be determined. In lieu of larger parks, the surrounding land use and percentage of open space may effectively expand park boundaries (Carbó-Ramírez & Zuria, 2010). Trees in adjacent property may expand habitat by providing space for birds to nest and decreasing the effective distance between neighboring parks (Carbó-Ramírez & Zuria, 2010; de Castro Pena et al., 2017).

Trees within parks also provide space and protection for nesting birds, and it has been found that tree abundance and canopy complexity within parks positively correlate with bird abundance and diversity (de Castro Pena et al., 2017). However, parks often contain tree monocultures planted around the same time, thereby limiting the canopy complexity since it results in trees of virtually identical size (Batáry et al., 2017). Canopy complexity facilitates bird diversity in some ecosystems such as Kibale National Park in Uganda (Şekercioğlu, 2002), but this may not be of equal importance in temperate ecosystems with different native vegetation. Some birds also prefer open space as it allows for them to either spot prey or predators from a distance (McCaffrey & Mannan, 2012), so parks that contain a mixture of both covered and open ground may have higher bird diversity. Evaluating the preferred habitat spaces of birds residing within urban areas can guide management and design of parks to minimize disturbance on bird populations.

Not all parks provide suitable habitat for birds, and many features of city parks deter bird establishment. Many urban parks lack sufficient tree canopy cover, native vegetation, water, and space, or are surrounded by dense buildings or busy roads (Carbó-Ramírez & Zuria, 2010). Additionally, different birds may use park space differently, so features that increase abundance or diversity of one species group may have a negative or neutral effect on another (Sandström, Angelstam & Mikusiński, 2006). Most birds are adapted to specific nesting behaviors that require suitable habitat types; for example, many waterfowl nest on the ground within dense riparian vegetation, while most passerine species nest in branches or holes of trees (Sibley, 2001). Furthermore, birds source food in very different ways, most passerine species feed on insects, fruits, while raptors feed on rodents and small birds, and waterfowl feed on aquatic vegetation or invertebrates (Sibley, 2001). These feeding and nesting habits are specific to habitat types, which in turn can mean that different bird species have different environmental preferences when choosing parks to reside within. Determining which park features increase bird occupancy, both overall and within different species categories, will allow cities to minimize their impacts on bird ranges and the negative effects of urban development on bird species.

To investigate which urban park attributes contribute to bird community structure, I chose the City and County of Denver, located on the eastern range of the Rocky Mountains of Colorado. This area is rapidly developing, but has invested great interest in the wellbeing of birds within urban spaces. Denver has recently joined the Urban Bird Treaty, which aims to reduce negative impacts on native bird species (US Fish and Wildlife Services, 2014). With this commitment, the city needs to improve park conditions for bird habitat. The lack of studies on which park attributes are preferred by various bird types, especially along the Front Range of Colorado, makes the design of suitable parks in the area difficult.

To evaluate what features of urban parks in Denver correspond to different bird communities, I modeled the total richness and abundance of birds as well as the proportion of types of birds (e.g. migratory, native, passerine, waterfowl or raptor), as a function of park vegetation, connectivity, water bodies and size. Passerine species depend mainly on trees and ground vegetation for nesting and feeding grounds, while lakes and riparian vegetation serve these purposes for waterfowl (Sibley, 2001), thus I hypothesize that proportion of greenspace, vegetation complexity, and percent canopy cover will have the most significant influence on passerine species, while the proportion of lake will have the highest influence on waterfowl. Since previous studies have found trees to be positively linked to increased bird species richness in urban parks (Carbó-Ramírez & Zuria, 2010; de Castro Pena et al., 2017) I expect to find the same in this study. In addition to this, I predict that tree species richness within the park and park area will have a significant positive effect on total species richness, since both predictors are likely to increase the habitat heterogeneity within the park, allowing for more species to find something suitable. Lastly, I predict that park area and proportion lake will have the strongest positive influence on total bird abundance because larger habitat areas are likely to support larger numbers of birds.

Methods

Site Selection

The City of Denver served as a case study to examine park features that influence native bird abundance and species richness. Since the city itself was interested in this topic, only parks within the city boundaries were selected. I used eBird data (eBird, 2017) within the county of Denver and selected the 33 urban parks with birding data recorded from 2015 through 2017 (Figure 1). Park size ranged from 0.011km^2 to 1.27km^2.

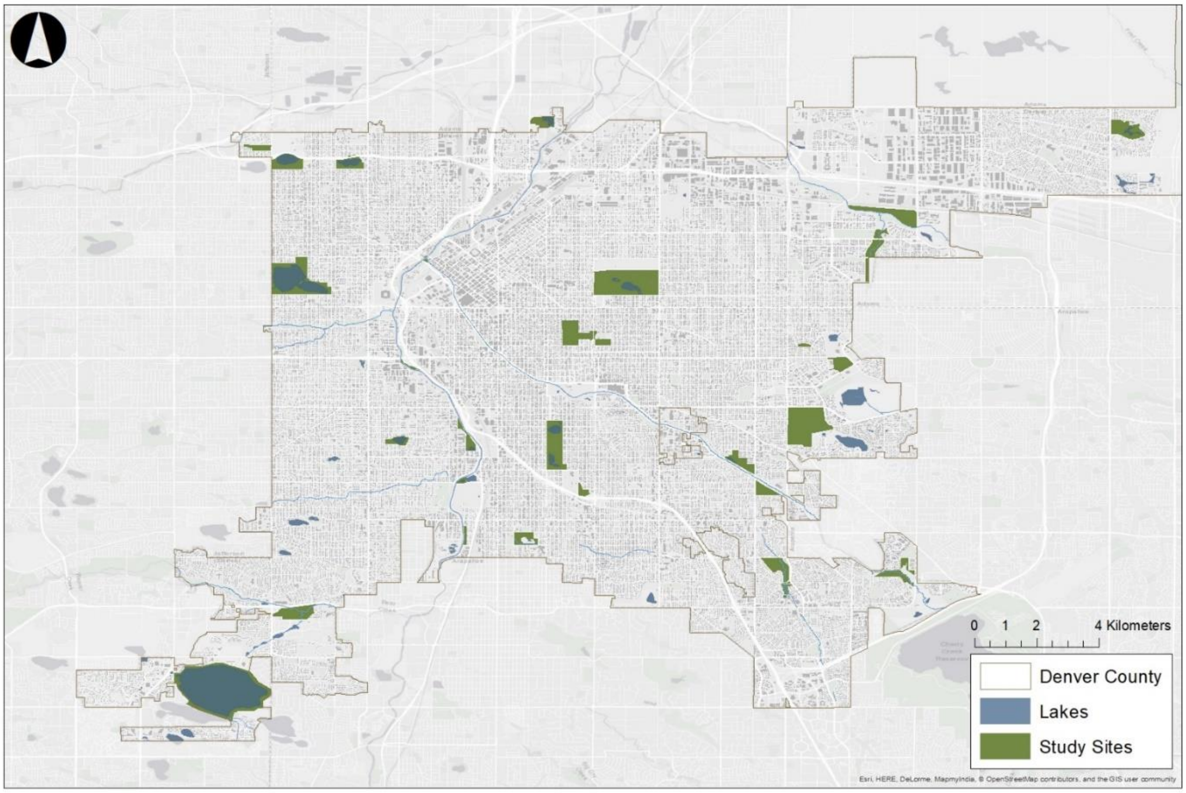

Figure 1: Shows all 33 study sites located across the City and County of Denver, CO.

Urban Park Data

I downloaded GIS shape files for city and park features from the City and County of Denver open source website (Geo-spatial Denver, 2018). These files included the county boundary, urban parks, parkways, tree canopy cover & assessment, pavement edge, building outlines, unpaved trails, court surfaces, lakes, rivers, sidewalks, parking lots, and driveways.

In ArcGIS 10.6 (ESRI, 2017) I generated 500m buffers around each park, then merged all impervious surfaces (buildings, driveways, pavement edges, court surfaces, sidewalks, and driveways) into a single layer.

To determine the proportion of greenspace within each park, I subtracted impervious surfaces and lakes from the park area and divided this number by the total park area. I repeated this procedure within the 500m and 1km buffers. Greenspace within buffers was considered a connectivity variable during analysis.

To calculate the proportion of lake area within each park I calculated the surface area of lakes within each park and divided this number by the total park area. Since river surface area varies seasonally, I instead calculated drainage density, which is the length of stream divided by the park area.

To obtain proportion of complex vegetation cover, I manually traced any non-lawn vegetation patches using satellite imaging, which I also obtained through the City and County of Denver website. I then divided the area of these created shape files by the greenspace area within each park.

To calculate the canopy cover within each park, I trimmed the tree canopy cover layer by the park boundary and calculated the area, then divided this by the total park area. I repeated this within the 500m and 1km buffers to use as habitat connectivity variables.

To estimate canopy complexity I used the number of diameter at breast-height (DBH) categories and the tree species richness within each park. DBH is the best indicator of both tree height and canopy size, and thus differences in DBH can serve as indicators of canopy complexity (Popescu, Wynne & Nelson, 2003). The only tree data available for DBH was categorical, so as a proxy, I counted the number of size categories within each park. Species richness is also an important proxy to canopy complexity, as different tree species provide different structures for nesting and feeding (Popescu, Wynne & Nelson, 2003). I counted the number of tree species within each park and this served as the second variable in canopy complexity.

To ascertain how urban park connectivity affects bird abundance, I calculated the area of parks within a 1km and 500m buffer surrounding each park. I also measured the smallest distance between each park and the nearest surrounding park.

A summary of all environmental predictors & their variability can be see in Table 1.

Table 1: Shows the mean, standard deviation, and range of environmental data.

Variable	Mean	SD	Lowest Value	Highest Value
% Greenspace	84.194	18.01	15.34	100
% Complex Vegetation	29.296	39.745	0	100
% Tree Canopy	12.373	10.72	0.34	42.72
Canopy Complexity (Size)	7.2857	2.0341	3	10
Canopy Complexity (Richness)	51.643	37.848	12	199
Distance to Nearest Park (km)	0.1127	0.177	0	0.638
Area of Parks within 500m (sq. m)	181278	179402	0	573160
% Greenspace within 500m	57.221	9.6381	31.08412	84.19388
% Canopy within 500m	14.877	16.822	1.106321	90.670707
Area of Parks within 1km (sq. m)	418518	340155	33471.2	1136811.6
% Greenspace within 1km	57.125	7.9315	39.01548	73.79897
% Canopy within 1km	13.158	7.4242	1.681809	26.532064
% Lake	9.8929	17.315	0	84
Drainage Density (m/ sq.m)	0.0053	0.0088	0	0.0307
Area (sq. km)	0.2584	0.3068	0.0114762	1.2714575

eBird Data

I used eBird data collected within Denver County parks. eBird data is generated through public volunteers who record bird sightings and their locations through a phone app (eBird, 2017). eBird data contains individual sightings for each species within a park on any given day (eBird, 2017). I used data over the three most recent years (2015-2017) and used counts over the entire year rather than per season, since many birds using urban parks are migratory and choosing a specific season may not capture their presence. The data were reorganized to show the total abundance of each species spotted within each park.

Lastly, to determine whether different park features may influence different birds, I grouped the bird sightings into categories of native or introduced, migratory or resident, and passerine (songbird), waterfowl, or raptor based on Sibley 2001. Within each park I calculated proportion of each bird type, i.e. proportion migratory, etc.

Data Analysis: Generalized Linear Models

To determine which park features have the greatest influence on total bird abundance and richness, I used generalized linear models (GLMs) with Poisson distributions in the base R stats package (R Core Team, 2017). These models assume that the log mean value is a function of the predictors. Since parks are not surveyed equally and the range in birding effort was between 4.58 hours (Congress Park) to 1,153.43 hours (City Park), I used these birding effort hours to normalize the abundance and richness within each park using an offset argument. I separated models into three categories based on independent variables that could influence bird community structure: vegetation, connectivity, water, and area. Within each category, models included all combinations of park features related to that category. Vegetation models included any combination of greenspace, proportion mixed vegetation, proportion canopy cover, and canopy complexity. Connectivity models included distance to nearest park, number of parks within 500m or 1km, percent greenspace within 500m and 1km, percent canopy cover within 500m and 1km, and area of parks within 500m or 1km. Water models included percent lake surface and drainage density. I used all-subset selection in the leaps R package (Thomas Lumley, 2017) to determine BIC scores of all possible model combinations, and chose the one with the lowest score as the best model for that category. To determine the overall best models, I created a model that combined the most significant variables within each best model category (as determined by the previous model selection) and used all-subset selection to determine the model with the lowest BIC score.

Since parks often want to manage for specific bird types, I repeated the above step for native birds, migratory birds, and birds within each category (songbird, waterfowl or raptor) using GLMs with binomial distributions where the log odds of finding a bird of a particular type varied as a function of the predictors included within each model.

Data Analysis: Ordination

Lastly, to visualize differences in community structure along environmental gradients, I performed a non-metric multidimensional scaling ordination (NMDS) in which each bird category was symbolized differently (native vs. non-native, migratory vs. resident, and passerine vs. waterfowl vs. raptor). For this, I removed any species that did not make up at least 1% of the population at one or more sites, as well as any species that only occurred two or fewer sites, this reduced the number of species from 353 to 79, indicating that the majority of species observed were rare. Remaining bird abundances were log transformed due to large differences in abundance between parks. I used the vegan R package (Oksanen et al., 2017) to calculate the Bray-Curtis distances between site species compositions. Using the ecodist package in R (Goslee & Urban, 2007) I looked at the stress of ordinations run using between one and 5 dimensions and determined that a two-dimensional NMDS resulted in a stress below <0.2.

For the actual ordination, I once again used the ecodist R package (Goslee & Urban, 2007), this time only running in two dimensions. I determined the minimum stress within the ordination using the ecodist R package (Goslee & Urban, 2007) before calculating Euclidean distances (using ecodist) and plotting these against the species composition distances to check for a linear relationship. I rotated the axis so that the first (x) axis explains the greatest variation, using the base stats package (R Core Team, 2017). Then I calculated the species ordination scores using the 'wascores' function in the vegan R package (Oksanen et al., 2017). Environmental gradient arrows were calculated based off of site scores using the envfit function within the vegan R package (Oksanen et al., 2017). Lastly, I plotted these species scores and overlaid environmental gradient arrows with a p-value <0.05 using ggplot (Wickham, 2009). I colored each species score based on category to visualize difference in bird types across environmental gradients.

Results

Generalized Linear Models- Park Connectivity

Only bird species richness was significantly influenced by any variables within the connectivity categories from GLMs. The best predictors for total avian richness were distance to the nearest park, area of parks within 500m, percent canopy cover within 1km, and area (Pseudo R^2=0.542, BIC=-14). After controlling all variables included within the model, a 1km increase in distance to nearest park corresponds to a 29.46% decrease in total bird species richness (Table 3; p-value=0.014, 95% CI: 6.94% to 46.67%). A 250m^2 increase in area of parks within 500m corresponds to a 25.06% decrease in bird species richness (Table 3; p-value<$2x10^{-16}$, 95% CI: 30.05% to 19.74%). A 1% increase in canopy cover within 1km corresponds to a 0.7% increase in bird species richness (Table 3, p-value=0.0041, 95% CI: 0.23% to 1.2%).

To assess whether the relationship of park area within 500m and species richness differed as a function of park area, I added an interaction between these variables in the species richness model. Doing so caused the relationship of area of park area within 500m and species richness to flip, so that a 250m^2 increase in 500m buffer park area results in a 90.69% increase in bird species richness (p-value<$2x10^{-16}$, 95% CI: 23.48% to 194.22%). The interaction between area of the park and area of parks within 500m was significantly negative (*p*-value= 2.13 $x10^{-5}$).

Generalized Linear Models- Within Park Predictors

Due to the number of models fit for this analysis, this section will only address models with the lowest BIC scores from each category of explanatory variables (Tables 1-7) as these models are considered the best models for predicting bird community structure within urban Denver parks. Note, that these model outputs are multivariate, and values reported are after controlling for other variables included in the best model for that bird group (see Tables 1-7).

Greenspace had significant influences across the most models, and was positive for passerine species; a 1% increase in greenspace within a park corresponds to a 1.46% increase in the odds of a bird being a passerine species (Table 6; p-value<$2x10^{-16}$, 95% CI: 1.44% to 1.47%). But the opposite was true for both waterfowl and migratory birds, which had negative relationships to percent greenspace. A 1% increase in the percent greenspace within a park corresponds to a 1.97% decrease in the odds of a bird being migratory (Table 5; p-value<2 $x10^{-16}$, 95% CI: 1.95% to 1.98%) and a 1.46% decrease in odds of a bird being a waterfowl species (Table 7; p-value<$2x10^{-16}$, 95% CI: 1.46% to 1.48%). Percent lake, which is inversely related to greenspace (p-value= $7.64x10^{-16}$), only had a significant negative influence on raptor species; a 1% increase in percent lake corresponds to a 0.56% decrease in the odds of a bird being a raptor species (Table 8, p-value<2 x10-16, 95% CI: 0.47% to 0.64%).

Drainage density had the same effects on passerine and waterfowl species as did greenspace, wherein it was positively related to passerine species, but negatively correlated to waterfowl. A $1m/0.5km^2$ increase in drainage density corresponds to a 0.0161% increase in the odds of a bird being passerine (Table 6, p-value<2 x10^{-16}, 95% CI: 0.0158 to 0.0163) and a 0.0172% decrease in odds of a bird being a waterfowl species (Table 7, p-value < 2x10^{-16}, 95% CI: to 0.0174% to 0.0176%).

Similar to greenspace and drainage density, the percent of complex vegetation had varied effects on bird categories. Total abundance of birds was negatively correlated to complex vegetation and a 1% increase in complex vegetation coverage corresponded to a 0.407% decrease in total bird abundance (Table 2; p-value<2x10^{-16}, 95% CI: 0.401% to 0.413%). Conversely, raptor species were positively related; a 1% increase in complex vegetation corresponds to a 1.81% increase in odds of a bird being a raptor species (Table 8, p-value<2 x10^{-16}, 95% CI: 1.76% to 1.86%).

Canopy cover within the park had no significant influence on any bird species, though both canopy complexity variables did. The number of DBH categories (which ranges from 1 to 9) was the only variable that had a significant influence on the proportion of native birds; a 1-unit increase in the number of tree size categories corresponds to a 10.12% increase in the odds of a bird being native (Table 4; p-value < 2x10^{-16}, 95% CI: 9.87% to 10.52%). Tree species richness had a positive effect on total abundance; a 1 species increase in tree species richness corresponds to a 0.218% increase in total bird abundance (Table 2; p-value<2 x10^{-16}, 95% CI: 0.214% to 0.222%).

Table 2-3: Poisson generalized linear models predicting abundance (Table 2) and richness (Table 3) as function of park-level variables. These models were chosen by selecting the best model (lowest BIC) score within each predictor category (greenspace, connectivity, water, area, and combined) from a complete model containing all variables within that category. These show the percent change in 1: total abundance and 2: total richness for each 1-unit increase in the corresponding predictor, after controlling for any other predictors as indicated within each model. For example, a 1% increase in complex vegetation corresponds to a 0.40% decrease in total abundance, after controlling for canopy complexity richness. Note, for Area, these numbers are the percent change in 1: total abundance, and 2: total richness for an e-fold increase in area. **Bold: $p<0.0001$,** *Italic: $p<0.001$,* Regular: $p<0.05$.

2: Total Abundance

Model Category	df	Pseudo R^2	BIC	% Greenspace	% Complex Vegetation	% Tree Canopy	Canopy Complexity Size	Canopy Complexity Richness	Distance to Nearest Park	Area of Parks within 500m	% Greenspace within 500m	%Canopy within 500m	Area of Parks within 1km	% Greenspace within 1km	%Canopy within 1km	% Lake	Drainage Density	log(Area)
Greenspace	3	0.563	-23		**-0.40%**			**0.20%**										
Connectivity	3	0.236	0.71						**0.10%**	**0.0001%**								
Connectivity	2	0.004	6.4											**-0.47%**				
Water	2	0.390	1.8													**0.88%**		
Area	2	0.474	-9.6															
Combined	3	0.563	-23		**-0.40%**			**0.20%**										**31.89%**

3: Total Richness

Model Category	df	Pseudo R^2	BIC	% Greenspace	% Complex Vegetation	% Tree Canopy	Canopy Complexity Size	Canopy Complexity Richness	Distance to Nearest Park	Area of Parks within 500m	% Greenspace within 500m	%Canopy within 500m	Area of Parks within 1km	% Greenspace within 1km	%Canopy within 1km	% Lake	Drainage Density	log(Area)
Greenspace	3	0.542	-3.5	**-0.64%**	**-0.41%**													
Connectivity	4	0.311	-8.6						**0.18%**	**0.0001%**	**0.75%**							
Connectivity	2	0.005	4.7											**-0.40%**				
Water	2	0.390	0.78													**0.88%**		
Area	2	0.263	-7.9															**-33.75%**
Combined	5	0.304	-14						*-29.46%*	**-25.06%**					*0.70%*			**-32.63%**

Table 4-8: Binomial generalized linear models predicting proportion native (Table 4), migratory (Table 5), passerine (Table 6), waterfowl (Table 7), and raptor (Table 8) species as function of park-level variables. These models were chosen by selecting the best model (lowest BIC) score within each predictor category (greenspace, connectivity, water, area, and combined) from a complete model containing all variables within that category. These tables show the percent change in odds of a randomly selected bird at a site being a 3: Native, 4: Migratory, 5: Passerine, 6: Waterfowl and 7: Raptor species for every 1-unit increase in corresponding predictor variable, after controlling for any other variables included within that model. For example, a 1m increase in distance to nearest park corresponds to a 0.25% increase in the odds of a bird being native, after controlling for area of parks within 500m and percent canopy cover within 500m. Note, for Area, these numbers are the percent change in odds for an e-fold increase in area. **Bold: p<0.0001,** *Italic: p<0.001,* Regular: p<0.05.

4: Proportion Native

Model Category	**df**	**Pseudo R²**	**BIC**	**% Greenspace**	**% Complex Vegetation**	**% Tree Canopy**	**Canopy Complexity Size**	**Canopy Complexity Richness**	**Distance to Nearest Park**	**Area of Parks within 500m**	**% Greenspace within 500m**	**%Canopy within 500m**	**Area of Parks within 1km**	**% Greenspace within 1km**	**%Canopy within 1km**	**% Lake**	**Drainage Density**	**log(Area)**
Greenspace	2	0.132	2.5				**10.12%**											
Connectivity	4	0.661	3.8						**0.25%**	**0.003%**		**-0.66%**						
Connectivity	2	0.028	5											**0.000003%**				
Water	2	0.129	6.3													**0.60%**		
Area	2	0.448	1.6															**38.08%**
Combined	2	0.132	2.5				**10.12%**											

5: Proportion Migratory

Model Category	**df**	**Pseudo R²**	**BIC**	**% Greenspace**	**% Complex Vegetation**	**% Tree Canopy**	**Canopy Complexity Size**	**Canopy Complexity Richness**	**Distance to Nearest Park**	**Area of Parks within 500m**	**% Greenspace within 500m**	**%Canopy within 500m**	**Area of Parks within 1km**	**% Greenspace within 1km**	**%Canopy within 1km**	**% Lake**	**Drainage Density**	**log(Area)**
Greenspace	2	0.41	-2	**-1.97%**														
Connectivity	2	0.097	4.1						**0.10%**									
Connectivity	2	0.304	6.8											**-7.37%**				
Water	2	0.268	-1.5													**1.45%**		
Area	2	0.482	3.2															**72.91%**
Combined	2	0.41	-2	**-1.97%**														

6: Proportion Passerine

Model Category	**df**	**Pseudo R^2**	**BIC**	**% Greenspace**	**% Complex Vegetation**	**% Tree Canopy**	**Canopy Complexity Size**	**Canopy Complexity Richness**	**Distance to Nearest Park**	**Area of Parks within 500m**	**% Greenspace within 500m**	**%Canopy within 500m**	**Area of Parks within 1km**	**% Greenspace within 1km**	**%Canopy within 1km**	**% Lake**	**Drainage Density**	**log(Area)**
Greenspace	2	0.383	0.7	**2.17%**														
Connectivity	2	0.423	3.7								**3.08%**							
Connectivity	2	0.232	5.8											**7.32%**				
Water	3	0.424	0.72													**-0.99%**	**0.0002%**	
Area	2	0.413	6.2															**-42.68%**
Combined	3	0.453	-0.078	**1.46%**													**0.0161%**	

7: Proportion Waterfowl

Model Category	**df**	**Pseudo R^2**	**BIC**	**% Greenspace**	**% Complex Vegetation**	**% Tree Canopy**	**Canopy Complexity Size**	**Canopy Complexity Richness**	**Distance to Nearest Park**	**Area of Parks within 500m**	**% Greenspace within 500m**	**%Canopy within 500m**	**Area of Parks within 1km**	**% Greenspace within 1km**	**%Canopy within 1km**	**% Lake**	**Drainage Density**	**log(Area)**
Greenspace	2	0.380	0.58	**-2.20%**														
Connectivity	2	0.441	3.3								**-3.38%**							
Connectivity	2	0.237	5.7											**-7.17%**				
Water	3	0.426	0.4													**1.02%**	**-0.0002%**	
Area	2	0.407	6.3															**78.17%**
Combined	3	0.454	-0.4	**-1.46%**													**-0.0172%**	

8: Proportion Raptor

Model Category	**df**	**Pseudo R^2**	**BIC**	**% Greenspace**	**% Complex Vegetation**	**% Tree Canopy**	**Canopy Complexity Size**	**Canopy Complexity Richness**	**Distance to Nearest Park**	**Area of Parks within 500m**	**% Greenspace within 500m**	**%Canopy within 500m**	**Area of Parks within 1km**	**% Greenspace within 1km**	**%Canopy within 1km**	**% Lake**	**Drainage Density**	**log(Area)**
Greenspace	3	0.774	-2.5	**0.63%**	**1.75%**													
Connectivity	2	0.347	2.6								**1.75%**							
Connectivity	2	0.527	2.3											**12.40%**				
Water	2	0.351	1.1													**-2.48%**		
Area	2	0.00005	7															-0.45%
Combined	3	0.774	-7.1		**1.81%**											**-0.56%**		

0.2267, p-value=0.020) and percent greenspace within 500 meters (r2= 0.3170, p-value=0.001, Table 9). Percent lake and percent greenspace are opposite in their gradients, while percent greenspace within 500m and percent natural vegetation were close to each-other and perpendicular to the percent lake/greenspace (Figures 2-4).

There was a significant difference between centroids of native and introduced birds (r2= 0.0966, p-value= 0.003, Figure 2). Migratory and resident birds also differed significantly despite a large amount of overlap (r^2=0.2278, p-value=0.001, Figure 3). Lastly, there was also significant difference between passerine, waterfowl and raptor species were clearly distinguishable (r^2= 0.2220, p-value= 0.001, Figure 4). Furthermore, raptors were completely within the range of passerine species (Figure 4).

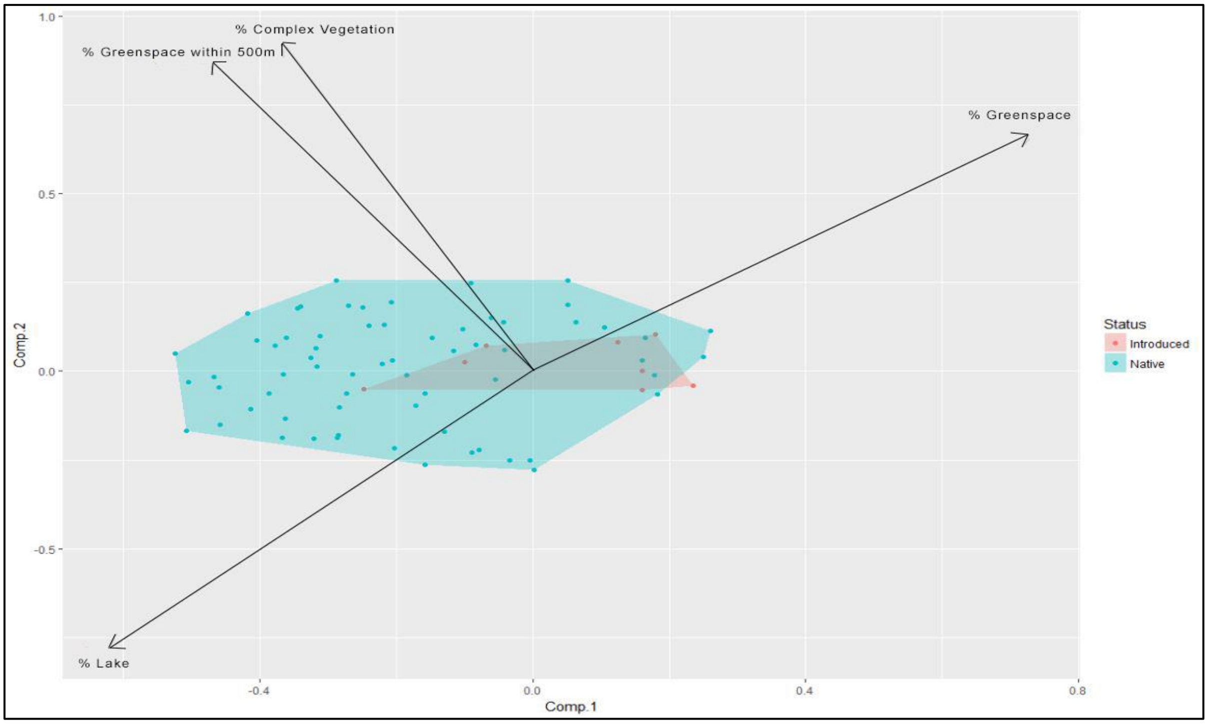

Figure 2: NMDS of 79 included bird species separated by native status. Arrows indicate significant environmental gradients across parks and are based on site scores. Points are weighted average species scores.

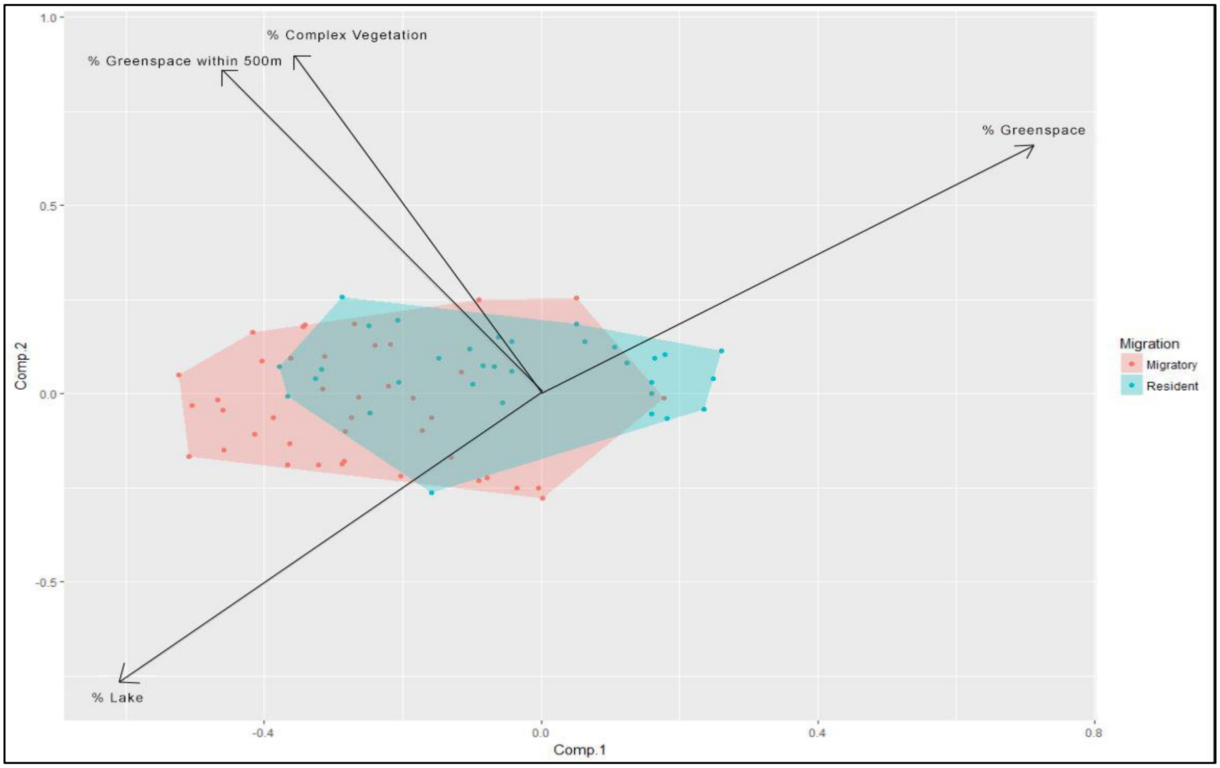

Figure 3: NMDS of 79 included bird species separated by bird migratory status. Arrows indicate significant environmental gradients across parks and are based on site scores. Points are weighted average species score.

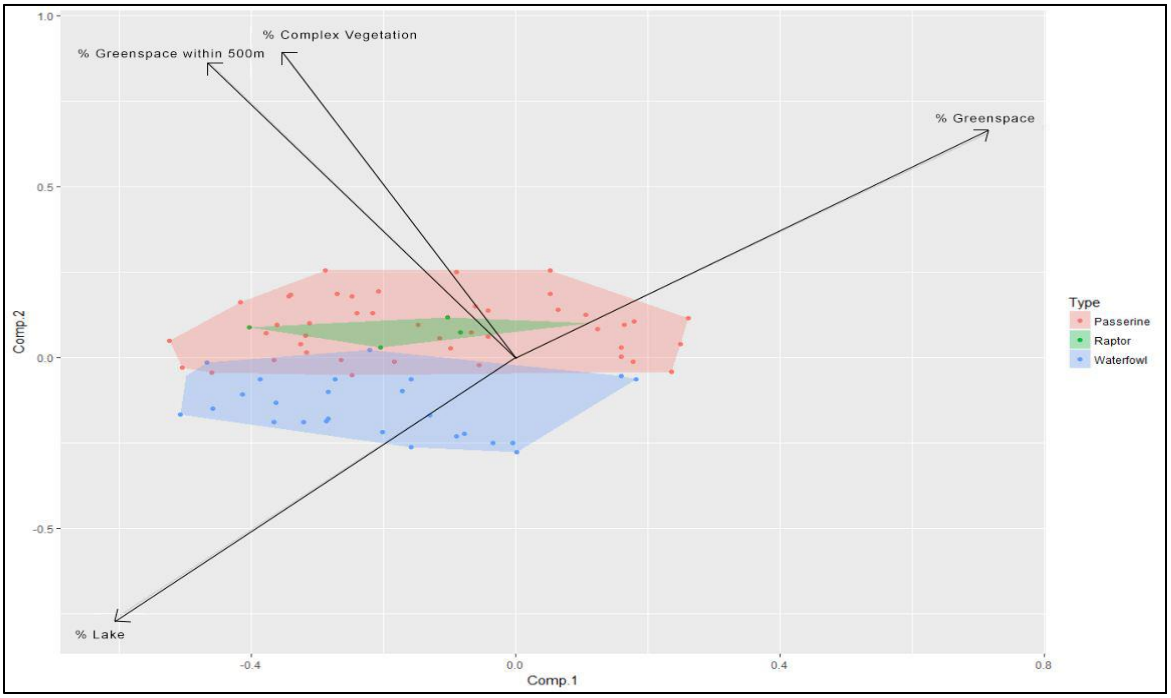

Figure 4: NMDS of 79 included bird species separated by bird type. Arrows indicate significant environmental gradients across parks and are based on site scores.

Table 9: Environmental gradients based on site scores, relative to NMDS axes 1 & 2. significant values are bolded.

Environmental Parameter	NMDS Axis 1	NMDS Axis 2	r^2	*p*-value
% Greenspace	***0.73548***	***0.67754***	***0.2460***	***0.013***
% Lake	***-0.62147***	***-0.78344***	***0.2500***	***0.014***
Drainage Density	0.63367	-0.77361	0.0868	0.241
% Natural Vegetation	***-0.36726***	***0.93012***	***0.2267***	***0.020***
% Canopy	0.78906	0.61432	0.0242	0.681
% 500m Greenspace	***-0.47407***	***0.88049***	***0.3170***	***0.001***
% Canopy 500m	0.36959	0.92919	0.0828	0.272
Area Parks 500m	-0.61961	0.79046	0.1326	0.332
% 1km Greenspace	-0.61252	0.79046	0.1326	0.125
% 1km Canopy	0.10192	0.99479	0.1082	0.181
Area Parks 1 km	-0.52107	0.85351	0.0190	0.739
Distance to Nearest Park	-0.91644	-0.40018	0.1333	0.120

Discussion

This study shows the varied responses of different bird groups to environmental gradients within Denver parks; there were no environmental predictors that had positive effects across all birds, thus there is no single best way to design a park to benefit all birds within Denver. This indicates the need for diverse habitat types that can be utilized by different birds. This is the first study to examine how park design and location within cities influence different bird groups, although it is not the first study to look at environmental gradients on urban birds as a whole (Savard et al., 2000; Fernandez-Juricic & Jokimäki, 2001; White, Antos, Fitzsimons & Palmer, 2005; Sandström et al., 2006; Carbó-Ramírez & Zuria , 2011; McCaffrey & Mannan, 2012; de

Castro et al., 2017). By grouping birds into different categories, this study accentuates the need for diversity in park design to support different types of birds.

My study also aimed to determine which gradients have an influence over total bird abundance and richness. When looking at overall bird abundance, my findings suggest that greenspace and complex vegetation are the most important variables. Contrary to my hypothesis, complex vegetation negatively influenced bird abundance, but this could be explained by other park attributes I did not measure, such as accessibility. Parks with higher proportions of complex vegetation were mostly large parks designated as “natural-areas”. This designation means that the park is less focused on recreation and tends to have fewer pathways. This could discourage or prevent birders from accessing and observing species further off paths (Winnasis, Hakim& Imron, 2018). This issue could be corrected in future studies by either controlling for path density within parks, or having regulated observations at points scattered through a park, independent of path proximity. Studies on urban birds and vegetation have shown that having complex vegetation is positively correlated with bird abundance (Gavareski, 1976; White, Antos, Fitzsimons & Palmer, 2005) which is what I had expected to observe in this study. In line with the findings of multiple papers, I did observe a positive correlation between bird abundance and proportion of greenspace (Gavareski, 1976; Blair, 1996).

For total bird richness, my findings also show significant importance of multiple connectivity variables. Increased distance to the nearest park had a negative impact on total richness, indicating that the closer placement of parks allows for more increased exchange of bird species, which is well supported in other studies of bird richness and habitat connectivity (Minor & Urban, 2008). In support of my hypothesis, percent canopy cover within a 1km-buffer had a positive correlation with bird species richness. This observation has been repeated in multiple studies looking at street-tree-canopy as a connective pathway for urban birds (Fernandez-Juricic & Jokimaki, 2001; Carbó-Ramírez & Zuria, 2010; de Castro Pena et al., 2017). Contrary to what others have found, my results show a negative relationship between species richness and area of the park. Area and richness are often associated because larger areas usually encompass a greater variety of habitat types (Coleman, Mares, Willig & Hsieh, 1982), and thus support more species types, but this may not have been the case in urban settings as many parks are manicured and have similar habitat across the entire area, or this may have been a result of already controlling for multiple habitat variables within my models. This could also be a result of lower accessibility in larger parks, limiting the ability of observers to see birds in areas far from pathways (Ewing & Cervero, 2001).

While it can be important to know how to manage the entire bird assemblage within parks, it may be more effective to manage for specific bird groups. Different categories of birds can have opposite habitat requirements, and thus managing for the benefit of one may cost another (MacArthur, MacArthur & Preer, 1962). For this study I categorized birds into major management groups that will allow for urban park planners to prioritize specific bird categories (which are not all mutually exclusive) and analyzed them separately to determine different habitat effects.

One such bird category is native birds, which are a common focus for restoration projects, because they are typically more specialized and sensitive to development (Batáry et al., 2017). My results show that the number of DBH tree-size categories is the most important predictor of proportion of native birds. This is to say, that parks with greater variation in tree size are more likely to support a higher proportion of native birds. While this has not been specifically investigated in other studies, it could be because the greater canopy complexity allows for more specialized native birds to nest and feed within these settings as they have a lager variety of tree types to choose from. My ordination shows that introduced species occupy a small subset within the range of native birds across the environmental gradients. This suggests that the specialization of native birds may allow them to live in areas less favored by the introduced species. These results suggest that higher values of percent greenspace, percent lake, percent complex vegetation and percent greenspace within 500m, will favor a higher proportion of native birds.

In management, a second important category of birds is migratory status because migratory birds are federally protected, and the Urban Bird Treaty also aims to increase habitat for these species. My results show that migratory birds are negatively impacted by greenspace, which is the opposite of the total bird abundance findings above. This has not been specifically investigated in other studies, but is most likely because 91.8% of all migratory birds recorded in this study were waterfowl, and proportion of greenspace is inversely related to percent lake cover. This idea was supported through ordination analysis which showed an inverse relationship between percent greenspace and percent lake, and that migratory birds tended to prefer habitat with higher percent lake (Figure 3).

Lastly, cities may want to manage for different bird types within their parks for a variety of reasons. I found that passerine species were positively correlated to percent greenspace and drainage density, which makes sense as passerine species depend on trees for nesting and ground vegetation for foraging (Sibley, 2001). Previous studies have found that passerine species are influenced by canopy cover (Carbó-Ramírez & Zuria, 2010), and while my results did not directly indicate that, greenspace and tree cover are closely linked since trees only grow in greenspace. Increased drainage density allows for water without compromising the amount of available nesting habitat. The ordination analysis supported these claims for greenspace, but did not show drainage density to be a significant driver of community composition. Conversely, waterfowl were negatively correlated to both greenspace and drainage density. This is likely because increased proportion of greenspace comes at the cost of proportion water. In my data, no parks with lakes had high drainage density, thus drainage density within parks may only be negatively correlated to proportion of waterfowl as a result of parks having lakes lacking above-ground drainage streams (and thus drainage density being positively correlated with proportion greenspace itself).

While passerine and waterfowl species are virtually opposite in their habitat requirements, raptors were positively correlated with complex vegetation and negatively correlated with percent lake. Raptors are dependent on small animals such as rodents or small birds for prey, and previous studies have found rodent abundance to be positively related to vegetation complexity (Panzacchi et al., 2010), so increased amount of complex vegetation could be correlated to increased prey abundance for these predatory birds. It is also worth noting that the environmental range of raptors is completely encapsulated by the range of passerine species, which is likely because songbirds are prey for raptors (Sibley, 2001).

Although I found several strong relationships between park level variables and the bird community, there were a few areas that could be improved, such as the use of publicly sourced birding data. While I corrected for effort effects on richness and total abundance, not all public birders are equally skilled and they may have differed in their observation accuracy (McCaffrey, 2005). Additionally, parks are not all equally accessible. The two parks with the highest species richness (City Park & Marston Lake) also had the highest number of observation hours, and are both very popular and highly accessible parks. As a result, parks with higher observation time and higher numbers of bird observation are likely to have higher species richness, which is not something that observation hours can correct for. This could be corrected for in future studies by rarifying species richness based on the total number of birds observed. Additionally, it would be best to have qualified birders to collect the data within a set time frame who have permission to venture off trail in large parks with fewer paths.

Despite these limitations of a purely observational study that utilized data collected for other purposes, this study is the first to provide a framework for future studies. In future studies, trained birders should collect the data from random points that are not tied to designated paths. This small change in data collection would ameliorate most of the error that may have been present in this study. Due to the nature of this subject, it would be difficult to do studies like this experimentally, but cities within different regions should conduct similar studies, as different species communities may respond differently to habitat gradients.

This study has many implications for the management of urban parks and specific bird groups, however the most important implication is the role of habitat diversity within urban park networks to support different bird groups. Having some parks with lakes and complex vegetation or others without lakes or complex vegetation allows for a wider range of bird species to establish within the city. No one habitat type is likely to support the entire community of birds, so planning urban parks to be unique in their habitats is essential. This study builds off multiple former studies showing the importance of connectivity of parks on bird species richness, so beyond building new parks, cities should also plan for tree plantings and greenways that allow species to exchange across parks within the city.

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CHAPTER 4. ENVIRONMENTAL STAKEHOLDER ANALYSIS: KLAMATH DAM REMOVAL- THE CHANCE TO REVIVE SALMON RUNS, BUT AT WHAT COST?

Shortly after the United States won independence, the new country was faced with an abundance of public land in the largely unsettled and underexplored (by Europeans) Western United States (Bradsher, 2012). To encourage the settlement of these "open" lands, The Homestead Act was passed in 1862, promising free land to anyone willing to settle it (Bradsher, 2012). Farmers and ranchers, alongside their families, headed westward to promised sites they had not seen, other than on a map (Bradsher, 2012). And, while many people arrived on fertile land, many others, unknowingly, had been assigned to regions too dry or unreliable to support crops or livestock (Bradsher, 2012). As more and more people struggled to make ends meet on their barren land, public efforts called for the construction of dams to store water in these regions (Billington, Jackson & Melosi, 2005). Some dams were smaller in scale, built by landowners themselves, while others were huge infrastructural undertakings of the federal government and hydroelectric companies (Billington et al., 2005). These dams were hugely successful in increasing accessibility and reliability of water, but this didn't come without consequences.

Many of these dams halted the free flow of water draining into the Pacific Ocean, consequently blocking of thousands of miles of important spawning grounds for anadromous fish, most famously salmon. The construction of these dams is thought to be almost entirely responsible for the dramatic declines of salmonid species that have occurred within the last 100 years (Nehlsen, Williams & Lichatowich, 1991). Salmon provide essential functions to terrestrial ecosystems surrounding their migration streams, as well as important economic and food sources to the surrounding fisherman and Native American tribes (Gende, Edwards, Willson & Wipfli, 2002). As dams age and deteriorate, and as public awareness of the environmental consequences of dams grows, there are numerous public efforts to remove dams rather than repair them, but this can have dire consequences for the farmers and ranchers who have benefitted from their presence.

Here, I focus on a set of five dams in the Klamath River which straddles the Oregon-California border (Gilman, 2016; Leslie, 2017). Four of these dams have been slated for removal after inspection revealed they cost more to run and repair than can be justified by the operating company, PacifiCorp (Leslie, 2017). The removal of these dams has been long fought for by the numerous Native American tribes surrounding this river basin, whose cultures and economic wellbeing greatly depend on the once-abundant salmon runs in the Klamath River (Magagnini, 2017). Tribespeople, local fishers, and tourism-centered employees alike have all been overlooked with the maintenance of these dams, and all celebrate their slotted removal (Gilman, 2016; Leslie, 2017). However, this is a grim decision for the thousands of farmers and ranchers further upstream, who will no longer be guaranteed stable water availability, and will return to a much less secure and less predictable yield (Gilman, 2016; Leslie, 2017). To minimize harm to any one set of stakeholders, people from both sides of the dam-removal debate will need to come together and reach an agreement that allows for river reclamation while also supplying a minimum guaranteed amount of water through irrigation water shares allocated to farmers and ranchers, as had previously been proposed in the Klamath Basin Plan of 2015 (Leslie, 2017). To understand the various sides and stakeholders of this issue I will walk through each perspective, starting with the ecosystem benefits of salmon themselves, then discussing the cultural and economic benefits of salmon runs, then finally discussing the farmers and ranchers upstream of these dams.

Most populations of fish called salmon are anadromous, meaning they are born in fresh water streams but migrate to the ocean, where they live most of their adult life (Gende et al., 2002). The adult salmon then return to freshwater streams to mate, an event widely known as a salmon run (Gende et al., 2002). Salmon runs, which occur annually, comprise tens of millions of individual fish of multiple salmonid species; different stocks of each species will return to the same river they were born in, ranging from Alaska to Northern California (Gende, 2002). When adult salmon enter freshwater streams to begin migration to spawning sites, they are full of lipids to give them the energy to swim upstream, sometimes thousands of miles inland (Hendry & Berg, 1999; Gende et al., 2002).

As salmon use these stored nutrients and expel their waste into the stream, they change the surrounding water chemistry (Donaldson, 1967). This results in augmented algal biomass and riparian plant growth (Gende et al., 2002). Additionally, several predatory mammals and birds gather at spawning grounds to feast on the abundant influx of protein, fat, calcium, and other nutrients brought inland by the salmon (Reimchen et al., 2000; Gende et al., 2002). These terrestrial animals are dependent on this important food source for energy and nutrients to help their over-winter survival, growth, and general health (Reimchen et al., 2000; Gende et al., 2002). As these mammals and birds migrate away from the stream after spawning is complete, they continue the flow of marine nutrients inland (Hilderbrand, Hanley, Robbins & Schwartz, 1999; Gende et al., 2002). Many of these animals are important seed distributers, and without this ecological service the composition of the surrounding plant habitats may be dramatically altered (Hilderbrand et al., 1999; Gende et al., 2002).

To regain the ecological benefits brought to terrestrial habitats by salmon runs, we must make it possible for salmon to migrate upstream to their former habitats, doing so will also increase their abundance for harvest and economic gain. One of the primary methods of restoring salmon populations is to remove dams that form barriers to former salmon spawning habitats. These dams were initially built to hold water in dry areas and generate hydroelectric power. In multiple areas, such as California, 80% of former salmon habitat has been made inaccessible by these barriers (Quinones et al., 2015). The Klamath River, is among this blocked habitat, containing five total dams, four of which are approved for removal starting in 2020 (Gilman, 2016; Leslie, 2017).

Klamath dam removal has been long advocated for by the Klamath Tribes in Oregon, who feel they have been short-changed because the dams, which only benefit upstream agriculture and ranching, deplete one of their most important cultural and economic resources (Magagnini, 2017). Many other tribes along this river, including the Karuk, Yurok, and Hoopa Valley tribes, have also been strong advocates of dam removal, working together to gain public and private support (Magagnini, 2017; Klamath River Renewal Corporation, n.d.). These tribes are tied to the Klamath River culturally, spiritually and economically and each have their own unique ceremonies to celebrate the return of salmon to the rivers during the summer salmon runs (Magagnini, 2017). These stakeholders are motivated by the value of salmon as cultural heritage, but beyond this, they depend on salmon economically. Their families have been sustained by salmon for centuries, but for the first year ever, no fall chinook salmon were harvested during the 2017 run, due to critically low populations (Magagnini, 2017). The dwindling numbers of available salmon have caused economic and health declines in tribe members who no longer eat salmon regularly and have instead turned to buying processed foods (Magagnini, 2017).

Almost all commercial fisheries in the Klamath are operated by local tribes. But salmon stocks in this area, once abundant and profitable, are facing a possible complete closure following record-low runs in 2017 (Smith, 2017). Each year, The Pacific Fishery Management Council releases a salmon management strategy that allocates allowed harvest for various fisheries, tribes, and recreational fishermen (Smith, 2017). This plan is based on known harvest rates from previous year runs, and in the current plans, the complete closure of Klamath fisheries is being proposed (Smith, 2017). This closure would eliminate hundreds of jobs and leave many people without access to this food source, as well as limiting fishing tourism (Smith, 2017). Tribe members and fishermen alike feel this situation could have been prevented if they been listened to in former proposals for river management and water disputes (Leslie, 2017).

In 2002, the tribes and fisheries lost a water access dispute with farmers who used Klamath water for irrigation. However, this point of dispute was readdressed when four PacifiCorp-operated dams were recently assessed for relicensing in 2016 (Leslie, 2017). These four dams were proposed for removal in 2015 as part of the Klamath Basin Plan, before relicensing was mandated, but this plan was dismissed by congressional Republicans fearing the long-term implications of dam removal for other economically important dams (Leslie, 2017). However, during the relicensing process, it was determined that, since the dams produce very little electricity, dam removal was cheaper than repair and construction of fish ladders resulting in a definitive decision to remove the dams without the compromises in water shares for farmers, that was written into the Klamath Basin Plan (Leslie, 2017; Klamath River Renewal Corporation, n.d.). This decision is a victory for the tribes, fishermen, and environmentalists alike, but it has obvious drawbacks for farmers and ranchers upstream of the dams, who have been using reservoir water to irrigate crops and water livestock (Leslie, 2017).

In the original Klamath Basin Plan, a clause was included that gave a minimum amount of basin water for agriculture during a dry year, which was intended to minimize negative impacts to farmers and their communities (Leslie, 2017). However, the senior water rights of the Klamath tribes were recently affirmed, giving farmers less sway in bargaining for water shares as tribes know that more water from the basin means better health for the river system (Leslie, 2017; Magagnini, 2017). Before the dams were slotted for removal due to age and inefficiency, farmers and ranchers had been open to negotiation for guaranteed minimum water shares in exchange for agreeing to the dam removal, which was the basis of the Klamath Basin Plan (Leslie, 2017). But, guaranteed dam removal, independent of farmers consent, has eliminated this as a possible compromise between the opposing groups (Leslie, 2017). As it stands now, the removal of the Klamath dams will have only detrimental effects for farmers and ranchers who currently depend on these reservoirs for irrigation and watering, and who will be left with no water security following their removal because the one remaining dam will not provide sufficient water (Leslie, 2017).

Issues like this are going to become more prevalent as dams age and lose efficiency and as water becomes less available with increasing demand and changing climate. It is important to find a middle ground that can benefit all parties, or at minimum reduce harm to any of them. Many groups including surrounding tribes, environmental non-profits, tourism companies, and fisheries are likely to benefit from the removal of these four Klamath river dams. The Klamath River ecosystem is also likely to benefit, since in previous dam removals, salmonid have been shown to utilize these newly accessible areas (Stanley & Doyle, 2003).

However, in the shadow of this victory lies farmers and ranchers who lose the security and reliability of water needed to operate their businesses. It is easy to forget that almost every American depends on farmers for food, and if they are overlooked repeatedly in similar situations surrounding water, American agriculture will be significantly impacted. Food prices would likely increase, and poverty rates of farming or ranching families would skyrocket. Additionally, dam removal alone will not revive salmon populations, and at best will just slow their declines (Kareiva, Marvier & McClure, 2000). Additional measures would be needed to reverse the damage done to salmon populations by the construction of dams, which may or may not be feasible for the tribes, non-profits, and companies that would benefit from increased salmon runs.

To best serve all stakeholders, I think that the Klamath Basin Plan that was proposed in 2015 should be reimplemented to guarantee farmers and ranchers at least a baseline amount of water for use during dry years, even if it is not enough to as large of yields as may have been possible with dams present. My reason for this decision is largely based in sustainability as dams are loosing their functionality with sediment build up, and will not likely be able to provide sufficient water in the long term anyways. Additionally, technology with desalination or indoor farming may eventually reduce the high water demands by farms, while advances in technology are unlikely to resolve declines in salmon populations. The exact amount of water should be determined by a committee of agricultural and ecological specialists who understand the needs of both the environment and crops. If farmers and ranchers want more than their baseline amount of allocated water, they should be required to donate a portion of their profits towards other salmon-enhancing efforts. Tribes and non-profits should be freely allowed to monitor the use of water by farmers and ranchers to minimize violation of baseline limits. Lastly, the federal government should allocate money to assist farmers and ranchers in affording technologies that minimize their reliance on water for crops, such as hydroponics, or switching to more drought-tolerant crop species. In issues such as these, it is critical to consider all impacted parties to try and find the solution that minimizes harm to any one group of stakeholders, but it is also crucial to consider the sustainability of any choices made. While farming technologies are likely to improve with time, potentially reducing their water demands, salmon populations are unlikely to recover without action.

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