THE EFFECT OF RIPARIAN ALTERATION ON BROOK TROUT POPULATIONS, STREAM MORPHOLOGY, AND INVERTEBRATE COMMUNITIES

By

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A Thesis

Submitted in partial fulfillment of the requirements of the degree

MASTER OF SCIENCE

IN

NATURAL RESOURCES

College of Natural Resources

UNIVERSITY OF WISCONSIN

Stevens Point, Wisconsin

May 2018

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ABSTRACT

Riparian vegetation influences stream morphology, habitat and biological communities providing a source of connectivity between aquatic and terrestrial ecosystems. The objective of this study was to determine how riparian vegetation, including the removal of Tag Alder in the riparian corridor and addition of brush bundles, effects stream morphology, Brook Trout *Salvelinus fontinalis* habitat and feeding, and invertebrate communities. We sampled four sites, May through September 2017, with different types of riparian vegetation at Green Meadow Creek Lincoln County, WI. Three of these sites were references that represent the different types of vegetation found in the area, second growth forest, shrub or Tag Alder *Alnus incana*, and sedge meadow. The fourth site was experimental and consisted of a Tag Alder riparian area. Here, Tag Alder was removed and used as brush bundles within the stream to alter morphology. Habitat measurements included width, depth, substrate composition, canopy cover, and riparian land use. Trout were sampled using an electrofishing depletion method and stomach contents were obtained using a gastric lavage. Aquatic invertebrates were sampled using a Surber sampler and D-net while terrestrial invertebrates were sampled using pan traps. At the treatment site, brush bundles are already narrowing the stream while slightly increasing depth. The percentage of gravel has increased at the treatment site while the percentage of sand has decreased. Brook Trout selection of prey varied with the different types of riparian vegetation. Brook Trout in woody riparian areas feed more on terrestrial Trichopterans and Ephemeropterans while trout at grassy riparian site feed more on Ephemeroptera nymphs and terrestrial Dipterans. Selection for terrestrial invertebrates is highest during summer months and lowest during the late spring and early fall. Terrestrial invertebrate abundances are highest during the mid summer months while aquatic

invertebrate abundances are highest during the late spring and early summer. The removal of Tag Alder and addition of brush bundles have begun to alter channel morphology and those changes, coupled with the alteration of riparian vegetation, may alter diets of Brook Trout.

ACKNOWLEDGMENTS

Funding for this project was provided by the University of Wisconsin Stevens Point, including the Office of Student Creativity and Research. I would like to thank my committee members Dr. William Fisher, Dr. Joshua Raabe, and Dr. Kyle Herrman for their assistance with the study design and comments that improved the manuscript. I would also like to thank Tom Daigle and family members for providing access to the stream and assisting with the Tag Alder removal. Finally, I would like to thank the UWSP undergraduate and graduate student volunteers that assisted with field work throughout the project.

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Introduction

Riparian zones are an important component of healthy stream ecosystems, providing a critical link between aquatic and terrestrial ecosystems and influencing the biological, physical, and chemical characteristics of streams (Gregory et al. 1991). Vegetation within riparian zones is influenced by geographic location and the successional stage of the surrounding land (Gregory et al. 1991). Another contributing factor to the type of riparian vegetation is past and present land use or disturbances within the stream watershed. For example, most stream sites within two forested watersheds in the southeastern United States in the 1990's also had forested watersheds in the 1950's; however, some sites with agricultural watersheds in the 1990's were more forested in the 1950's indicating that agriculture was a form of disturbance (Harding et al 1998). In Wisconsin, Wang et al. (1997) found that streams with forested watersheds were positively correlated to stream habitat quality and biotic integrity indices, whereas agricultural watersheds were negatively correlated. It can take decades or longer for watersheds and riparian areas to recover from long-term disturbances, such as agriculture or other human activities (e.g., urbanization), while recovery from short-term disturbances, such as floods, is much shorter (Niemi 1990). Lyons et al. (2000a) suggested that a forested watershed should have forested or woody riparian vegetation; however, grassy riparian areas can also be managed for in forested watersheds if landowners carefully weigh all options and agree on management goals.

Following disturbance of a forested ecosystem, such as logging within the watershed or riparian zone, the area goes through natural successional stages (Swanson 2011). Grasses are the first vegetation to regenerate followed by shrubs and other woody vegetation, such as Speckled or Tag Alder *Alnus incana*, which dominate the area by shading out the grasses. Finally, hardwood and softwood trees shade out the smaller woody vegetation. Each stage influences the stream and biological communities differently. Grassy riparian areas tend to have narrower stream channels, low habitat diversity, and trap sediments, whereas forested riparian areas have slightly wider channels, more diverse habitat, and input more coarse woody debris into the stream (Jackson et al. 2015). However, both riparian types typically increase bank stability and soil strength due to their root systems (Simon and Collison 2002).

Stream and riparian areas are managed in a variety of ways, often to enhance the channel morphology and instream habitat. Goals set by management agencies and landowners for stream and riparian habitat management are based on location and the techniques used to reach those goals. Common in-stream habitat management techniques include the addition of boulders, coarse woody habitat, current deflectors, half logs, dams (or removal of dams), and the reshaping of the bank to increase stability (Wesche 1985, DeWeber and Wagner 2015). Another in-stream management technique is brush bundling (Hunt 1993), which involves removing shrub riparian vegetation and placing some of it in the stream to trap sediment and divert flow leading to narrower and deeper stream channels. Riparian vegetation management techniques include using herbicides to prevent regeneration of unwanted species, prescribed burns, planting of desired vegetation, and timber harvest (Lyons et al. 2000b; Phillips et al. 2000; Gregory 1997).

The removal of woody vegetation is another riparian management technique that is less common but can modify stream morphology and potentially improve riparian habitat (Hunt 1979).

Removal of riparian vegetation can potentially increase stream water temperatures due to lack of canopy cover, which could be detrimental to some fish populations such as trout. However, Hunt (1979) found that removal of riparian vegetation did not result in water temperatures rising above levels detrimental to trout. Hetrick et al. (1998) removed riparian vegetation to study the effect on juvenile Coho Salmon *Oncorhynchus kisutch* feeding and found decreased feeding on terrestrial invertebrates after removal. Both Hunt (1979) and Hetrick et al. (1998) strongly suggest considering the effect riparian vegetation removal on stream temperatures. Intensive rotational grazing (IRG) of livestock is a management action that can prevent regeneration of woody vegetation and allow grasses to revegetate the riparian area (Lyons et al. 2000b).

Aquatic and terrestrial communities are influenced by the different types of riparian habitat. Grass, or herbaceous, riparian vegetation results in more undercut banks and increased sediment trapping within the stream compared to woody vegetation riparian areas (Lyons et al. 2000a). Woody riparian areas input coarse woody debris that can slow flows, reduce downstream flooding, and provide cover (Bennett et al. 2008). These different riparian types can influence trout prey selection on both aquatic and terrestrial invertebrates (Wilson et al. 2014). Further, aquatic and terrestrial invertebrate availability can depend on the abundance of each and may influence feeding behaviors of trout (Nakano and Murakami 2001).

Literature Review

Brook Trout Habitat

Brook Trout *Salvelinus fontinalis* are a cold-water salmonid species that are habitat specialists. Maximum summer water temperature has a profound effect on where Brook Trout occur within a stream (MacCrimmon and Campbell 1969). Brook Trout exhibit increased movement upstream during spring and summer due to rising water temperatures downstream (Meisner 1990). Further, the amount of fish movement may depend on stream size and groundwater occurrence. Increasing water temperature cause trout to seek areas of cooler temperatures in streams (Cross et al. 2013). Discharge stability and fine suspended sediments also influence Brook Trout location during summer months, but not as much water temperature (Barton et al. 1985). Eifert and Wesche (1982) found that the amount of surface area where depths were greater than 15 cm and substrate sizes were greater than or equal to 7.6 cm in diameter, or substrates with aquatic vegetation, was indicative of streams dominated by Brook Trout.

In the winter, Brook Trout have lower metabolic rates due to decreased water temperatures and they feed less, causing them to select areas where water velocity is low with cover (woody debris, undercut banks, and vegetation) to reduce energy consumption and predation risk (Cunjack 1996). Brook Trout will also aggregate in areas of groundwater discharge within the stream because these areas create pockets of warmer temperatures that lead to increased winter survival for all age classes (Cunjack 1996). During winter, Brook Trout over age-1 tend to live in deeper areas of the stream compared to the younger trout (Cunjack and Power 1987). Brook Trout feed to survive during the winter compared to summer when food consumption is primarily used for growth. Thus, habitat availability during winter months is critical to Brook Trout survival.

Brook Trout create spawning sites, or redds, by selecting specific habitats and substrate types for construction. During spawning season in fall, Brook Trout prefer areas to build their redds where groundwater is upwelling, creating cooler water temperatures (Snucins et al. 1992). In winter, these groundwater upwellings contain warmer water than ambient stream temperatures (Cunjack 1996), increasing the chance of egg survival. The redds are built upon smaller-sized gravel at depths between 6-8 inches (Snucins et al. 1992). Sandy substrate slows down and reduces overall emergence of Brook Trout from eggs (Hausle and Coble 1976) and also has the potential to suffocate eggs by burying them due to streamflows (Alexander and Hansen 1986).

Brook Trout Diet and Growth

Brook Trout diet changes as they age and grow larger. Brook Trout fry feed primarily on smaller invertebrates, like blackfly larvae and zooplankton (Williams 1981). When Brook Trout reach the parr and juvenile life stages they feed more on larger invertebrates. As adults they feed on larvae, pupae and adult invertebrates (Williams 1981), terrestrial invertebrates that fall into the stream (Li et al. 2016), smaller-sized fish including sculpins (East and Magnan 1991), and crustaceans, such as crayfish (Momot 1967) and scuds (Elwood and Waters 1969).

Brook Trout preference for aquatic and terrestrial invertebrates changes seasonally. During late spring and summer, salmonids feed primarily on terrestrial invertebrates that fall into the stream, whereas during fall and winter they feed primarily on benthic aquatic invertebrates (Baxter et al. 2005). One explanation for the shift in diet preference is the reduction in terrestrial invertebrate abundance due to decreasing air temperatures and increasing mortality. Terrestrial invertebrates can make up to 50% of annual salmonid diets and between 50% and 86% during summer (Baxter et al. 2005). Li et al. (2016) found that terrestrial invertebrate consumption by Pacific salmonids (*Oncorhyncus* spp.) from a southwestern Oregon stream was highest in summer, shifting from a mixed diet of benthic, terrestrial, and adult aquatic invertebrates in spring. They also found that terrestrial invertebrate consumption in the fall made up around half of the fish's diet (Li et al. 2016). The primary reason terrestrial invertebrates make up a large proportion of salmonid diets is because of the increased size, visibility and availability and higher energy potential of terrestrial invertebrates compared to aquatic invertebrates (Nakano et al. 1999; Baxter et al. 2005). Wilson et al. (2014) found that the biomass of benthic aquatic invertebrates influenced the amount of terrestrial invertebrates in Brook Trout diets; specifically, that an increase in benthic invertebrate biomass led to a decrease in feeding on terrestrial invertebrates. There are other factors that can influence fish feeding habits. Fish size is one factor that influences prey size and feeding behavior. A positive correlation between fish size and terrestrial prey size was observed for salmonids in 25 streams in Finland and was explained by gape limitations and territoriality by the larger fish (Syrjanen et al. 2011). Larger salmonids fend off the smaller ones in order to gain access to the best feeding grounds. Sweka and Hartman (2008) found that in the absence of terrestrial invertebrate inputs, Brook Trout growth was slower and aquatic invertebrates alone could not maintain the amount of growth provided by terrestrial invertebrates. Several studies on trout diets have stressed the importance of terrestrial invertebrate inputs, making sure managers are aware of the consequences regarding their management actions and how it could potentially affect terrestrial populations (Wipfli 1997; Li et al. 2016; Cloe and Garman 1996).

Brook Trout growth is influenced by many abiotic and biotic factors including turbidity, water temperature, competition and the interaction of these factors. Sweka and Hartman (2001) found that when turbidity was high Brook Trout switched feeding strategies and became more active causing them to use more energy, which led to lower growth rates. Water temperature is another aspect that influences Brook Trout growth, with preferred temperatures between 10-19ºC and the upper limit above 22ºC (Cross et al. 2013). When woody riparian habitat was removed resulting in higher stream temperatures, Brook Trout growth rates increased in central and western Wisconsin streams (Hunt 1979). Rainbow Trout *Oncorhyncus mykiss* and/or Brown Trout *Salmo trutta* living sympatrically with Brook Trout may negatively influence Brook Trout growth. Growth rates of newly emerged Brook Trout decreased when Rainbow Trout emerged concurrently due to interspecific competition (Rose 1986). Dewald and Wilzbach (1992) conducted a laboratory experiment and found that in the presence of Brown Trout, Brook Trout lost weight and contracted a disease, not from Brown Trout, that killed 33% of the Brook Trout. Temperature and intraspecific competition also affects Brook Trout growth. Robinson et al. (2010) found that when summer water temperatures were high and Brook Trout densities were similar across years, growth decreased, and water temperatures were still high but densities were low, growth did not change. Therefore, although water temperatures may be an important factor Brook Trout affecting growth, density can be just as important.

Brook Trout Abundance, Riparian Conditions and Stream Channel Modification

There is conflicting evidence about which riparian types, grassy or woody, most benefit trout populations. Wooded riparian areas provide large woody structure that creates overwintering habitat for fish and shade that reduces water temperatures during summer (Lyons et al. 2000a). These characteristics increase the likelihood for survival throughout the year. However, Peterson (1993) found Brook Trout to occur in significantly larger numbers in streams that have grassy riparian areas compared to woody areas in forested areas of New York. In a southeastern Wisconsin stream, Slawski (1997) found higher numbers of Brook Trout in a mixture of woody and grassy riparian areas compared to riparian areas dominated by one or the other. Managing for a mixture of both grassy and woody riparian areas could be the best management strategy to enable angler access and high trout abundance (Lyons et al. 2000a). Other factors influencing Brook Trout abundance are the presence of other trout species in the stream, specifically Brown Trout, which Fausch and White (1981) found to feed on juvenile Brook Trout.

Changes in trout abundance relative to riparian or watershed alterations most likely are influenced by geographic location and land use within the watershed (Lyons et al. 2000a). For example, Hunt (1979) found that trout abundances in two out of three central Wisconsin streams decreased each year after removal of woody riparian habitat over a period of four years. In contrast, logging conducted in 16 streams with forested watersheds in New Hampshire yielded no significant change in Brook Trout abundance (Nislow and Lowe 1996).

One technique used to modify stream channel morphology and habitat is bundling brush in stream channels often using recently removed riparian vegetation. Brush bundles are placed at the inside bends of streams and accumulate sediment and change channel morphology (Hunt 1993). The Wisconsin Department of Natural Resources evaluated the effects of brush bundles on the relative abundance of trout in five streams across the north-central Wisconsin and found that abundance varied in each stream (Avery 2004). Keith et al. (1998) also found variable abundance estimates for two different salmonid species when adding brush bundles in a southeastern Alaska stream. Based on these results, it cannot be conclusively stated that the addition of brush bundles will increase or decrease abundance of salmonids.

Aquatic and Terrestrial Invertebrates and Riparian Habitat

Riparian habitat and adjacent land use influences the abundance and diversity of aquatic and terrestrial invertebrates. Forested stream ecosystems have an abundance of terrestrial invertebrates (Baxter et al. 2005). Terrestrial invertebrates fall into the stream and become a part of the drift, along with aquatic invertebrates where they are then consumed by drift feeding fishes, such as Brook Trout. Hetrick et al. (1998) found that drift of both aquatic and terrestrial invertebrate was higher in forested riparian systems. Wipfli (1997) found that terrestrial invertebrate abundance varied with tree species and alder species and dense shrubs led to increased abundance. This could be due to more habitat or cover to escape terrestrial predators compared to old growth forests and grassy areas or better habitat for their prey, but Wipfli (1997) did not speculate on these factors. Mason and MacDonald (1982) found that riparian areas consisting of deciduous trees had higher inputs of terrestrial invertebrates compared to coniferous riparian forests. In contrast, streams with grassy riparian zones have higher abundances of benthic aquatic invertebrates because of increased sunlight penetration, which allows more periphyton to grow increasing their available food supply (Hetrick et al. 1998). However, open canopy, grassy areas support higher abundances of specific taxa (fewer taxonomic groups) and functional feeding groups, and have different taxa compared to closed canopy riparian areas because of different food sources (Wallace and Eggert 2009). Further, taxa varied depending on food type and availability (Wallace and Eggert 2009). Weigel et al. (2000) found that benthic aquatic invertebrate assemblages were more sensitive to changes within the watershed than riparian land use because invertebrate assemblage response was inconsistent in habitats from grassy buffers strips and intensive rotational grazing (IRG).

Terrestrial and aquatic invertebrate diversity is also related to riparian habitat. Watershed land use during the 1950's was found to be the best predictor of diversity of current aquatic invertebrate communities (Harding et al. 1998). Watersheds primarily consisting of agriculture lead to decreased diversity compared to forested watersheds (Harding et al. 1998). Conversely, forested watershed and riparian habitats had increased diversity in aquatic invertebrates compared to agricultural watersheds (Harding et al. 1998; Sponseller et al. 2001). Terrestrial invertebrate diversity decreases with intensive livestock grazing such as IRG because the cattle grazing on the plants causes a negative disruption in the plant-insect relationship (Kruess and Tscharntke 2002).

Physical and Chemical Attributes of Forest and Grassland Riparian Areas

Grassy riparian areas influence the physical and chemical properties of streams. Lyons et al. (2000b) found that, bank stabilization and the prevention of erosion was best when the streambank was one meter or less in height for southwestern Wisconsin streams influenced by grazing. This could be due to the shorter root system certain types of grasses have compared to woody vegetation. Grassy riparian areas often have narrow, deeper stream channels and more undercut banks compared to wooded riparian areas due to an increase in sediment trapping and storage (Lyons et al. 2000a). Increased bank stability and decreased bank erosion also can contribute to the narrower, deeper channels (Lyons et al. 2000b). Jackson et al. (2015) found that streams with grassy riparian areas had lower woody debris inputs and more uniform channel widths throughout the stream compared to streams with woody vegetation. The type of cover in grassy riparian areas includes aquatic macrophytes (Lyons et al. 2000b). However, grassy riparian areas have increased water temperatures due to lack of shading. Warmer water temperatures could potentially have harmful effects on coldwater fish species in the stream, especially trout (MacCrimmon and Campbell 1969). Overhanging grasses and undercut banks can act as a cooling mechanism for water temperature by decreasing sunlight penetration. Undercut banks in grassy riparian areas contain aquatic invertebrates that are readily available for foraging trout (Rhodes and Hubert 1991).

Woody riparian vegetation also influences stream morphology and associated biological communities. In general, woody riparian areas tend to create wider, shallower streams with more fine sediment (Dosskey et al. 2010; Hession et al. 2003). However, Rosgen (1996) found that streams with grassy riparian areas were wider than forested riparian streams. Jackson et al. (2015) found that gradients in a select few southeastern United States streams were greater when a forested riparian area was present compared to riparian areas lacking forest. These differences could be due to differences in soil condition, vegetation type, flow, slope, and disturbance history (Montgomery 1997). Wide, shallow streams are not as suitable for Brook Trout as are narrower, deeper streams. Woody riparian vegetation produces wood structure that provides good overwintering habitat for Brook Trout (Lyons et al. 2000a). Woody debris also create dams that retard streamflow minimizing flooding in downstream areas (Cunjack 1996). However, debris dams reduce the stream power and can cause more sediment to be deposited in the channel (Bennett et al. 2008). Woody riparian areas also deposit organic matter into the stream in the form of leaves, woody debris and other detritus that creates habitat for certain aquatic invertebrates (Lyons et al. 2000a).

Effects Woody Riparian Habitat Management

Experimental removal of woody riparian vegetation has been conducted in Wisconsin streams and in other states to evaluate the effect on salmonid species. Three streams around central and western Wisconsin, Little Plover River, Lunch Creek, and Spring Creek, were evaluated for changes in trout abundance and growth rates before and after the removal of Speckled Alder (Hunt 1979). On all three streams there were treatment zones, which consisted of the removal of Speckled Alder, and reference zones with no removal, and the Little Plover River also had a grassy meadow zone. Five years after removal, mean width decreased in two of the streams, mean depth varied in each stream, and average water temperatures were not significantly different in two of the three streams (Hunt 1979). Trout abundances in the treatment zones increased in only one stream and were slightly lower in the other two. However, growth rates of trout increased in all three streams. Overall, the results of this experiment were inconclusive because of variation in stream morphology and trout abundances and also abnormal weather conditions. Hunt (1979) suggested that collection of stream and fish data prior to removal is essential in determining if riparian brush removal is a viable option when trying to improve stream morphology and trout abundances and growth rates.

Ecological considerations must be weighed before removing woody vegetation. Lyons et al. (2000a) cautioned against removing woody riparian vegetation in forested watersheds and encouraged the use of grassy riparian areas only in agricultural or urban watersheds. Lyons et al. (2000a) also found that altering riparian habitat in reaches between 150-500 meters significantly affected bank erosion and stream channel morphology but did not significantly change the fish communities. Further, if removal of wood occurred, Lyons et al. (2000a) recommended grassy riparian reaches between 30-100 meters in length mixed in with longer stretches of woody riparian habitat. Davies and Nelson (1994) also recommended buffer strips of at least 30 meters in length. However, Barton et al. (1985) determined that a buffer of 10 meters for a 3 km stretch produced optimal weekly maximum temperatures for trout. Differences in recommended riparian buffer width depends on stream and riparian conditions and the goals set for the buffer (Mayer et al. 2005).

Intensive Rotational Grazing as a Riparian Management Tool

Active management is needed to maintain grassy riparian areas once woody vegetation is removed. One active management strategy is intensive rotational grazing (IRG) (Lyons et al. 2000b). Intensive rotational grazing, continuous grazing, grassy riparian zones, and woody riparian zones were studied in four southwestern Wisconsin trout streams to compare their effects on stream banks, stream habitat characteristics, and trout abundance (Lyons et al. 2000b). Results indicated that IRG maintained a grassy riparian zone and had minimal effects on bank erosion and no effect on fish communities (Lyons et al. 2000b). Buckhouse et al. (1981) also found that moderate grazing had little effect on streambank erosion. However, runoff and sediment loads increased in areas that were open to grazing (Rauzi and Hanson 1966). This could be due to trampling by the cattle, which decreased riparian vegetation that helps filter and retain runoff and sediments (Kauffman and Krueger 1984). Light to no grazing was found to increase fish production because of better habitat available to the fish compared to areas that were opened to heavier or more continuous grazing strategies (Bowers et al. 1979). One must be careful with grazing riparian areas as overgrazing causes wider stream channels, increased sediment and runoff, decreased fish production, and increases in water temperatures due to the lack of riparian vegetation (Kauffman and Krueger 1984). There is no published research on the effects of using IRG in Northern Lakes and Forests ecoregion to maintain grassy riparian areas following Tag Alder or any other woody riparian vegetation removal.

Objectives

To determine if riparian alteration and the addition of brush bundles:

1. Narrows and deepens the stream channel
2. Increases macroinvertebrate abundances
3. Decreases terrestrial invertebrate abundances and diversity
4. Changes Brook Trout diets before and after alteration

2. Methods

Study Area

This study was conducted in Green Meadow Creek in Lincoln County, Wisconsin in the Northern Wisconsin Forest Ecoregion (Figure 1). There were four study sites located along a 1000-m segment of Green Meadow Creek. Reconnaissance of the study stream was conducted in the summer and fall of 2016. The treatment site is 271 meters long and is dominated by Tag Alder in the riparian area. This stream reach is wide, average of 4.36 meters, and comprised of mostly gravel and sand substrate. The habitat type is primarily runs and riffles with few pools. The three reference sites are approximately 100 meters in length. A Tag Alder reference site located downstream of the treatment site has similar physical and riparian characteristics as the treatment site. This reach is comprised primarily of pools and runs. A grassy riparian reference site is located downstream of the Tag Alder reference site. The riparian habitat in this reach is a sedge meadow. A forested riparian reference site is located upstream of the treatment site. The riparian habitat in this reach is mature forest with ferns in the understory. Stream habitat is primarily gravel and there is an even number of riffles, runs, and pools. The treatment site is longer than the reference sites because it encompasses the entire section of land that the landowner wanted Tag Alder to be removed from. The reference sites reach length was determined by average width according to the “Guidelines for Evaluating Habitat in Wisconsin Wadable Streams” (Simonson et al. 1994).

Experimental Design

The treatment site was selected by the landowner as the section of the stream where they want Tag Alder removed (Figure 1). The Tag Alder reference site was then selected based on similar hydrological characteristics and riparian vegetation and proximity to the treatment site. Two other reference sites were selected to represent the different types of riparian vegetation that can be found in northern Wisconsin streams. These two reference sites also are indicative of what the riparian area may look like after alteration has taken place. The sample population will not be representative of all Wisconsin streams so interpretations of the data will be restricted to the dataset and other streams with similar hydrological, biological, and riparian characteristics in the Northern Wisconsin Forest Ecoregion.

The experimental treatment was the removal of Tag Alder in approximately 30-foot strips along both streambanks. Some of the removed Tag Alder was used for brush bundling in the stream to alter channel morphology (stream width and depth) and substrate. Removal and brush bundling of Tag Alder took place at the treatment site.

Field Methods

Stream and riparian habitat.—Two permanent transects were installed at each reference site for stream channel measurements. The treatment site received four permanent transects based on the size of the reach. Permanent transects were marked with rebar in the bank. Stream channel measurements were gathered pre- and post-treatment at each site from May to September using the USDA Forest Service methodology ( ), which included measuring channel width and depth. We measured stream substrate, bank erosion, riparian land use, depth to fines, and canopy cover at transects perpendicular to the stream (Simonson et al. 1994). Each transect width was divided by five to obtain four evenly spaced points. Depth and depth to fines was measured at each point using an aluminum meter stick and recorded in inches where it was later converted to centimeters. Substrate type and percentage was measured at each point using a 0.3048-meter by 0.3048-meter square where percentages of substrate type were quantified visually. Canopy cover was measured at each point using a spherical densiometer. There were two ways bank erosion was measured. The first was to measure the length of continuous bare soil, on both sides, to the nearest 0.01 meters within one meter of the stream. The second method was to measure the percent of eroded bank to the crest or within five meters of the stream edge. This was done by measuring the length of continuous bare soil and dividing it by either how far it is the crest or by five. Riparian land use was measured as the percent of both the left and right bank within the 5 meters of the stream edge under the following the categories: cropland, pasture, barnyard, developed, meadow, shrubs, woodland, wetland, exposed rock, and other. Riparian buffer width was measured as the length of undisturbed land uses within 10 meters of the stream.

In-stream chemical characteristics were measured monthly from May to September 2017 at each site. A Hydrolab MS5 - multiparameter mini sonde was used to measure dissolved oxygen, conductivity, and pH. A HOBO Water Temperature Pro v2 Data Logger was installed at each site to record water temperatures every 15 minutes. Data was offloaded monthly during May to September during collection of channel morphology data.

Riparian alteration.—Before the Tag Alder removal, brush bundles were staked out at the inside bends of the stream by the Wisconsin Department of Natural Resources personnel using 3-foot wooden stakes and a hydraulic pounder attached to a UTV. Tag Alder removal along the stream and brush bundling took place in June 2017 with the help of volunteers and landowners. Tag Alder was cut using chainsaws and bow saws in strips of about 30 feet (Hunt 1979). Tag Alder over 30 feet away from the stream was removed in early October 2017. Trees not being used for brush bundling were placed in a pile away from the stream. To construct brush bundles, trees were cut down to appropriate size based on the size of the bundle are held in place using wooden stakes and bailing twine. Twine was tied to each stake and run across the top creating a spider web design to ensure that the bundle will not fall apart. Once removal and bundling is completed, the landowner will integrate the treatment site into his rotational grazing system as an active management technique to promote grassy vegetation growth (Lyons et al. 2000).

Fish sampling.--Brook Trout populations were sampled May through September using a multiple pass depletion method. A backpack electrofisher (Model LR-20B, Smith-Root, Inc.) was used to sample fish. One assumption of this survey method is that it is a closed population (Lockwood and Schneider 2000). To ensure that the sample reach is closed, two block seines, 9.144 meters by 1.219 meters, were set at each end of the stream reach to inhibit fish immigration and emigration. The sample reach for all sites, excluding the treatment site, was calculated by taking the mean stream width (MSW) and multiplying by 35. This resulted in all reaches being approximately 100 meters in length. For the treatment site, a 100-meter section was randomly selected for fish sampling due to the site being 271 meters in length. After each pass, all fish caught were identified to species and length was recorded. Stomach contents were then removed from Brook Trout using a gastric lavage (Light et al. 1983). The gastric lavage device was made up of a 12cc disposable syringe and 3.175 mm tubing. This is non-lethal and does not affect growth or survival of Brook Trout larger than 50 mm (Hafs et al. 2011). The syringe was filled with stream water and the tube placed through the mouth and into the stomach where the water was then flushed out, forcing stomach contents out through the mouth. The stomach contents then fall into a small funnel with a sieve attached where they were then flushed into a container with 80% isopropyl alcohol for preservation. Diet contents were identified down to the lowest taxonomic level. After fish length was measured and stomach contents removed the fish were placed outside of the downstream block seine to ensure they did not reenter the sampling area. Electrofishing time and reach length was recorded to estimate catch per unit effort.

Invertebrate sampling.--Aquatic invertebrate sampling was conducted pre- and post-treatment from May to September 2017 at each site using a Surber sampler and D-net. Terrestrial invertebrates were sampled with pan traps during the same time period. Stratified random sampling was used to sample aquatic invertebrates with the strata consisting of riffles and runs for Surber samples and bank side vegetation for D-nets. Samples were allocated proportionally based on the number of riffles, runs, and pools in each stream site. At each sampling location where the Surber Sampler was used, a D-net was also used to sample invertebrates near the bank or in the overhanging grasses. To standardize D-net sampling, two consecutive scoops or dips were performed. Samples from the Surber were emptied onto a sieve where material not containing invertebrates were discarded and the rest placed into a container with 80% Isopropyl alcohol for preservation. The same process was used for D-net samples.

Pan traps for terrestrial invertebrate sampling consisted of aluminum pans 33 cm by 23 cm attached with clips to wooden stakes pounded into the stream bottom. Three pan traps were placed in the stream at the downstream, midstream, and upstream portions and let sit for one week. Water from the stream was placed into the pan along with a few drops of liquid soap to ensure insect capture (Rundio and Lindley 2012). After one week the pans will be poured through a sieve where invertebrates will then be placed into containers with 80% Isopropyl alcohol for preservation. All invertebrates were identified to the Family level while any that were not identifiable to Family were classified to Order (Utz and Hartman 2007, Ersbak and Haase 1983).

Laboratory Methods

Terrestrial and aquatic invertebrate samples were brought back to the lab after collection and preservation. The samples were then identified down to the lowest taxonomic level using a stereomicroscope.

Preserved stomach contents from Brook Trout were brought back to the lab and identified down to the lowest taxonomic level. Percent by volume was then measured for each fish stomach using a graduated cylinder and the water displacement method (Hynes 1950). The graduated cylinder was filled with 100 mL of water and the stomach contents were then placed into the cylinder. The water level was then measured again. Percent by volume was then calculated by taking the difference in water levels divided by 100. Dry weights were then measured by oven drying stomach contents at 65 C for 24 hours (Elliott and Persson 1978).

Data Analysis

To examine the differences in stream habitat pre- and post-treatment for mean stream width, depth, depth to fines, substrate, bank erosion and canopy cover paired t-tests were performed with alpha set at 0.05. Analyses of Variance (ANOVA) were performed for the same metrics listed above to compare between the four study sites.

Shifts in Brook Trout diets were quantitatively described using frequency of occurrence, proportion by weight, and prey selectivity methods (Garvey and Chipp 2007). Frequency of occurrence is represented by the following equation:

$$O_i = \frac{J_i}{P}$$

where J_i is the number of fish containing prey i and P is the number of fish that have food in their stomach.

This analysis provides information about how often a prey item is consumed but provides no information regarding the importance of the prey item to the overall diet (Garvey and Chipps 2007). Percent composition by number (%N) is represented by the following equation:

$$\%N = (p_i/p)$$

where p_i is the total number in each prey category from an individual stomach and p is the total number of prey for all categories from the same stomach.

This method is biased towards smaller more abundant prey items but is important for calculating other metrics, such as prey selectivity. Prey selectivity will be calculated using the log of the odds ratio (LOR) (Schabetsberger et al. 2003). LOR is represented from the following equation:

$$LOR = \ln(d_i\,(100 - e_i)/e_i(100 - d_i))$$

where d_i is percentage of prey category in the diet and e_i is the percentage of prey category in the environment.

This method will allow us to quantify the changes in selectivity of the Brook Trout pre- and post-treatment. Wet weight and an electronic balance will be used to calculate percent composition by weight (%W) (Garvey and Chipps 2007). Percent composition by weight is represented by the following equation:

$$\%W = (w_i/w)$$

where w_i is the weight of each prey category found in an individual fish's stomach and w is the total weight of all prey categories found in the same stomach.

Brook Trout relative abundance estimates from the multiple-pass depletion survey was calculated using trout per mile (Garvey and Chipps 2007, Hunt 1979).

Relative abundance estimates were compared pre- and post-treatment and between sites using paired t-tests. Population size estimates from the maximum-likelihood estimator will also be compared pre and post treatment and between sites.

Aquatic and terrestrial invertebrate data collected from Surber Samplers, D-nets, and pan traps respectively was used to estimate abundance and diversity. Relative abundance estimates at number of individuals per m^2 for each order and family was calculated based on Surber Sampler, D-net, and pan trap dimensions. Diversity was calculated using the Simpson Index of Diversity (D) as it is more weighted on dominant species and has more biological significance (Washington 1984). This is calculated using the following equation:

$$Simpson\ Index\ of\ Diversity = 1 - D$$

where D is represented by the following equation:

$$D = \frac{1}{\sum_{i=1}^{s} p_i^2}$$

where *s* is the number of taxon and *p* is the proportion of individuals from one particular taxon divided by the total number of individuals found. This index was calculated using the vegan package in Program R.

3. Conclusions

Results from this study suggest that riparian alteration and the addition of brush bundles has the potential to decrease stream width and increase depth. The removal and bundling took place in June and by the end of the study (September) the width was already decreasing and depth increasing. Given continued monitoring at the permanent transects at the treatment site I would expect this trend to continue along with a higher percent stream bottom covered in gravel due to scouring. Continued monitoring of water temperature would also be imperative to ensure water temperatures at the treatment site stay within the Brook Trout preferred ranges (10-19 °C).

Results also suggest that riparian alteration and the addition of bush bundles does not influence relative abundances of Brook Trout. However, these are short-term results that have the potential to change over a more prolonged monitoring period. The same thing can be said for changes in diet contents too. Post-treatment diets suggested that Brook Trout started selecting less for larval Odonates and more for adult

Ephemeropterans and Dipterans, larval Trichopterans, and gastropods. However, these changes could be due seasonal preferences as Brook Trout have documented diet shifts in summer months from aquatic to terrestrial invertebrates. Diet data also suggested that, although, adult Dipterans made up a large number of diet contents they provided less overall weight to the diet while adult Ephemeropterans and Trichopterans provided a more substantial amount to diets. Again, continued monitoring would allow more insight into the true effects of the riparian alteration on Brook Trout populations. Future monitoring could also involve studying Brook Trout movement between sites, which could potentially influence diets. Movement could be studied using fin clipping or PIT tags.

Results from aquatic invertebrate sampling suggested that the riparian alteration has not had a direct influence abundance estimates from Surber and D-net samples post-treatment. Abundances of all taxa involved remained similar pre- and post-treatment. Abundances of terrestrial invertebrates were similar increased slightly for all taxa excluding Dipterans. Riparian alteration may have had an effect but changes are likely due to seasonal variation in abundances. Also, only one month of pre-treatment data was gathered due to flooding washing away June pan traps. Therefore, I cannot say definitively if riparian alteration had an effect of terrestrial invertebrate abundances. Diversity estimates of aquatic invertebrates at the treatment site increased post-treatment suggesting that riparian alteration may have an effect on aquatic invertebrate diversity. However, watershed land use may be more of a driving factor for aquatic invertebrate diversity. Terrestrial invertebrate was variable throughout the study suggesting that riparian alteration has not had an effect on terrestrial invertebrate diversity. However, there does seem to be a riparian vegetative influence on diversity as the woody riparian sites had higher diversity estimates compared to the sedge meadow reference site. Therefore, continued monitoring of the treatment site should take place as the vegetation will continue to become more grass dominated compared to woody.

Continued monitoring should take place at the treatment site over the next several years to see the true effects of the alteration and addition of brush bundles. This would also help increase sample size, which would then help increase the power of statistical analyses. A reference stream with similar stream and riparian characteristics should also be added to help provide better comparisons and provide more definitive conclusions regarding the effects of the Tag Alder removal. Overall, riparian alteration and the addition of brush bundles will influence stream morphology with the potential to alter Brook Trout populations, increase aquatic invertebrate diversity, and alter abundances of both terrestrial and aquatic invertebrates.

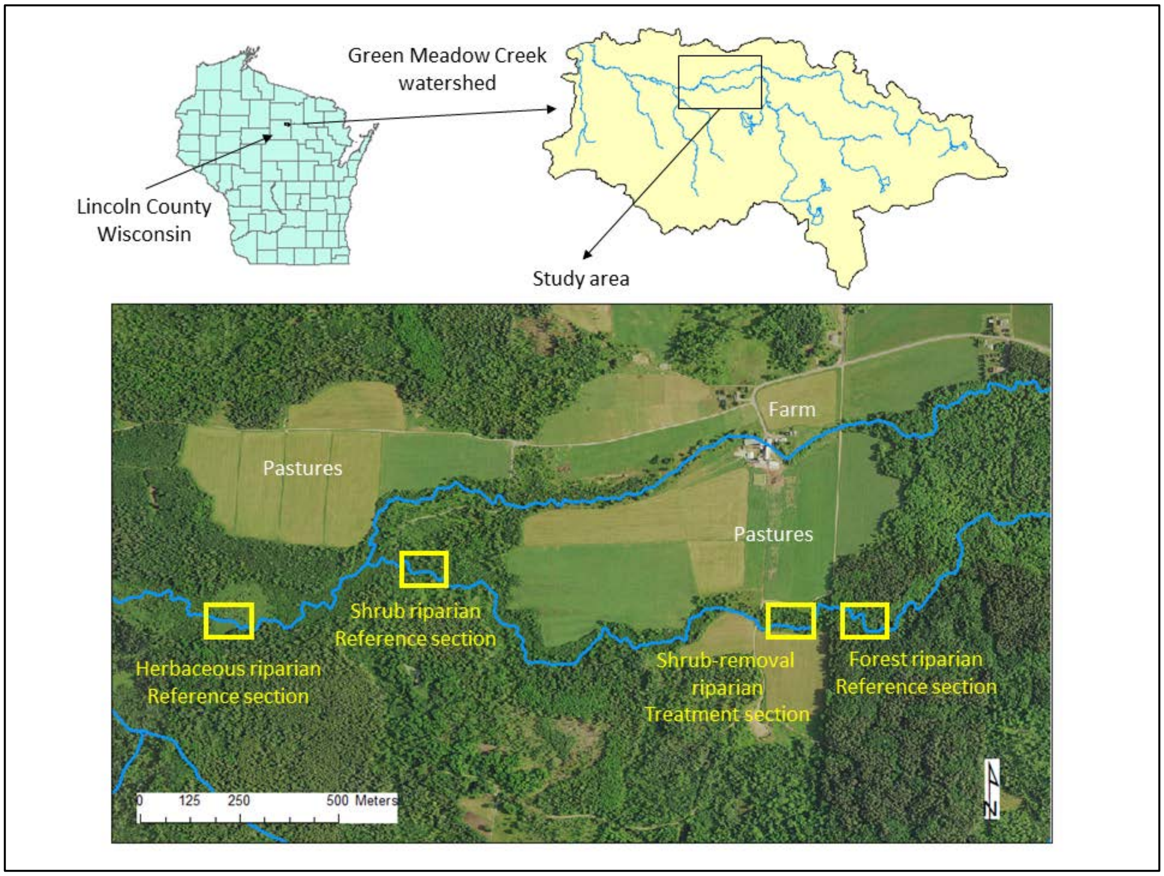

Figure 1. Figure of Green Meadow Creek located in Lincoln County, WI with study sites.

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4. North American Journal of Fisheries Management submission

The Influence of Riparian Vegetation on Brook Trout Diets and Habitat

Abstract

Riparian vegetation influences stream morphology, habitat and biological communities providing a source of connectivity between aquatic and terrestrial ecosystems. The objective of this study was to determine how riparian vegetation, including the removal of Tag Alder in the riparian corridor and addition of brush bundles, effects Brook Trout Salvelinus fontinalis *habitat and feeding and invertebrate communities. We sampled four sites, May through September 2017, with different types of riparian vegetation at Green Meadow Creek Lincoln County, WI. Three of these sites were references that represent the different types of vegetation found in the area, second growth forest, shrub or Tag Alder* Alnus incana*, and sedge meadow. The fourth site was experimental and consisted of a Tag Alder riparian area. Here, Tag Alder was removed and used as brush bundles within the stream to alter morphology. Trout were sampled using an electrofishing depletion method and stomach contents were obtained using a gastric lavage. Aquatic invertebrates were sampled using a Surber sampler and D-net while terrestrial invertebrates were sampled using pan traps. At the treatment site, brush bundles are already narrowing the stream while slightly increasing depth. The percentage of gravel has increased at the treatment site while the percentage of sand has decreased. Brook Trout selection of prey varied with the different types of riparian vegetation. Brook Trout in woody riparian areas feed more on terrestrial Trichopterans and Ephemeropterans while trout at grassy riparian site feed more on Ephemeroptera nymphs and terrestrial Dipterans. Selection for terrestrial invertebrates is highest during*

summer months and lowest during the late spring and early fall. Terrestrial invertebrate abundances are highest during the mid summer months while aquatic invertebrate abundances are highest during the late spring and early summer. The removal of Tag Alder and addition of brush bundles have begun to alter channel morphology and those changes, coupled with the alteration of riparian vegetation, may alter diets of Brook Trout.

Riparian zones connect terrestrial and aquatic ecosystems and affect the biological, physical, and chemical characteristics of the stream. This connectivity provides reciprocal subsidies of energy flow between aquatic and terrestrial ecosystems including aquatic invertebrate emergence into the riparian area and terrestrial inputs of invertebrates into the stream from the riparian zone (Nakano and Murakami 2001). Vegetation within riparian areas is influenced by geographic location and the successional stage of the surrounding land (Gregory et al 1991), as well as past disturbances and present land use within the watershed (Harding et al. 1998). Different types of riparian vegetation along with invertebrate abundances can affect trout prey selection of both aquatic and terrestrial invertebrates (Nakano and Murakami 2001; Wilson et al. 2014).

Brook Trout *Salvelinus fontinalis* populations are influenced by stream habitat and riparian vegetation, that can be managed to restore previous conditions and alter current ones. Brush bundling, a stream management technique that involves removing shrub riparian vegetation and placing some of it in the stream to modify (e.g., narrow and deepen) channel morphology, has been used in several central Wisconsin forested streams to restore and enhance habitat in trout streams (Hunt 1993; Avery 2004). Removing riparian vegetation can lead to a loss of shade resulting in increased stream temperature (Beschta 1997), which can have a profound effect on where Brook Trout occur in a stream (MacCrimmon and Campbell 1969). Barton et al. (1985) found that trout streams in southern Ontario had weekly maximum water temperatures that were less than 22°C compared to warmer non-trout streams. There is differing evidence about which riparian types, grassy or woody, produce the best trout habitat (Slawski 1997;) However, managing for a mixture of both grassy and woody riparian areas could be the best management strategy to enable angler access and high trout abundance (Lyons et al. 2000).

Brook Trout feed on both aquatic and terrestrial invertebrates and their preference for them changes seasonally. During late spring and summer, salmonids feed primarily on terrestrial invertebrates that fall into the stream, which can provide up to 50% of their annual diet, whereas during fall and winter they feed mostly on benthic aquatic invertebrates (Baxter et al. 2005). Wilson et al. (2014) found an inverse relation between the biomass of benthic invertebrates and terrestrial invertebrates in Brook Trout diets; specifically, that an increase in benthic invertebrate biomass led to a decrease in feeding on terrestrial invertebrates; however, the terrestrial invertebrates were still available to consume but not selected for. Several studies have related riparian canopy cover to the abundance of salmonids, such as Coho Salmon *Oncorhynchus kisutch*, Dolly Varden *Salvelinus malma*, Cutthroat Trout *Oncorhynchus clarki*, and Rainbow Trout *Oncorhynchus mykiss* (Hetrick et al. 1998 and Wilzbach et al. 2005); however, there is no literature on how feeding and diets vary with different types of riparian vegetation, specifically for Brook Trout.

The objective of this study was to determine how riparian vegetation, including the experimental removal of Tag Alder in the riparian corridor and the addition of brush bundles, affects Brook Trout habitat and feeding and invertebrate communities in a headwater stream. We sampled Brook Trout populations, their diets and available prey to assess differences among forested, shrub and grass riparian areas in a headwater stream in the north central forest region of Wisconsin. Riparian Tag Alder was removed from one section of the stream and added as brush bundles to evaluate the response of Brook Trout abundance and feeding to riparian alteration and stream channel modification.

Methods

Study Site

This study was conducted in Green Meadow Creek, a 13.16 km third-order, cool-warm stream in Lincoln County, Wisconsin (WIDNR 2011). Land use within Green Meadow Creek's watershed (179.35 mi^2) is primarily forests and wetlands (88.3%) with the rest a mixture of grasslands and agriculture (WIDNR 2011). Green Meadow Creek originates in Jackknife Lake located in Harrison, Wisconsin (also known as Harrison Hills flowage area). This area has steep gradients (15-35%) in the headwaters that flatten out further downstream. Green Meadow Creek drains into Lake Alice of the Wisconsin River in King, Wisconsin.

We sampled four sites, one treatment and three reference sites, within a 1,900 m stretch of the stream May through September 2017 (Figure 1). Reconnaissance and preliminary sampling of the stream was conducted in the summer and fall of 2016 to characterize riparian and stream habitat. The treatment site was 271m long and was dominated by Tag Alder *Alnus incana* in the riparian area. Other less dominant tree species in the riparian corridor included Paper Birch *Betula papyrifera*, Black Ash *Fraxinus nigra*, and White Spruce *Picea glauca*. The treatment site was longer than the reference sites because it encompasses the entire section of the stream where Tag Alder was removed. A Tag Alder reference site, 100 m in length, located 1,054 m downstream of the treatment site had similar physical and riparian characteristics as the treatment site. This reach was comprised primarily of pools and runs. A grassy riparian reference site, 100 m in length, was located 480 m downstream of the Tag Alder reference site and consisted of Brownish Sedge *Carex brunnescens* for riparian vegetation. A forested riparian reference site, 100 m in length, was located 254 m upstream of the treatment site. The riparian habitat in this reach was second growth forest consisting of White Pine *Pinus strobus*, Red Pine *Pinus resinosa*, and Quaking Aspen *Populus tremuloides.* Stream habitat was primarily gravel and there was an even number of riffles, runs, and pools. The soil at the sedge meadow reference site was classified as muck with 0-2% slope while the soil type at the mature forest reference site was classified as muck and sandy loam with 15-35% slope. For comparison, the Tag Alder reference site was classified as muck and loamy sand with 0-4% slope and the treatment site was classified as muck and sandy loam with 0-6% slope.

Field Methods

<u>Stream and Riparian Habitat –</u> Reach lengths for the three reference sites were 100 m (approximately 35 times the mean stream width, MSW) (Lyons 1992; Simonson et al. 1994). Transects within each reach started at the downstream end and moved upstream, with spacing between each equal to two MSW (Simonson et al. 1994). Measurements of stream habitat, channel morphology, and riparian conditions occurred at each transect in the summer and fall of 2016 (Simonson et al. 1994). Measurements were then performed again in 2017. These measurements included width, depth, depth to fines, substrate, canopy cover, riparian buffer width, riparian land use within 5 m of the stream edge, bank erosion as the length of continuous bare soil within 1 m of the stream, and bank erosion as the percent of eroded bank to the crest within 5 m of the stream edge (Simonson et al. 1994). Depth and depth to fines were measured using a meter stick and width was measured using a steel tape. Substrate was visually classified in a 0.3048-meter by 0.3048-m square and classified into the categories following Platts et al. (1983), Rankin (1989), and Simonson et al. (1994). Canopy cover was measured using a spherical densiometer. Riparian land use was visually assessed as the percent of both the left and right bank within the 5 m of the stream edge under the following categories: cropland, pasture, barnyard, developed, meadow, shrubs, woodland, wetland, exposed rock, and other. Riparian buffer width was measured as the length of undisturbed land within 10 m of the stream. Two permanent transects were installed at each reference site to monitor stream habitat, channel morphology, and riparian conditions. We installed four permanent transects at the treatment site based on the size of the reach. Permanent transects were monumented with rebar stakes in the stream bank. Stream channel measurements were made at the permanent transects from May through September of 2017.

In-stream chemical characteristics were measured at each site from May to September 2017. A multiparameter mini sonde (Hydrolab MS5) was used to measure dissolved oxygen, conductivity, and pH. A water temperature logger (HOBO Water Pro v2) was installed at each site to record water temperatures every 15 minutes. A water level gaging station with air and water pressure loggers was installed between the sedge meadow and Tag Alder reference sites.

Riparian Treatment – Before we removed Tag Alder at the treatment site, areas for placement of brush bundles were marked on the inside bends of the stream and 0.9 m wooden stakes were installed using a hydraulic pounder attached to a utility vehicle. Tag Alder was mechanically removed from the treatment site in 9-meter strips adjacent to the stream in June of 2017 using chainsaws and bow saws. Branches were trimmed from trees and placed in the areas previously staked out for the brush bundles. Twine was tied to each stake and run across the top creating a spider web design to ensure that the bundle was secure. All Tag Alder that was further than 9 m away from the stream was removed in October of 2017 and not used for brush bundling. All Tag Alder not used for brush bundling was piled in the floodplain for the landowner to dispose of to encourage the growth of grassy vegetation.

Fish Sampling – Preliminary fish sampling using a backpack electrofisher (Model LR-20B, Smith-Root, Inc.) and a single pass method occurred in fall 2016 to gather basic fish assemblage data for Green Meadow Creek. During May through September of 2017, Brook Trout populations were sampled monthly using a multiple pass depletion at all stream sites (Van Deventer and Platts 1983; Panek and Densmore 2013). Three passes were performed as apart of the survey unless the first two passes did not capture any Brook Trout. To ensure that the sample reach was closed to meet the closed population assumption (Lockwood and Schneider 2000), two block seines, 9.144 meters by 1.219 meters, were set at each end of the stream reach to inhibit fish immigration and emigration. Fish were identified to species and length was recorded for all fish. Electrofishing time was recorded for each pass and reach length was noted to estimate catch per unit effort.

Stomach contents were obtained from Brook Trout using a gastric lavage (Light et al. 1983). Each stomach was pumped three times and sampling occurred either in the morning or early afternoon. This is a non-lethal method for obtaining diet data and does not affect growth or survival of Brook Trout larger than 50 mm (Hafs et al. 2011). Brook Trout 140mm and over were used for gastric lavage sampling. The gastric lavage was comprised of a 12cc disposable syringe and 3.175 mm vinyl tubing. Stomach contents were stored in 80% isopropyl alcohol and identified down to Order.

Invertebrate Sampling – Aquatic and terrestrial invertebrates were sampled monthly at all stream sites from May through September 2017. Aquatic invertebrates were sampled using a Surber sampler and a D-net with 500-micron nylon (Stark 1993). The Surber sampler was used to sample riffle and run invertebrates and the D-net was used to sample invertebrates near the bank or in the overhanging vegetation. Terrestrial invertebrates that fell into the stream were sampled using pan traps filled with water and 2 drops of soap (Baxter et al. 2004). Three pan traps were deployed at each site for one week before being sieved into a storage container with 80% isopropyl alcohol. Pan traps were located at the downstream, midstream, and upstream locations of the stream reaches. All invertebrates were brought back to the lab and identified down to the Family level.

Data Analysis

Pre- and post-treatment differences in stream habitat, channel, and riparian measurements at the Tag Alder treatment site were evaluated with paired t-tests. Pre-treatment data was collected in the fall of 2016 and May and June of 2017. Post-treatment data was collected July through September 2017. The Shapiro-Wilks test for normal distribution was performed for each parameter along with Levene Test for Homogeneity of Variances ($\alpha \leq 0.05$).

Repeated measures Analysis of Variance (ANOVA) was used to compare habitat, channel, riparian and water temperature measurements among the four study sites ($\alpha \leq 0.05$) using the same procedure described above for testing normality and homogeneity of variances. If the tests for normality or equal variances failed, the data were then log transformed. If the log transformation did not pass tests for normality or equal variance then the non-parametric Kruskal-Wallis Rank Sum Test was performed.

Brook Trout diets were quantitatively described by site using frequency of occurrence, percent by volume using the water displacement method, proportion by weight using oven dry mass, and prey selectivity methods (Garvey and Chipps 2007). Prey selectivity was calculated using the log of the odds ratio (LOR) (Schabetsberger et al. 2003). LOR is represented from the following equation:

$$LOR = \ln(d_i\,(100 - e_i)/e_i(100 - d_i))$$

where d_i is percentage of prey category in the diet and e_i is the percentage of prey category in the environment.

Absolute and relative abundance estimates (trout per mile) and lengths of Brook Trout were calculated (Garvey and Chipps 2007) for each site and month. Trout per mile estimates were compared pre and post treatment and among sites using paired t-tests and ANOVA. Estimates of population size based on the *k-pass removal* method were calculated using the Carle and Strub (1978) method with the FSA package (Ogle 2016) in Program R, which is a maximum weighted likelihood estimator. This method has reduced bias and variance when compared to other removal methods (Carle and Strub 1978; Seber 2002) and also estimates probability of capture and population size more accurately (Hedger et al. 2013). Population size estimates were also compared pre and post treatment and among sites. Brook Trout lengths were compared among sites using the Kruskal-Wallis test.

Aquatic and terrestrial invertebrate data collected from Surber Samplers, D-nets, and pan traps was used to estimate abundance and diversity. Relative abundance estimates were calculated for D-net and pan trap samples based on net and pan dimensions, respectively. Absolute abundance estimates (930 cm^2) were calculated for Surber samples. Diversity was calculated using the Simpson Index of Diversity as it is more weighted on dominant species and has more biological significance (Washington 1984). This index was calculated using the Vegan package in Program R.

Results

Stream habitat.— Stream habitat differed longitudinally from the upstream mature forest reference site to the downstream sedge meadow reference site (Table 1). The mature forest site was steeper, wider, and shallower compared to the sedge meadow reference site, whereas the Tag Alder reference site and Tag Alder treatment site had similar width and depth but the reference site had a shallower slope. Depth to fines was highest in the sedge meadow site and steadily decreased upstream towards the mature forest site. The width-to-depth ratio was lowest in the sedge meadow site and increased upstream with the Tag Alder treatment site and mature forest site having similar ratios. Percent gravel increased and percent sand decreased from the sedge meadow site to the mature forest site. Shading was higher in the forest and Tag Alder sites compared to the sedge meadow site.

Stream and riparian habitat at the treatment site differed between pre- and post-treatment time periods (Table 2). The stream was significantly narrower, and depth increased post-treatment following the addition of brush bundles. Gravel increased and sand decreased significantly, whereas both cobble and detritus decreased. Shading significantly decreased following the removal of Tag Alder.

Mean monthly water temperatures ranged from 12.21-17.59 ºC across all study sites (Table 3). The sedge meadow reference site consistently had the lowest average monthly water temperature among all sites. The treatment and mature forest sites had significantly higher water temperatures compared with the Tag Alder reference and sedge meadow sites from July through September.

Water level data indicated multiple significant precipitation events throughout the summer months, with the largest events occurring in mid to late June, early August, and mid-August (Figure 2). Following these large precipitation events, water levels receded to base flow levels within one week.

Brook Trout abundance.— A total of 106 Brook Trout were captured and population estimates and relative abundance varied longitudinally among sites. The two upstream sites, treatment and mature forest reference, yielded the highest density and relative abundance of Brook Trout throughout the study, whereas the two downstream sites, sedge meadow and Tag Alder reference, yielded the lowest estimates (Table 4). Pre-treatment means at the treatment site, 15.5 (0.71 SD) for population density and 266 (12.73 SD) for trout/mile, were not significantly different (t-stat=1.73, p=0.21, for population density) and (t-stat=-0.39, p=0.73, for trout/mile) from post-treatment mean density 12.33 (3.05 SD) and 226 trout/mile (55.97 SD). Population estimates remained fairly constant for the treatment site while abundance at the mature forest reference site increased each month. The highest CPUE was also found at the mature forest reference site and the treatment site (Table 4).

The size of Brook Trout did not differ longitudinally among study sites. The Tag Alder reference site had the highest mean length at 181.24 mm (33.94 SD, min=114, max=245) while the treatment site had the lowest mean length at 164.96 mm (47.58 SD, min=78, max=280). For comparison, the sedge meadow reference site had a mean length of 176.09 mm (11.07 SD, min=159, max=198) and the mature forest reference site had a mean length of 170.97 mm (53.02 SD, min=102, max=290). Significant differences were not detected for length among sites (Kruskal-Wallis, df=3, H=3.30, p=0.34).

Brook Trout diets and invertebrate communities.— Aquatic invertebrate abundances differed longitudinally from the downstream sedge meadow site to the upstream mature forest site. In general, the density (Surber samples) of Ephemeroptera, Plecoptera, and Trichoptera increased while the abundances of Odonata and Coleoptera decreased from downstream to upstream (Figure 3). The mature forest and treatment sites had the highest abundances of Plecoptera, whereas the sedge meadow site had the lowest abundance of this Order. The treatment site had the highest abundance of aquatic Trichoptera, whereas the sedge meadow site had the lowest. Larval Diptera did not differ longitudinally, but were most abundant in the sites with Tag Alder. Nearshore (D-net samples) Plecoptera and Trichoptera abundance increased whereas Ephemeroptera, Odonata, and Coleoptera abundance decreased moving upstream (Figure 4). Larval Diptera abundance differed little nearshore among sites.

Terrestrial invertebrate abundances (pan trap samples) also differed longitudinally. From the downstream sedge meadow to the upstream mature forest, abundances of Ephemeroptera and Plecoptera increased while the Diptera abundances decreased (Figure 5). Terrestrial Trichoptera showed little longitudinal differences; the highest abundances occurred at the Tag Alder reference and the treatment sites (Figure 5). The highest abundance of Diptera occurred at the sedge meadow site while the lowest was at the treatment site (Figure 5).

Diversity estimates for aquatic and terrestrial invertebrate families varied longitudinally from the downstream to upstream sites. Diversity estimates for the aquatic and terrestrial invertebrates increases as movement from the sedge meadow reference site to the mature forest reference occurs. Diversity for both types of invertebrates at the sedge meadow reference site increased monthly from May to September of 2017.

I examined 81 Brook Trout stomachs (9 at the sedge meadow reference, 13 at the Tag Alder reference, 28 at the treatment site, and 31 at the mature forest reference site). Brook Trout in the sedge meadow reference section fed primarily on Ephemeroptera nymphs and terrestrial Dipterans (Figure 6). Ephemeroptera nymphs, gastropods, and aquatic and terrestrial Trichopterans were the primary items consumed at the Tag Alder reference site. Brook Trout at the treatment site primarily ate Ephemeroptera nymphs, larval Trichopterans, and terrestrial Dipterans, whereas those at the mature forest reference site primarily consumed terrestrial Dipterans and Ephemeroptera nymphs.

Ephemeroptera nymphs had the highest frequency of occurrence among all the study sites in May through September 2017 (Table 5). However, Ephemeroptera nymph frequency did decrease slightly between June and July. Overall, Ephemeroptera nymphs, terrestrial Trichoptera, larval Diptera and adult Diptera were the four most frequent organisms to occur in Brook Trout diets. Frequency of occurrence of Ephemeroptera nymph and larval Trichoptera was highest in the treatment site pre-treatment while the most frequent organisms post-treatment were larval and adult Dipterans. Terrestrial Ephemeroptera and Trichoptera decreased slightly post-treatment.

Larval Dipterans were found to be frequent in occurrence but did not make up a large portion of the overall weight in the diet (Table 5, Table 6). Ephemeroptera nymphs and larval Odonata provided the largest amount of weight to pre-treatment Brook Trout diets at the treatment site while adult Ephemeropterans and Trichopterans provided the largest post-treatment weights.

Prey selectivity by Brook Trout differed from the downstream sedge meadow site to the mature forest site (Figure 7). Selectivity for Ephemeroptera nymphs and larval Diptera was lowest at the sedge meadow site and highest at the mature forest site, whereas selectivity for larval Trichoptera was highest at the sedge meadow site and lowest at the mature forest site with the Tag Alder reference and treatment sites having similar selection for larval Trichoptera.

At the Tag Alder treatment site, Brook Trout selected mostly larval Odonata and Diptera and aquatic Coleoptera adults while selecting against terrestrial Diptera and Trichoptera prior to the removal of Tag Alder (Figure 8). Post-treatment, selectivity was highest for larval Dipterans and Trichopterans and adult Ephemeropterans, whereas gastropods showed highest selection against.

Discussion

We found a differential response of Brook Trout feeding to riparian vegetation, including a change in Brook Trout diets following the experimental removal of riparian Tag Alder and addition of brush bundles. Brook Trout prey selectivity differed among study sites and the removal of riparian alteration may have altered prey selectivity. Stream habitat changed in response to riparian alteration and the addition of brush bundles corresponded with increasing depths and the percent of substrate as gravel.

Influence of Riparian Vegetation on Stream Characteristics

Riparian vegetation influences stream habitat characteristics in different ways. The differences in average width among the four study sites can most likely be attributed to the different types of riparian vegetation, soil types, slope, and hydrology. Riparian areas with sedge as vegetation are more likely to be narrower and deeper compared to riparian areas with woody vegetation (Lyons et al. 2000), which is what we observed in Green Meadow Creek. The lack of significant difference in pre- and post-treatment widths at the treatment site is likely the result of the limited post-treatment sampling period (3 months). The effects of riparian alteration on channel morphology can take years to observe (Hunt 1979). However, the addition of brush bundles has the ability to speed up the modification of stream channels (Karle and Densmore 1994). The differences in average depth and depth to fines that we observed among sites could also be linked to the riparian vegetation present in that area (Lyons et al. 2000 and Gran and Paola 2001). Riparian areas with woody vegetation are commonly shallower than grassy riparian areas (Lyons et al. 2000). The lack of significant difference in average depth between the treatment site and mature forest reference site is likely due to proximity to each other and similarity in substrate. Both sites are primarily made up of gravel and sand substrate. Higher percentages of gravel at the treatment site and mature forest reference site compared to the Tag Alder and sedge meadow reference sites could be the result of stream gradient (Gregory et al. 1991). Both the treatment and mature forest reference sites in Green Meadow Creek are upstream and they have steeper slopes than the downstream Tag Alder reference and sedge meadow reference sites (Table 1). The two upstream sites are also comprised of sandy loam and muck soil while the two downstream sites are primarily muck and loamy sand. Green Meadow Creek originates in the Harrison Hills area, which has a large gradient change that could cause silt and other sediment to be swept downstream. The gradient begins to level off near the sedge meadow reference site and the dominant soil in this area becomes muck, which could be the reason this site has significantly higher percentages of sand and silt. We observed more macrophytes in the sedge meadow site compared to other sites likely because of increased sunlight penetration due to the lack of canopy cover (Cooper 1993). The lower average monthly water temperatures at the sedge meadow reference site compared to all other sites indicate a groundwater upwelling in or near that stream reach.

Influence of Riparian Vegetation on Brook Trout Feeding and Abundance

The high abundance of aquatic Ephemeroptera and their drift within the water column could be why they were found in the highest frequency of Brook Trout diets throughout all the sites (Elliott 1967; Anderson and Lehmkuhl 1968). At the treatment site, Ephemeroptera nymphs, larval Trichoptera and adult Trichoptera were the most frequently occurring organisms in diets during the pre-treatment period. This could be due to increased drift of aquatic and terrestrial invertebrates in woody riparian areas (Hetrick et al. 1998). Brook Trout prefer terrestrial invertebrates during the late spring and summer months (Baxter et al. 2005), while the pre-treatment sampling period occurred in mid to late spring. The decrease in frequency of adult Trichoptera at the treatment site post-treatment could be the result of the Tag Alder removal. Wipfli (1997) found that terrestrial invertebrate abundance was highest with alder tree species and dense shrubs. This is similar to what we found in our Tag Alder study sites. Overall, the frequency of aquatic and terrestrial prey sources at the treatment site was relatively even. Prey selection of aquatic or terrestrial invertebrates could also be due to the life stages of Brook Trout. Adult Brook Trout feed on various life stages of aquatic invertebrates and terrestrial invertebrates (Williams 1981 and Li et al. 2016). The Brook Trout population at the treatment site was primarily made up of larger, adult Brook Trout. Adult Ephemeropterans and Trichopterans comprised the most mass in post-treatment diets, likely due to increased abundances of terrestrial invertebrates in the summer months (Baxter et al. 2005). Although, adult Dipterans were one of the more frequently occurring diet items they provided little mass to the overall diets likely due to their smaller size.

The LOR provided useful information regarding prey selectivity but only when all available prey are sampled. Brook Trout likely fed on Odonata nymphs due to their large size (Allan 1981). However, the variation in positive and negative selection of prey could be due to specialization of individual Brook Trout (Allan 1981). Post-treatment results showed increases in selectivity for most prey categories that could possibly be linked to the riparian alteration; however, it is too early in the post-treatment process to accurately detect the effects of the removal on Brook Trout diets. One limitation of this study was the small number of Brook Trout diets (N = 28) that were collected and analyzed from the treatment site in May through September.

Differences in relative abundances of Brook Trout among sites was likely related to differences in habitat conditions at each site. Although the sedge meadow reference site had the lowest average monthly water temperatures and highest average depth, it contained only sand, silt, and detritus as substrate. Brook Trout prefer substrate sizes of at least 7.6 cm and depths of at least 15 cm (Eifert and Wesche 1982), which we found at the mature forest reference and treatment site. We did not observe suspended sediments in the mature forest site and treatment site, which Brook Trout do not prefer (Barton et al. 1985).

We found Brook Trout to be most abundant at sites with mature woody vegetation (treatment site and mature forest reference site) and least abundant at sites with grass or shrubs only (sedge meadow reference site and Tag Alder reference site). Lyons et al. (2000) found that wooded riparian areas provided overwintering habitat for fish and reduced water temperatures, which presumably increased abundances although they did not speculate directly. Conversely, Peterson (1993) found the highest abundances of

Brook Trout in riparian areas with grassy vegetation compared to woody vegetation. However, Slawski (1997), found the highest abundances of Brook Trout in a mixture of the two riparian areas. Overall, there is differing evidence when it comes to Brook Trout abundances in relation to riparian habitat. Another reference stream would have provided additional information regarding the relationship between riparian vegetation and Brook Trout abundances.

The consistent population estimates for the treatment site could be due to a culvert at the downstream end of the reach that may reduce upstream migration of individuals into the reach. Culverts located on small streams, such as Green Meadow Creek, with slopes between 3-5% were found to be barriers for Brook Trout movement (Poplar-Jeffers et al. 2009). The culvert on Green Meadow Creek has a slope of approximately 5.3%. The larger population of Brook Trout in the mature forest reference site could be due to seasonality, suspended sediments in the treatment site, and/or disturbance from riparian alteration in the treatment site. Brook Trout spawn in the fall, which was towards the end of our sampling period, and select areas with groundwater upwelling (Johnson and Webster 1977). The mature forest reference site could potentially have groundwater upwelling that is causing increased abundances or there could be a section of the stream further upstream that Brook Trout are migrating towards to spawn. Another possible explanation for the increased abundances could be the sediments in the water column in the treatment site, which affects Brook Trout locations during summer months (Barton et al. 1985). Brook Trout in the four study sites were not differentially marked, which would have provided useful information on movement between study sites. However, movement of fish upstream from the Tag Alder reference and sedge meadow reference sites to the treatment and mature forest sites is unlikely due to the presence of a culvert at the start of the treatment site. Fish in the treatment site and mature forest reference site could have moved between the two sites.

Influence of Riparian Vegetation on Invertebrate Communities

Invertebrate abundance was related to riparian habitat. The treatment site had the highest abundances of Ephemeroptera, Plecoptera, and Trichoptera pre-treatment, which is likely due to stream flow and velocity and not substrate sizes (Pastuchova et al. 2008). Post-treatment the site had the highest abundances of Ephemeroptera, Trichoptera, and Diptera, which could be the result of riparian alteration or seasonality. Late summer and early fall provide allochthonous inputs, such as leaves, into the stream, which could increase abundances of shredders that feed on the inputs (Cummins et al. 1989). These late summer and early fall shredders are primarily Ephemeropteran, Trichopteran, and Dipteran (Cummins et al. 1989), that matches our highest abundances from Surber samples during the same time period. The sedge meadow reference site had the highest abundances of Ephemeroptera, specifically Baetidae, most likely because of the presence of aquatic macrophytes and overhanging vegetation, which provide surfaces to cling on to (Bergman and Hilsenhoff 1978). The D-net was better suited to sample this kind of habitat compared to the Surber sampler. The mature forest reference site had the highest abundances of Plecoptera primarily because of the abundance of gravel substrate at the site in which organisms can use the interstitial spaces (Flecker and Allan 1984). Trichoptera abundance was highest at the treatment site, possibly because of the variety of habitats at the site. Trichopterans utilize various habitat types to satisfy their feeding requirements (Wiggins and Mackay 1978). At the treatment site there was a mixture of riffles, runs, and pools for the diverse Trichopteran taxa to occupy. Post-treatment relative abundances of invertebrates provided similar results to pre-treatment data at the treatment site likely due to lack of time passage post Tag Alder removal. The sedge meadow reference site had the highest abundances of Ephemeroptera, Trichoptera, and Diptera due to individual specialization of these families, and this site also had the lowest diversity among all sites for terrestrial invertebrates. Commonly, the highest terrestrial invertebrate abundances occurs in closed canopies (Nakano et al. 1999). Pre- and post-treatment abundances of terrestrial invertebrates at the treatment site were similar with slight decreases in all orders, excluding Plecoptera, which saw a slight increase in abundance. However, there is only one month of pre-treatment data (May) as the June pan traps were washed away due to flooding, which makes comparisons difficult without gathering additional data.

Both watershed and riparian habitat influence invertebrate diversity. Harding et al. (1998) and Sponseller et al. (2001) found that forested watersheds and riparian habitats had higher aquatic invertebrate diversity compared to agricultural watersheds. We also found that aquatic invertebrate diversity was highest in the woody riparian habitats compared grassy riparian areas. Our diversity estimates also showed that the sedge meadow reference site had the lowest terrestrial diversity. This could be due to the lack of overhanging vegetation, which provides terrestrial invertebrates with a food source and cover (Nakano et al. 1999).

Management Implications

The experimental removal of Tag Alder in the riparian corridor and the addition of brush bundles will likely continue to alter stream channel morphology, specifically, decreasing channel width and likely deepening the stream as well. It will also likely increase the percentage of gravel substrate while decreasing the percentage of sand due to scouring. Given the short length of our study, we were not able to observe the effect of riparian alteration on Brook Trout populations. There seems to be little to no immediate effect from the alteration on Brook Trout abundances; however, there could be changes in feeding behavior, although natural shifts in diet preferences could explain these changes.

The treatment site riparian area will be managed for early successional stage vegetation through managed rotational grazing by the current landowner, who owns a rotational grazing beef farm. To ensure protection of the stream and riparian area, cattle should only be allowed to graze 1-3 days with 2-5 weeks of recovery and no more than 20 days during the growing season (Paine and Ribic 2002). Further, fencing along the riparian zone should be added to keep cattle from trampling the stream bank. Well managed rotational grazing can help protect the stream ecosystem and can help reduce the threat of invasive plant species taking over a riparian area (Paine and Ribic 2002).

The removal of riparian vegetation and addition of brush bundles has the capability of altering stream channel morphology and biological communities. The addition of brush bundles will likely decrease width while increasing depth due to scouring. An increase in gravel substrate can be expected from the scouring. The removal of riparian vegetation seems to have little short-term effect on Brook Trout populations; however, long-term effects may see changes in prey preferences between terrestrial and aquatic invertebrates. A reduction in terrestrial invertebrate abundances could be possible with the riparian vegetation removal due to habitat loss while aquatic invertebrate abundances will likely show little to no change.

Acknowledgments

Funding for this project was provided by the University of Wisconsin Stevens Point, including the Office of Student Creativity and Research. I would like to thank my committee members Dr. William Fisher, Dr. Joshua Raabe, and Dr. Kyle Herrman for their assistance with the study design and comments that improved the manuscript. I would also like to thank Tom Daigle and family members for providing access to the stream and assisting with the Tag Alder removal. Finally, I would like to thank the UWSP undergraduate and graduate student volunteers that assisted with field work throughout the project.

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Tables and Figures

Table 1. Selected habitat characteristics among study sites at Green Meadow Creek, Wisconsin from the pre-treatment period. Values in parentheses represent one standard error from the mean. Significant differences among sites are denoted with lowercase letters ($\alpha \leq 0.05$). SM= sedge meadow reference site, TA= Tag Alder reference site, TR= treatment site, and F= mature forest reference site.

Sites	Width (m)	Depth (cm)	Depth to fines	Width/depth ratio	Slope (%)	Gravel	Sand	Detritus
SM	2.36 (<0.01)a	73.73 (1.10)a	97.48 (1.58)a	3.20 (0.04)a	0.00	0.00 (<0.01)a	92.43 (0.50)a	2.33 (0.48)a
TA	4.50 (0.01)b	24.20 (0.23)b	30.07 (0.42)b	18.62 (0.20)b	0.14	42.12 (0.80)b	52.41 (0.53)b	5.47 (0.28)b
TR	4.36 (<0.01)c	19.75 (0.07)c	22.48 (0.62)c	22.06 (0.06)c	0.34	61.20 (0.46)c	35.99 (0.42)c	1.09 (0.03)a
F	4.22 (0.01)d	18.82 (0.11)c	21.46 (0.33)c	22.45 (0.15)c	0.91	82.98 (0.11)d	14.16 (0.06)d	1.25 (0.02)a
ANOVA								
F, α	26782, <0.01	1313, <0.01	4425, <0.01	1570, <0.01	-	106, <0.01	3545, <0.01	25, <0.01

Table 2. Habitat characteristics of the treatment site pre- and post-treatment at Green Meadow Creek, Wisconsin. Values in parentheses represent one standard deviation. Significant differences between pre- and post-treatment values are denoted with lowercase letters (α≤0.05).

Parameter	Pre-treatment	Post-treatment	t-test: t, p-value
Width (m)	4.36 (0.01)a	4.23 (0.06)b	3.92, 0.049
Depth (cm)	19.75 (0.15)a	20.70 (0.90)a	1.57, 0.258
Depth to fines (cm)	22.48 (0.62)a	21.40 (0.34)a	1.31, 0.320
Width/depth ratio	22.06 (0.14)a	20.46 (1.15)a	2.38, 0.166
Cobble	1.72 (0.04)a	1.66 (0.05)a	1.15, 0.368
Gravel	61.20 (1.03)a	64.18 (0.83)b	21.6, 0.002
Sand	35.99 (0.94)a	33 (0.84)b	25.36, 0.001
Detritus	1.09 (0.06)a	1.16 (0.04)a	-4, 0.058
Shading	96.54 (0.42)a	18.38 (0.86)b	106.92, <0.001

Table 3. Mean monthly water temperature (one standard deviation) for all sites May through September 2017 at Green Meadow Creek, Wisconsin. Significant differences among sites are denoted with lowercase letters (α≤0.05). SM= sedge meadow reference site, TA= Tag Alder reference site, TR= treatment site, F= mature forest reference site.

Sites	May	June	July	August	September
SM	12.56 (2.13)a	16.14 (2.29)a	16.95 (1.51)a	15.13 (1.58)a	13.76 (2.20)a
TA	12.28 (2.33)b	16.33 (2.27)b	17.25 (1.48)b	15.28 (1.59)b	13.91 (2.18)a
TR	12.27 (2.33)b	16.43 (2.28)b	17.59 (1.44)c	15.54 (1.61)c	14.15 (2.47)b
F	12.24 (2.34)b	16.39 (2.30)b	17.41 (1.70)d	15.44 (1.60)c	14.10 (2.21)b

Table 4. Monthly density (number [N]/100 m^2, 95% CI in parentheses) and trout/mile of Brook Trout in Green Meadow Creek, Wisconsin study sites. SM= sedge meadow reference site, TA= Tag Alder reference site, TR= treatment site, F= mature forest reference site.

	May		June		July		August		September	
Sites	N (CI)	Trout/ mi	N (CI)	Trout/ mi	N (CI)	Trout /mi	N (CI)	Trout /mi	N (CI)	Trout/ mi
SM	0	0	3 (0.53)	64	3 (0.53)	69	2 (0.65)	35	0	0
TA	1 (0)	37	2 (0.26)	74	2 (0.23)	55	2 (0.26)	74	1 (0)	19
TR	4 (0.35)	257	4 (0.98)	275	4 (0.46)	238	4 (0.44)	275	3 (0.59)	165
F	2 (0.34)	92	4 (0.54)	220	3 (0.17)	147	7 (0.89)	385	8 (4.18)	421

Table 5. Frequency of occurrence (percent) of Brook Trout diet contents among all sites in Green Meadow Creek, Wisconsin, 2017.

Order	SM	TA	TR	F
Ephemeroptera (A)	0.67	0.54	0.64	0.45
Ephemeroptera (T)	0.44	0.38	0.29	0.35
Trichoptera (A)	0.56	0.15	0.71	0.19
Trichoptera (T)	0.33	0.38	0.39	0.23
Odonata	0.33	0.15	0.18	0.1
Coleoptera	0.11	0	0	0.03
Lepidoptera	0.22	0.08	0.07	0.32
Diptera (A)	0.33	0.38	0.54	0.26
Diptera (T)	0.78	0.38	0.71	0.65
Hymenoptera	0	0	0	0.13
Megadrilacea	0	0	0.14	0
Trombidiformes	0	0	0.11	0
Gastropoda	0	0.46	0.11	0.23
Plecoptera	0	0	0	0.23
Anura	0	0	0.04	0

Table 6. Proportion by dry weight (grams) of Brook Trout diet contents among all sites in Green Meadow Creek, Wisconsin, 2017.

Order	SM	TA	TR	F
Ephemeroptera (A)	0.25	0.12	0.09	0.15
Ephemeroptera (T)	0.13	0.12	0.07	0.06
Trichoptera (A)	0.11	0.05	0.06	0.02
Trichoptera (T)	0.08	0.11	0.05	0.1
Odonata	0.16	0.13	0.05	0.09
Coleoptera	0.04	0	0	0.03
Lepidoptera	0.2	0.09	0.03	0.14
Diptera (A)	0.01	0.03	0.02	0.04
Diptera (T)	0.02	0.04	0.05	0.06
Hymenoptera	0	0	0	0.03
Megadrilacea	0	0	0.1	0
Trombidiformes	0	0	0	0
Gastropoda	0	0.31	0.08	0.17
Plecoptera	0	0	0	0.11
Anura	0	0	0.4	0

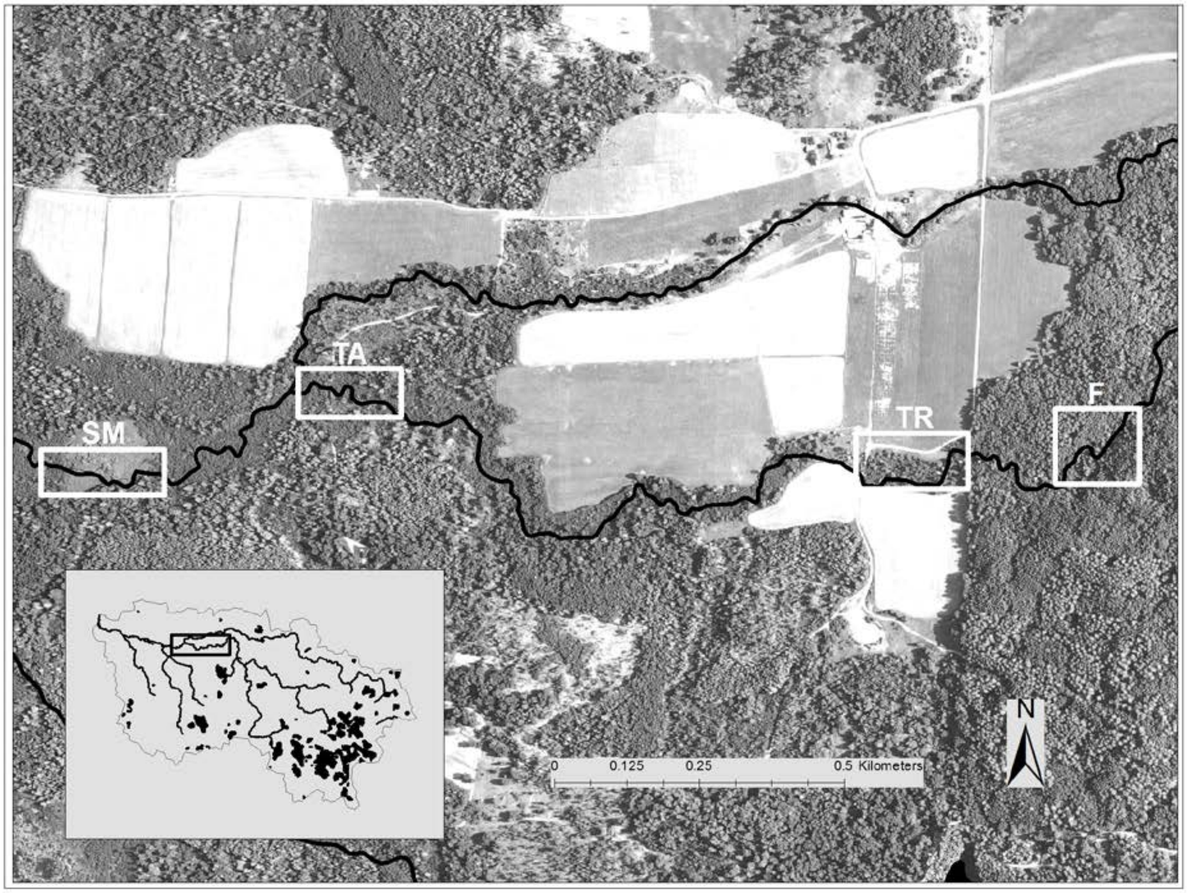

Figure 1. Green Meadow Creek study sites and watershed in Lincoln County, Wisconsin. SM=sedge meadow reference site, TA=Tag Alder reference site, TR=treatment site, and F=mature forest reference site.

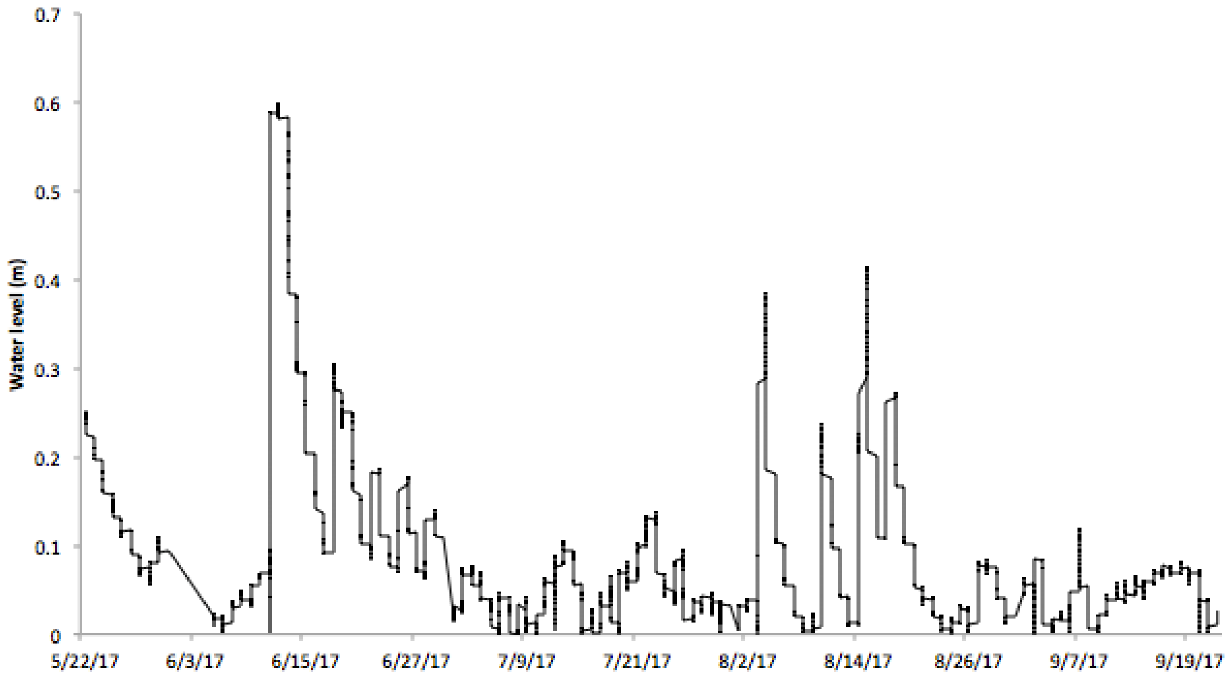

Figure 2. Water level of Green Meadow Creek, Wisconsin from 22 May 2017 through 22 September 2017. The 0.00 represents the base mark of 6.67 from the staff gage.

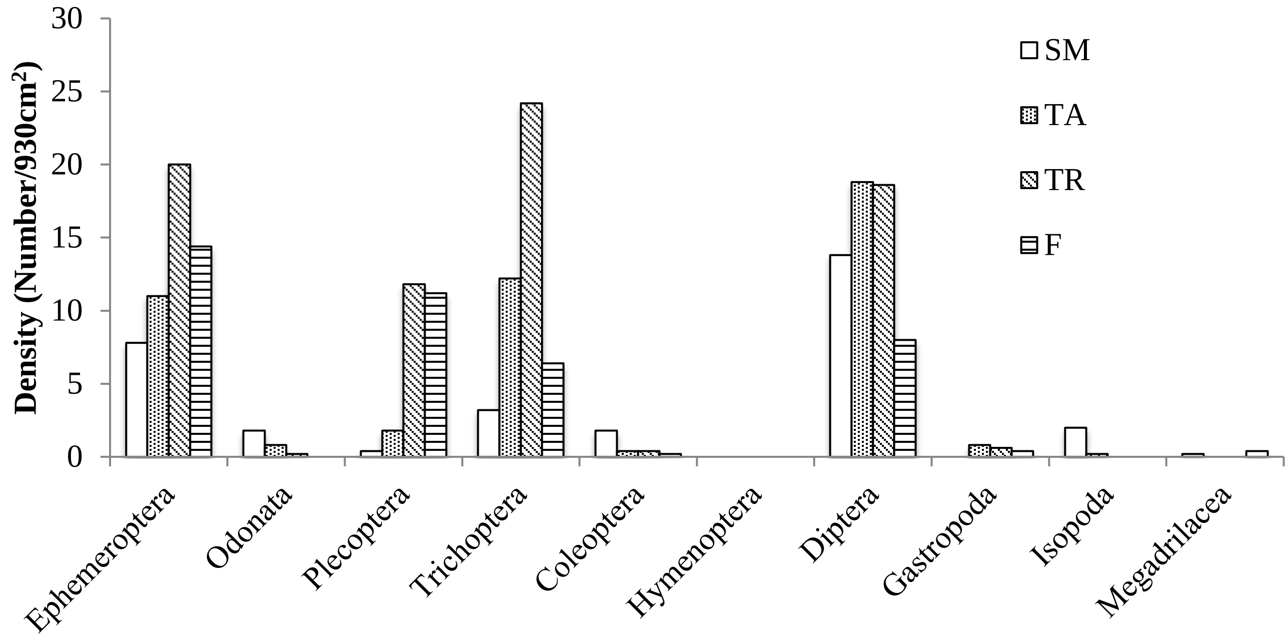

Figure 3. Density (number/930cm^2) of aquatic invertebrate orders from Surber samples among all study sites in Green Meadow Creek, Wisconsin.

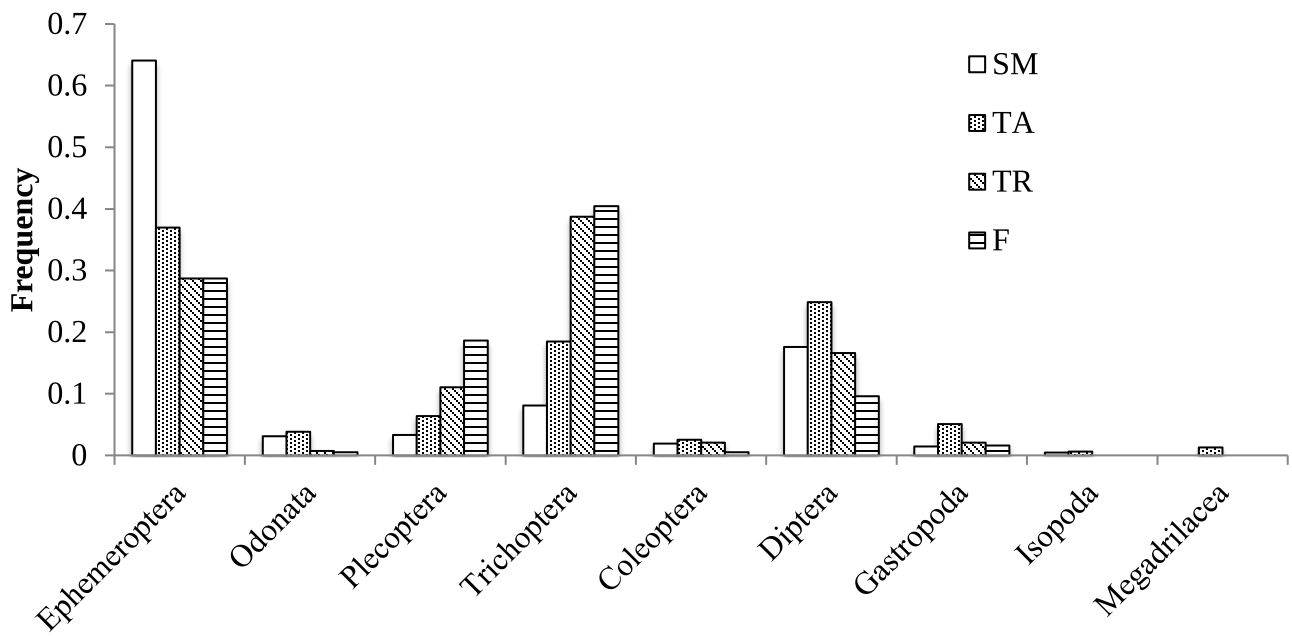

Figure 4. Relative abundance (number) from D-net samples of aquatic invertebrates among all study sites in Green Meadow Creek, Wisconsin.

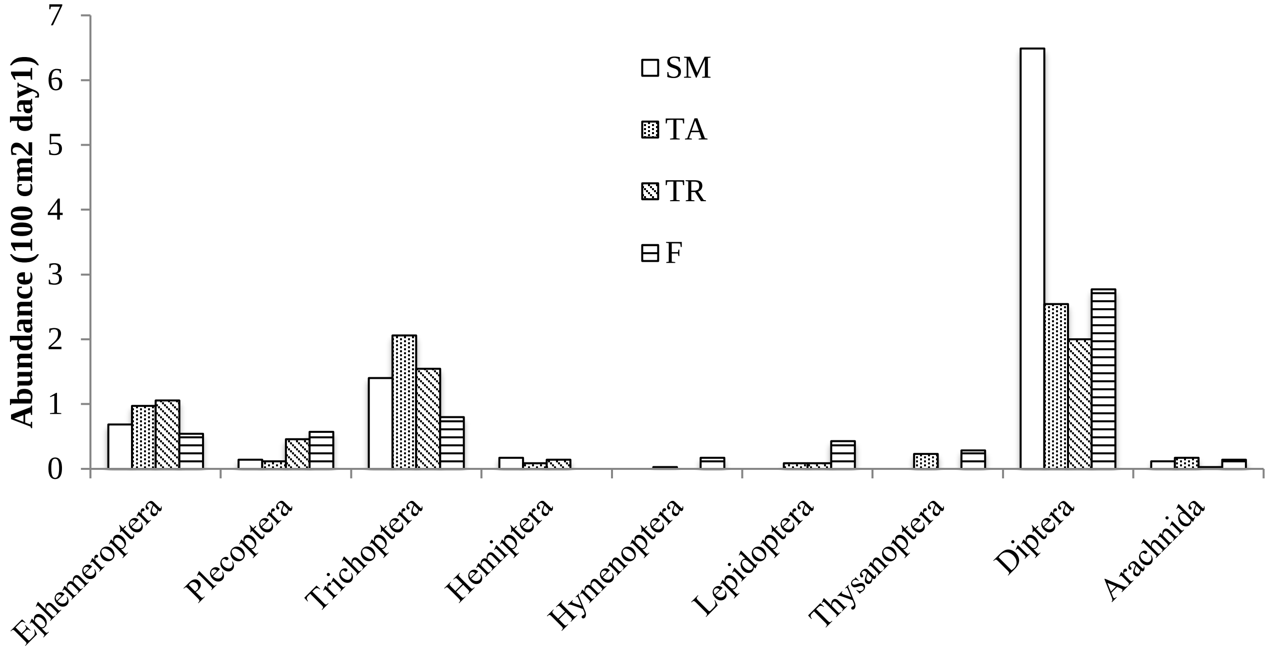

Figure 5. Abundance (100 cm^2 day^1) of terrestrial invertebrates among all study sites in Green Meadow Creek, Wisconsin.

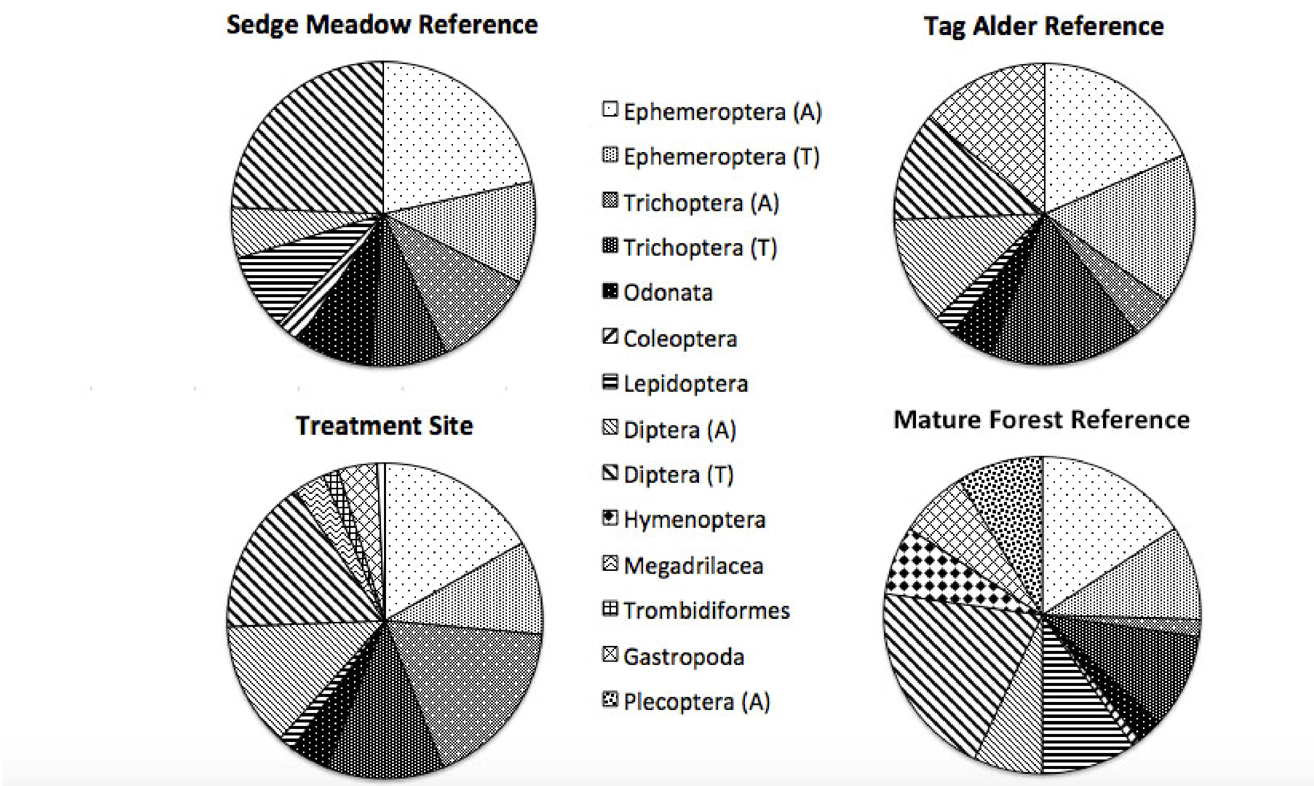

Figure 6. Brook Trout diet contents for each study site in Green Meadow Creek, Wisconsin, 2017.

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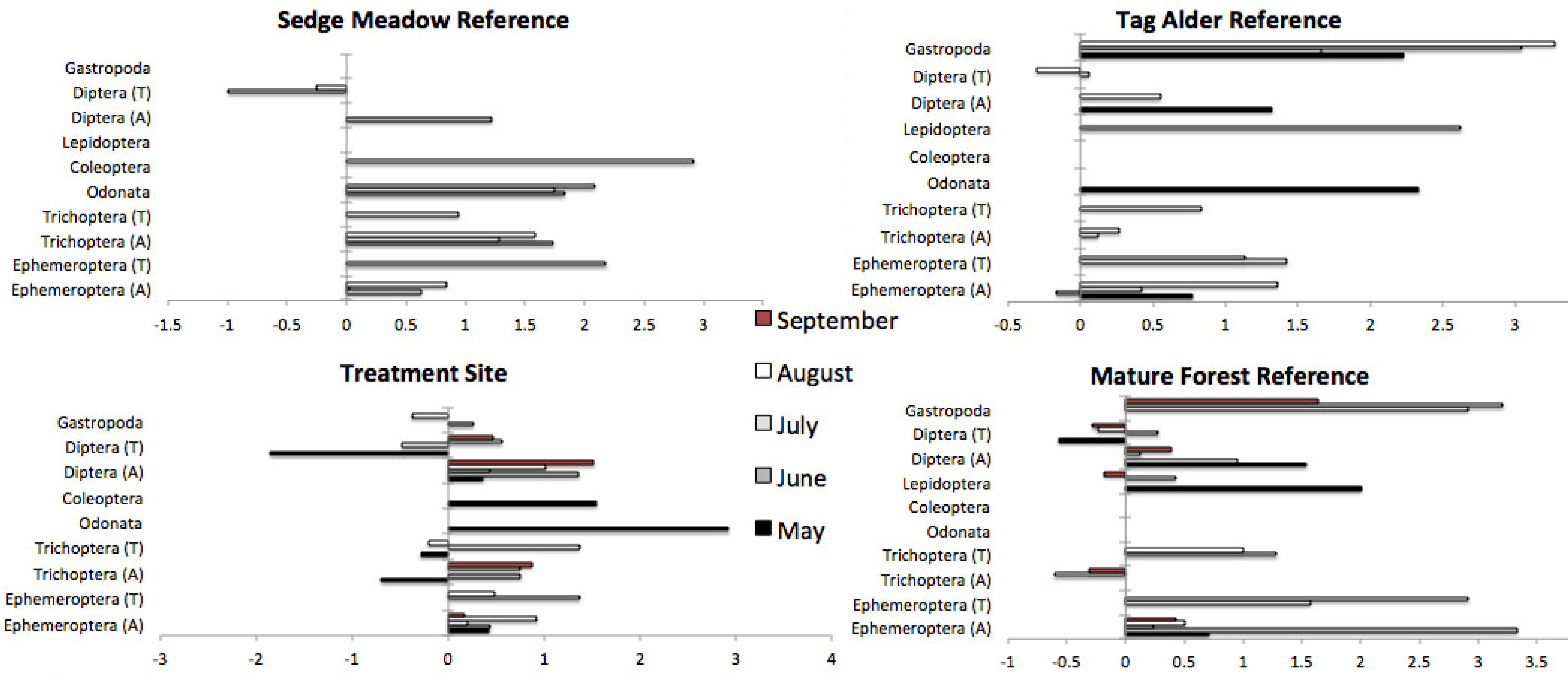

Figure 7. Monthly Brook Trout prey selectivity for each study site in Green Meadow Creek, Wisconsin, 2017. A=aquatic and T=terrestrial

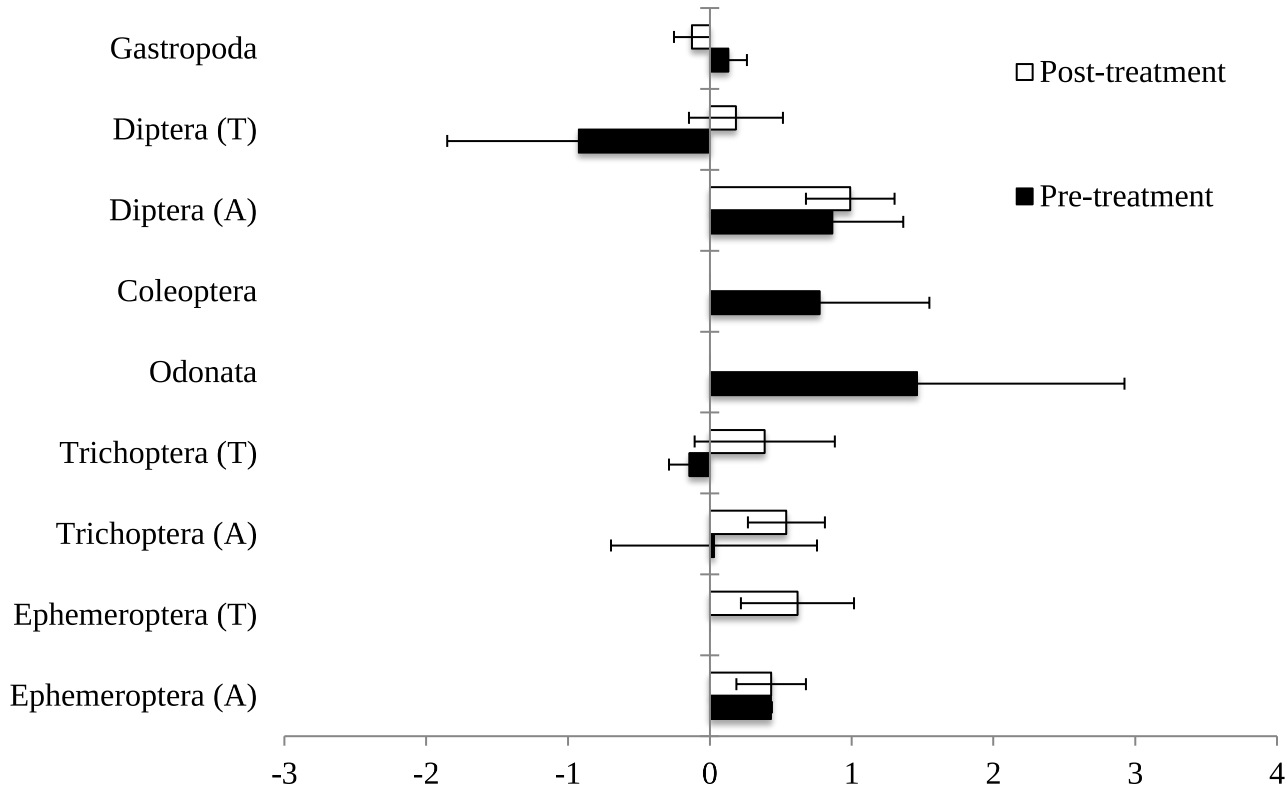

Figure 8. Brook Trout prey selectivity at Green Meadow Creek, Wisconsin, 2017 treatment site using the Log of Odds Ratio +1 SE. Letters in parentheses indicate aquatic (A) and terrestrial (T). Some diet items are not depicted due to lack of presence during invertebrate sampling.

Appendix 1. Fish assemblage in Green Meadow Creek Lincoln County, Wisconsin.

Table 1. Fish assemblage in Green Meadow Creek, Wisconsin, 2017.

Species	**Scientific name**	**Count**
Brook Trout	*Salvelinus fontinalis*	106
Mottled Sculpin	*Cottus bairdii*	80
Creek Chub	*Semotilus atromaculatus*	18
White Sucker	*Catostomus commersonii*	14
Bluegill	*Lepomis macrochirus*	13
Central Mudminnow	*Umbra limi*	11
Northern Redbelly Dace	*Chrosomus eos*	8
Largemouth Bass	*Micropterus salmoides*	8
Yellow Perch	*Perca flavescens*	8
Grass Pickerel	*Esox americanus*	3
Black Bullhead	*Ameiurus melas*	2
Pumpkinseed	*Lepomis gibbosus*	2
Total		**273**

Appendix 2. Monthly feeding of Brook Trout in Green Meadow Creek, Wisconsin, 2017.

Table 1. Monthly Brook Trout diet contents for Green Meadow Creek, Wisconsin, 2017. Note: Eph. (A)= aquatic Ephemeroptera, Eph. (T)= terrestrial Ephemeroptera, Trich. (A)= aquatic Trichoptera, Trich. (T)= terrestrial Trichoptera, Odon.=Odonata, Col.=Coleoptera, Lep.= Lepidoptera, Dip. (A)= aquatic Diptera, Dip. (T)= terrestrial Diptera, Hym.= Hymenoptera, and Plec.=Plecoptera.

Sites	Month	Eph. (A)	Eph. (T)	Trich. (A)	Trich. (T)	Odon.	Col.	Lep.	Dip. (A)	Dip. (T)	Hym.	Megad-rilacea	Gastropod	Plec. (A)	Anura
SM	May	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	June	3	1	2	0	1	0	1	2	3	0	0	0	0	0
	July	2	3	1	3	1	0	2	0	2	0	0	0	0	0
	Aug	3	0	1	0	1	1	0	0	4	0	0	0	0	0
	Sep	0	0	0	0	0	0	0	0	0	0	0	0	0	0
TA	May	1	0	0	0	1	0	0	0	2	0	0	1	0	0
	June	1	2	0	2	1	0	0	0	2	0	0	0	0	0
	July	2	4	1	5	0	0	1	0	2	0	0	3	0	0
	Aug	4	1	1	2	0	0	0	3	1	0	0	1	0	0
	Sep	0	0	0	0	0	0	0	0	0	0	0	0	0	0
TR	May	5	0	3	1	1	0	0	3	1	0	0	0	0	0
	June	4	8	4	9	0	0	2	4	6	0	0	1	0	0
	July	1	2	0	3	1	0	0	1	1	0	0	0	0	0
	Aug	9	2	9	2	2	0	0	7	10	0	2	2	0	0
	Sep	2	0	6	0	1	0	0	1	2	0	2	0	0	1
F	May	4	0	0	0	0	0	1	3	2	0	0	0	3	0
	June	2	2	0	2	0	0	3	1	4	1	0	1	4	0
	July	3	1	0	3	1	0	5	0	8	1	0	1	1	0
	Aug	6	8	1	6	1	1	1	3	8	4	0	4	2	0
	Sep	3	0	1	0	1	0	1	1	1	2	0	2	0	0

Appendix 3. Terrestrial and aquatic invertebrate diversity among sites.

Table 1. Monthly diversity of aquatic invertebrates among sites in Green Meadow Creek, Wisconsin, 2017, from the Simpson Index of Diversity.

Location	May	June	July	August	September
SM	0.619	0.609	0.623	0.708	0.769
TA	0.822	0.678	0.689	0.761	0.830
TR	0.907	0.908	0.922	0.921	0.916
F	0.859	0.859	0.889	0.897	0.873

Table 2. Monthly diversity of terrestrial invertebrates among sites in Green Meadow Creek, Wisconsin, 2017, from the Simpson Index of Diversity. Note: Dashes indicate missing data due to flooding.

Location	May	June	July	August	September
SM	0.675	-	0.738	0.753	0.803
TA	0.881	-	0.886	0.895	0.853
TR	0.909	-	0.875	0.922	0.865
F	0.903	-	0.904	0.853	0.794

Appendix 4. Aquatic and terrestrial invertebrate abundances among sites.

Table 1. Monthly abundance (930 cm^2) of aquatic invertebrates from Surber samples at the sedge meadow reference site in Green Meadow Creek, Wisconsin, 2017.

Order	May	June	July	August	September
Ephemeroptera	0	13	12	9	5
Odonata	1	3	1	3	1
Plecoptera	0	1	0	0	1
Trichoptera	0	6	2	4	4
Coleoptera	0	3	1	2	3
Hymenoptera	0	0	0	0	0
Diptera	13	9	18	14	15
Gastropoda	0	0	0	0	0
Isopoda	1	0	0	5	4
Megadrilacea	0	0	0	0	1

Table 2. Monthly abundance (930 cm^2) of aquatic invertebrates from Surber samples at the Tag Alder reference site in Green Meadow Creek, Wisconsin, 2017.

Order	May	June	July	August	September
Ephemeroptera	5	14	15	12	9
Odonata	2	0	1	0	1
Plecoptera	0	4	3	2	0
Trichoptera	18	7	15	14	7
Coleoptera	0	2	0	0	0
Hymenoptera	0	0	0	0	0
Diptera	9	21	22	26	16
Gastropoda	1	1	1	0	1
Isopoda	0	0	0	1	0
Megadrilacea	0	0	0	0	0

Table 3. Monthly abundance (930 cm^2) of aquatic invertebrates from Surber samples at the treatment site in Green Meadow Creek, Wisconsin, 2017. Tag Alder was removed during the month of June.

Order	May	June	July	August	September
Ephemeroptera	19	21	27	20	13
Odonata	1	0	0	0	0
Plecoptera	13	9	15	12	10
Trichoptera	37	20	25	24	15
Coleoptera	0	0	1	0	1
Hymenoptera	0	0	0	0	0
Diptera	23	14	21	18	17
Gastropoda	0	0	0	1	2
Isopoda	0	0	0	0	0
Megadrilacea	0	0	0	0	0

Table 4. Monthly abundance (930 cm^2) aquatic invertebrates from Surber samples at the mature forest reference site in Green Meadow Creek, Wisconsin, 2017.

Order	May	June	July	August	September
Ephemeroptera	17	17	13	16	9
Odonata	0	0	0	0	0
Plecoptera	18	18	2	9	9
Trichoptera	5	8	7	7	5
Coleoptera	0	0	1	0	0
Hymenoptera	0	0	0	0	0
Diptera	3	5	14	11	7
Gastropoda	1	0	0	0	1
Isopoda	0	0	0	0	0
Megadrilacea	0	2	0	0	0

Table 5. Relative abundance (count) of aquatic invertebrates from D-net samples at the sedge meadow reference site in Green Meadow Creek, Wisconsin, 2017.

Order	May	June	July	August	September
Ephemeroptera	41	66	54	57	51
Odonata	1	2	5	4	1
Plecoptera	1	2	3	4	4
Trichoptera	4	6	7	9	8
Coleoptera	2	3	1	0	2
Hymenoptera	0	0	0	0	0
Diptera	14	13	16	14	17
Gastropoda	0	2	1	1	2
Isopoda	0	0	0	2	0
Megadrilacea	0	0	0	0	0

Table 6. Relative abundance (count) of aquatic invertebrates from D-net samples at the Tag Alder reference site in Green Meadow Creek, Wisconsin, 2017.

Order	May	June	July	August	September
Ephemeroptera	8	14	13	12	11
Odonata	1	0	1	3	1
Plecoptera	2	3	1	1	3
Trichoptera	3	1	6	10	9
Coleoptera	1	1	0	1	1
Hymenoptera	0	0	0	0	0
Diptera	7	5	10	12	5
Gastropoda	2	3	1	1	1
Isopoda	1	0	0	0	0
Megadrilacea	0	1	1	0	0

Table 7. Relative abundance (count) of aquatic invertebrates from D-net samples at the treatment site in Green Meadow Creek, Wisconsin, 2017. Tag Alder removal occurred during the month of June.

Order	May	June	July	August	September
Ephemeroptera	15	21	16	13	18
Odonata	0	2	0	0	0
Plecoptera	2	4	7	11	8
Trichoptera	19	26	29	22	16
Coleoptera	2	1	0	2	1
Hymenoptera	0	0	0	0	0
Diptera	8	11	13	7	9
Gastropoda	1	1	2	1	1
Isopoda	0	0	0	0	0
Megadrilacea	0	0	0	0	0

Table 8. Relative abundance (count) of aquatic invertebrates from D-net samples at the mature forest reference site in Green Meadow Creek, Wisconsin, 2017.

Order	May	June	July	August	September
Ephemeroptera	6	9	13	11	15
Odonata	0	1	0	0	0
Plecoptera	2	3	9	11	10
Trichoptera	16	21	18	9	12
Coleoptera	1	0	0	0	0
Hymenoptera	0	0	0	0	0
Diptera	3	1	5	5	4
Gastropoda	1	0	1	1	0
Isopoda	0	0	0	0	0
Megadrilacea	0	0	0	0	0

Table 9. Abundance (100 cm^{-2} day^{-1}) of terrestrial invertebrates from pan traps at the sedge meadow reference site in Green Meadow Creek, Wisconsin, 2017.

Order	May	June	July	August	September
Ephemeroptera	9	0	7	12	11
Plecoptera	0	0	2	1	2
Trichoptera	25	0	16	18	13
Hemiptera	1	0	1	2	2
Hymenoptera	0	0	0	0	0
Lepidoptera	0	0	0	0	0
Thysanoptera	0	0	0	0	0
Diptera	73	0	55	61	38
Arachnida	2	0	1	1	0

Table 10. Abundance (100 cm^{-2} day^{-1}) of terrestrial invertebrates from pan traps at the Tag Alder reference site in Green Meadow Creek, Wisconsin, 2017.

Order	May	June	July	August	September
Ephemeroptera	1	0	10	5	8
Plecoptera	0	0	1	2	1
Trichoptera	7	0	12	15	15
Hemiptera	1	0	1	0	1
Hymenoptera	0	0	0	1	0
Lepidoptera	1	0	1	0	1
Thysanoptera	4	0	2	1	1
Diptera	20	0	26	21	22
Arachnida	1	0	2	2	1

Table 11. Abundance (100 cm^{-2} day^{-1}) of terrestrial invertebrates from pan traps at the treatment site in Green Meadow Creek, Wisconsin, 2017. Tag Alder was removed during the month of June.

Order	May	June	July	August	September
Ephemeroptera	4	0	7	6	10
Plecoptera	3	0	3	5	5
Trichoptera	3	0	7	12	2
Hemiptera	1	0	1	2	1
Hymenoptera	0	0	0	0	0
Lepidoptera	0	0	0	0	3
Thysanoptera	0	0	0	0	0
Diptera	19	0	22	14	15
Arachnida	0	0	1	0	0

Table 12. Abundance (100 cm^{-2} day^{-1}) of terrestrial invertebrates from pan traps at the mature forest reference site in Green Meadow Creek, Wisconsin, 2017.

Order	May	June	July	August	September
Ephemeroptera	0	0	2	2	5
Plecoptera	1	0	4	4	11
Trichoptera	2	0	8	9	9
Hemiptera	0	0	0	0	0
Hymenoptera	1	0	2	2	1
Lepidoptera	1	0	0	4	10
Thysanoptera	5	0	2	1	2
Diptera	26	0	29	31	11
Arachnida	1	0	1	2	1

Appendix 5. Water quality comparisons among all study sites.

Table 1. Average of water quality parameters from the Hydrolab MS5 multiparameter mini sonde among all study sites in Green Meadow Creek, Wisconsin, 2017. Values in parentheses indicate one standard deviation. Lowercase letters indicate significant differences among sites (α=0.05).

Site	DO%	DO mg/L	SpC	pH
SM	93.8 (1.60)a	9.45 (0.10)a	130.12 (7.12)a	7.20 (0.29)a
TA	90.9 (0.34)ab	9.13 (0.11)ab	141.02 (13.61)a	7.24 (0.25)a
TR	95.44 (2.17)ac	9.56 (0.23)ac	135.3 (8.58)a	7.41 (0.14)a
F	96.84 (1.90)c	9.76 (0.24)a	133.96 (7.92)a	7.52 (0.12)a