RESTORING WOODY-ENCROACHED SAVANNA AND CATTAIL-INVADED

WETLANDS USING PRESCRIBED BURNING AND GRAZING

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Restoring Woody-Encroached Savanna and Cattail-Invaded Wetlands
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The Supervisory Committee certifies that this ***disquisition*** complies with North Dakota

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ABSTRACT

Prairie-Oak Savanna is a temperate, globally-unique, endangered habitat type characterized by old-growth Bur Oak (*Quercus macrocarpa*) stands, expansive low prairie habitat, and wetland complexes. American Hazel (*Corylus americana)* and invasive hybrid cattails (*Typha x glauca)* are encroaching upon the oak savannas and marshes and are targeted by fire and grazing management however the effect of these treatments on savanna structure and secretive marsh bird and passerine populations is under-studied. We calibrate Terrestrial Laser Scanning (TLS) against conventional methods and demonstrate that TLS can estimate biomass and model the structure of savanna but not cattail-invaded wetlands. Species-level density estimates and community-wide ordination suggests little evidence of treatment effect on marsh birds and passerines although mean biomass and vegetation height were significant for both communities. Refuge-wide population estimates for Marsh Wren (*Cistothorus palustric*), Sedge Wren (*Cistothorus platensis*), Virginia Rail (*Rallus limicola*), and Sora (*Porzana carolina*) range from 0.02 to 0.37 birds/ha.

ACKNOWLEDGEMENTS

DEDICATION

To my mom, the woman who inspired my love of nature and who, in all the ways that matter,

was with me for every step of this journey.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1: SECRETIVE MARSH BIRD, PASSERINE RESPONSE TO PRESCRIBED BURNING, GRAZING, HERBICIDAL TREATMENTS IN CATTAIL-INVADED WETLAND

Abstract

As an invasive, emergent aquatic plant, hybrid cattail (*Typha x glauca*) often forms monodominant stands, which alter wetland vegetation structure and can negatively affect bird populations and wetland use. To reduce cattail density, wetland managers use burning, grazing, and herbicidal treatments but the relationship between these treatments and marsh birds need to be studied more. Our study objectives were to investigate the effect of hybrid cattail on secretive marsh bird and passerine abundance and examine how environmental variables (vegetative biomass/height, percent cattail/water) and burning, grazing, and herbicide treatments to control invasive cattail influence bird density, local bird populations, and bird community composition. We surveyed four passerine species and three secretive marsh bird species: Yellow-headed Blackbird (*Xanthocephalus xanthocephalus*), Marsh Wren (*Cistothorus palustric*), Sedge Wren (*Cistothorus platensis*), Swamp Sparrow (*Melospiza georgiana*), Virginia Rail (*Rallus limicola*), Sora (*Porzana carolina*), and Pied-billed Grebe (*Podilymbus podiceps*) in sixteen wetlands across eight treatments with full-factorial design. Although species-level density estimates and community-wide ordination suggests little evidence of substantial effect of treatments on marsh birds and passerines, vegetation height and mean biomass were significant (P = 0.02 and P<0.01, respectively) for both passerine and marsh bird community dynamic. This suggests that changes in bird-community response might be seen if treatments eventually affect these environmental variables. We extrapolated sampled densities to estimate refuge-wide populations for four species (Marsh Wren, Sedge Wren, Virginia Rail, Sora), which ranges from 0.02 to 0.37

birds/ha. The effects of invasive cattails are not immediately reversed after treatment, which might explain the lack of relationship between vegetation structure, treatments, and bird during our two-year study; a long-term study might better elucidate relationships between cattail treatments and bird populations. Secretive marsh birds are under-surveyed, which makes baseline population estimates valuable for their future local and regional management and for understanding the effect of invasive species management on bird-communities.

Introduction

Wetlands are important because they support water quality protection, flood regulation, cultural benefits, and wildlife habitat including stop-over during migration, yet wetland abundance is in decline due to land use and climate change (Mitsch and Hernandez, 2012). Invasive species, drainage for agriculture and development, urbanization, pollution, and fragmentation threaten the quality and availability of wetland habitat across North America (Mitsch and Hernandez, 2012). Management and restoration of wetlands is important because reduced habitat availability and quality subsequently reduces the population of birds that depend on wetland habitat (Kostecke 2002; Kostecke et al. 2004). Wetland vegetation structure is an important component of wetland habitat quality and is degraded when the invasive hybrid cattail (*Typha x glauca)* establishes monocultures (Zedler and Kercher, 2004).

Secretive marsh birds are cryptic waterbirds that live in dense, tall, emergent vegetation. The cryptic nature of secretive marsh birds and their association with habitat that is both difficult to access and limits detectability makes secretive marshbird species under-surveyed (Johnson et al., 2009). The relationship between secretive marsh birds and monoculture stands of invasive cattail is compelling because while secretive marsh birds are generally associated with dense, tall, emergent vegetation, limited research on these species suggests contradictory relationships:

some species like Soras (*Porzana carolina*) select invaded cattail stands specifically while species like the Virginia Rail (*Rallus limicola)* are positively associated with the amount of emergent vegetation (Johnson and Dinsmore 1986; Linz et al. 1997; Budd 2007). Evidence indicates that Pied-billed Grebes and other species of waterfowl respond favorably (higher density in treated versus untreated wetlands) to control mechanisms targeting cattail monocultures (burning, discing, and grazing) (Kostecke, 2002). For other species, specific habitat selection related to monodominant cattail stands is unknown, although dense stands are generally known to inhibit wetland use by most wildlife, so understanding relationships between cattails, wildlife and treatments meant to eradicate this invasive species is critical to future management (Kostecke, 2002). The two primary issues facing marsh birds are invasive species and habitat loss. These often occur together as invasive species can lead to habitat loss through the alteration of functional, structural and compositional characteristics of the ecosystems that they invade (Skagen and Thompson, 2013; Tuchman et al., 2009).

An invasive, emergent plant in North American wetlands, hybrid cattail *Typha x glauca* creates monodominant stands and outcompetes native aquatic vegetation through rapid self-replacement and spread via rhizomatous roots (Zedler and Kercher, 2004). Litter accumulation following the establishment of invasive cattails alters ecosystem processes and wetland plant community (Farrer and Goldberg, 2009, 2014; Larkin et al., 2011). Hybrid cattail homogenizes community composition, increases wetland vegetative density, and negatively affects aquatic invertebrate populations and distribution within a wetland (Farrer and Goldberg, 2014; Kosteche et al., 2005). These changes effectively limit waterfowl access to an important food source thereby reducing wetland use by birds who often select wetlands based upon resource availability (Kosteche et al., 2005; Kostecke, 2002). Ultimately, the variables that support

diverse communities of marsh bird and passerine populations such as food availability and heterogeneous structure are negatively impacted by cattail invasion (Johnson and Dinsmore 1986; Rogers et al. 2013).

Because cattail litter accumulation limits growth of other plant species, it is critical to disrupt positive feedbacks by first removing accumulated litter (Larkin et al., 2011; Tuchman et al., 2009). Prescribed burning, mechanical, grazing, and herbicidal treatments have been used extensively to control invasive aquatic species like hybrid cattails (Farrer and Goldberg, 2014; Linz and Homan, 2011; Mozdzer et al., 2008; Schultz, 1987; Smith and Kadlec, 1985; Solberg and Higgins, 1993). Each treatment is associated with a suite of potential benefits and tradeoffs (Table 1.1). While the effects of treatments on species diversity and invasive species have been well-studied there are no studies on secretive marsh birds and passerines following the application of fire, grazing, and herbicidal treatment combinations to reduce cattail density. Understanding this relationship could inform management decisions with the primary goal of optimizing habitat by eliminating or reducing the density and size of cattail stands. Such changes in cattail populations support the greatest diversity of marsh bird and waterfowl populations by increasing native flora diversity, creating structural and compositional heterogeneity, and supporting the integrity of nutrient cycling through the wetland complex (Farrer and Goldberg, 2014).

Table 1.1: Positive and negative effects of fire, grazing and herbicide on invasive cattail and wetland health and structure.

		Treatment Effects		
	Positive		**Negative**	
	Effect	*Citations*	*Effect*	*Citations*
Fire	•Removes litter	Kostecke et al. 2004; Payne 1992	• Benefits cattails, other invasives	Farrer and Goldberg 2014; Kostecke 2004; Thompson and Shay 1985, 1989; Laubhan 1995; Smith and Kadlec 1985b;
	•Litter removal initially interrupts positive feedback loops	Tuchman et al. 2009; Larkin et al. 2011	• Short-term negative effect with long-term benefits for cattails	
	• Temporarily increases light availability, surface water and water temperature	Smith and Kadlec 1985b; Thompson and Shay 1985	• Eventually increases shoot viability, density, biomass, and canopy • Rhizamotous network size, function not	
Grazing	• Reduces cattail litter depth and canopy		• Can reduce species richness and diversity of non-cattail species	Kostecke et al. 2004; Shultz 1987
	• Increases water temperature	Kostecke et al. 2004; Shultz 1987	• Invasion slowed but not halted	
	• Reduces live and dead cattail density and biomass		• Short-term effect, annual application likely required	
	• Can increase species diversity, not necessarily desirable natives		• Disturbed soil promotes new invasive species establishment • Can reduce native species diversity	
Herbicide	• Slow-acting but can remain active into next growing season • Effective in monodominant, high-	Michigan DEQ 2014	• Residual cattails, mat remain • Non-selective, can affect non-target species	Linz and Homan 2011; Michigan DEQ 2014; Mozdzer et al. 2008
	• Long term control (1 to 4 years) • Easy: treats large areas quickly	Solberg and Higgins 1993;	• May only be effective in early stages of invasion	Larkin et al. 2011; Tuchman et al. 2009
	• Minimal cattail regrowth after 1 year	Mozdzer et al. 2008	• Decreases non-cattail productivity	Kostecke et al. 2004

Objectives of this study were to (1) measure the vegetation response to burning, grazing, and herbicidal treatments meant to reduce cattail density and litter; (2) investigate the response of secretive marsh birds and passerines to these treatments and (3) Survey passerine and secretive marsh bird species to estimate population size across the Sherburne National Wildlife Refuge in Central Minnesota. We hypothesized that (1) wetlands with little emergent vegetation or wetlands with high cattail density would have lower species richness and abundance; (2) wetlands with monodominant stands of cattails would have few species present due to a lack of heterogeneity and resource availability and (3) treatment type would affect cattail density and growth, thereby affecting passerine and marsh bird presence and abundance.

Materials and Methods

Study Area and Site Selection

This study was conducted at Sherburne National Wildlife Refuge (SNWR) in Sherburne County, MN (45°30′N 93°44′W). The 12,500 ha refuge lies in a prairie-oak savanna transitional zone and contains 5,700 ha of wetland managed for migratory waterfowl along the Mississippi flyway. Approximately ten percent of the wetlands on SNWR are primarily or entirely invaded by hybrid cattails, *Typha x glauca* (USFWS 2005). In 2014, SNWR launched a management program for the reduction of *Typha x glauca* that includes airborne herbicidal applications of Imazapyr, prescribed fire, and cattle grazing. Seven treatment combinations were investigated in this study and compared to untreated wetlands:

- Untreated

- Herbicide

- Grazing

- Burning

- Herbicide + Grazing

- Herbicide + Burning

- Burning + Grazing

- Herbicide + Grazing + Burning

Two wetlands were surveyed in each treatment with the exception of Burning + Herbicide and Burning as there was only one wetland for each of them on SNWR where these treatments were applied.

Survey points were 50m from wetland edges and at least 200-m apart to reduce the potential of double-sampling birds during the call-broadcast and point-count distance sampling (Conway and Gibbs, 2011). Wetlands were identified based upon refuge-established treatment plans and all sampled wetlands were invaded by hybrid cattail .

Bird Surveys

We used the North American Marsh Bird Monitoring Protocol (NAMBMP) as a protocol for our sampling design (Conway, 2011), which consisted of a 5-minute passive point-count survey to detect passerines followed by a 6-minute call-broadcast survey to detect secretive marsh birds. To facilitate density estimation via distance sampling, we recorded the distance from the observer for each individual (Thomas et al., 2010). Broadcast sequence of bird calls was provided by the NAMBMP. The sequence of bird calls are ordered by species dominance to avoid scaring away less dominant birds before their respective call, thereby reducing the possibility that the call-broadcast sequence would affect the movement of birds to or away from the point (Conway, 2011). All visual and aural detections at each point were recorded along with the distance and direction from the survey point for each individual.

Surveyed bird species were selected prior to field work and include wetland-obligate species whose distributions overlapped our field site (four passerine and six secretive marsh birds) known to rely on wetlands to reproduce, although the allowable percent of cattail invasion varies among species or, in some cases, is unknown (Linz et al. 1996; Panci 2013). Passerines included Sedge Wren *(Cistothorus platensis)*, Marsh Wren (*Cistothorus palustric)*, Swamp Sparrow (*Melospiza georgiana),* and Yellow-headed Blackbird (*Xanthocephalus xanthocephalus)*. Secretive marsh birds include American Bittern *(Botaurus lentiginosus)*, Least Bittern *(Ixobrychus exilis)*, Virginia Rail (*Rallus limicola)*, Sora (*Porzana carolina*), Pied-billed Grebe (*Podilymbus podiceps*), and American Coot *(Fulica americana).*

We conducted bird surveys from June 8 to July 4 in 2015 and May 25 to June 30 in 2016, completing two and three survey rounds, respectively. Surveys were conducted 30-minutes prior to sunrise to 3 hours after sunrise using an MP3 player connected to a 90-dB loud external speaker. The speaker was positioned to face the center of the wetland and was 0.5m from the ground or water surface. Each individual was recorded only once at their first location even if they called again at a different distance as per the protocol of distance sampling (Buckland, 2001). Additionally, care was taken to avoid double counting of individual birds. Surveys were only conducted when wind speed was less than 12 km/hr. and there was no heavy rain. Wind speed, temperature, and time were recorded using weather data from the nearest weather station, and background noise was estimated using aural estimation guided by scales described in the Standardized North American Marsh Bird Monitoring Protocol (Conway, 2011).

Vegetation Sampling

From June 15 to July 4 in 2015 and June 10 to June 30 in 2016 we sampled four, $0.5m^2$ quadrats in each of the wetlands we surveyed. One quadrat was sampled in 2015 and three were

sampled in 2016. These quadrats were placed on the edge of the wetland nearest to each of the

bird survey points so that vegetation sampled was most representative of the vegetation seen at

the bird survey locations. Quadrats in 2016 were placed 0.5 m apart and all vegetation within

these quadrats were clipped at water level and bagged; harvested biomass was oven-dried at 60°

C for 48 hr and massed. Quadrats were associated with bird survey points described above; that

took place 50 m from the edge of the wetland. The edge of the wetland was classified as where

the water started or, in marshes that were dry or drawn down, where the cattails began. To

complement our vegetation sampling at the edge of wetlands, we non-destructively estimated

local environmental variables during each bird point count 50 meters from the wetland edge

including maximum vegetation height, percent water/cattail/non-cattail, water depth at point, and

overall water depth. These variables were measured using foldable rulers and ocular estimation.

Analysis

Comparison of Vegetation across Treatments

To determine how vegetation biomass varied across each of the treatment combinations,

we tested for a relationship between biomass and treatment type with linear mixed-effect

regression (LME) models using the lmer function in the lme4 package in the R statistical

environment (Bates et al., 2015; RStudio Team, 2015), with wetland and point as random effects.

We calculated the R^2 value for our LME model, conducted a pairwise comparison of all

treatments using the glht function with a Tukey-test in the multcomp package in R (Hothorn et

al., 2008).

To identify the effect of each cattail control treatment, in isolation and in combination,

we used multiple LME regressions to compare vegetative biomass against each treatment type—

grazing, burning, and herbicide—coded as binary dummy variables, with wetland and point as

random effects. We calculated corrected Akaike's Information Criterion (AICc) which is adjusted for small sample sizes for each model and conducted an AICc-based model comparison to rank our models using the AICc and aictab functions, respectively, in the AICcmodavg package in R (Mazerolle, 2016). Models were selected by lowest AICc value and greatest Akaike weight. We then calculated the R^2 values for our best and null LME regressions models and tested for model significance using the anova function in R (Bates et al. 2015). Next, we conducted a Type III Sums of Squares ANOVA to test the significance of our fixed-effect and interaction parameters using the anova and summary functions in the lme4 and lmerTest packages in R (Bates et al., 2015; Kuznetsova et al., 2016). We calculated 95% confidence intervals for all parameters to model the relative effect of each parameter using the confint function. Finally, we tested the effect of significant treatment parameters on biomass using the effect function in the effects package in R (Fox et al., 2016).

Bird Abundance and Community Composition

We examined how environmental variables influenced bird communities across different treatments using non-metric multi-dimensional scaling with a Bray-Curtis dissimilarity matrix via the metaMDS function in the vegan package for the R statistical environment (Oksanen et al., 2016; RStudio Team, 2015). We then tested for treatment effect by comparing our environmental variables—standing vegetative biomass, percent water, vegetation height, percent cattail, and percent non-cattails—against our ordination result with the vegan function envfit, stratified by wetlands.

We tested for a relationship between vegetation biomass and individual bird species abundance with generalized linear mixed-effect regression models (GLMM) using the glmer function in the lme4 package in R (Bates et al., 2015). Response variables in GLMMs included

all detected individuals per species, tested against clipped biomass as a fixed effect and site fitted as a random effect.

Species-level Abundance and Density

We estimated population density for four species with sufficient detections (n > 40): Marsh Wren, Sedge Wren, Virginia Rail, and Sora using Program Distance 6.2 (Thomas et al., 2010). We modeled our detection function using multiple covariate distance sampling using half-normal and hazard- rate functions with cosine expansion and estimated distances for each species were truncated and organized into distance bins to meet assumptions of distance sampling and minimize variation in measurements (Buckland, 2001). Akaike's Information Criterion (AIC) was used to rank models and the best-supported model was chosen based upon the smallest AIC value. This model was then used to obtain density estimates and 95% confidence intervals for each species. These density estimates were used to estimate population sizes for the entire refuge based upon total area of cattail coverage across Sherburne using refuge coverage data and ArcGIS10.3 (ESRI 2015). We extrapolated these data across all wetland area evenly as we did not observe differences among treatments in the community analysis or ordination to suggest wetland bird densities differ among wetlands by treatment status.

Results and Discussion

Comparison of Vegetation across Treatments

While our results indicate a significant overall association between vegetative biomass and treatment type (P < 0.01, R^2 = 0.31), only four treatment types showed statistically significant differences in pairwise comparisons. On average, the Herbicide + Grazing treatment had the greatest amount of biomass, which stands to reason as, unlike burning, neither of these treatments immediately remove vegetation and litter. Herbicide + Grazing was significantly

different from Herbicide + Burning (P = 0.03, 95% CI = 2.02–109.71), Grazing + Burning (P < 0.01, 95% CI = 9.70–121.18), Herbicide + Grazing + Burning (P < 0.01, 95% CI = 12.74–120.23), and Grazing (P = 0.03, 95% CI = 2.02–107.70); no other treatments differed significantly from the untreated wetlands. We suspect that the three treatments were statistically different from Herbicide + Grazing because they included a burning treatment, which causes an immediate and substantial decrease in vegetation so at these plots only new growth was present and dead standing litter was not adding to the total biomass.

Although all treatments individually tended to reduce biomass, only Grazing (P = 0.04, 95% CI = 77.81–1.16) and Grazing * Herbicide interaction (P < 0.01, 95% CI = 32.09–20.36) are significant. These results suggest that grazing alone is effective but herbicide treatment dampens the effect of grazing on vegetative biomass, which could result from either the timing of the herbicide and grazing applications or cattle movement related to available forage around the wetlands.

While herbicide kills live cattail plants and stops most new growth it does not actively reduce the amount of vegetation biomass through removal (AgroSciences 2002; Linz and Homan 2011). Instead, dead cattail plants will remain standing until environmental influences such as snow or decomposition flatten them, adding to the cattail litter mat, or another treatment that removes dead litter follows the application or herbicide (Linz and Homan, 2011). Imazapyr was applied one year prior to the introduction of cattle in treated wetlands so the effect of the herbicide left expanses of dead cattails behind.

Our sampled quadrats were on the edge of wetlands and the density of live cattails was likely reduced on the wetland perimeter because of the cumulative effect of cattle trampling vegetation while travelling to water and then trampling additional vegetation as they foraged

near the wetland perimeter for live cattails and other plants. Comparatively, wetlands treated with herbicide had little or no live vegetation present so while cattle trampled vegetative biomass while pursuing water they most likely did not linger on the wetland perimeter, opting to leave the area to find live vegetation for grazing. Grazing alone has a cumulative effect on vegetation near the wetland perimeter that causes it to have a greater effect than other treatments while herbicide dampens this effect by reducing the amount of vegetation available to graze and encouraging the cattle to forage elsewhere.

Community Analysis

Ordination provided no evidence of a relationship between bird community composition and treatment, although maximum vegetation height and vegetation biomass were significantly correlated with variation in bird community composition ($P=0.02$ and $P<0.01$, respectively) (Figure 1.1). The Sora and Virginia Rail use similar habitat types, often occurring together, and are typically associated with dense, emergent vegetation (Budd, 2007); however in our study, they showed distinct differences in selection for environmental variables. Soras were closely associated with vegetation height compared to other species like the Yellow-headed Blackbird or Sedge Wren while Virginia Rails selected wetlands with greater vegetative biomass. This suggests Soras select for invaded cattail stands (as they were correlated with vegetation height and cattails grow taller and more vigorously in their own litter) while Virginia Rails select for adequate cover and are not actively seeking out or avoiding cattail marshes (Figure 1.1) (Johnson and Dinsmore 1986; Budd 2007; Farrer and Goldberg 2009).
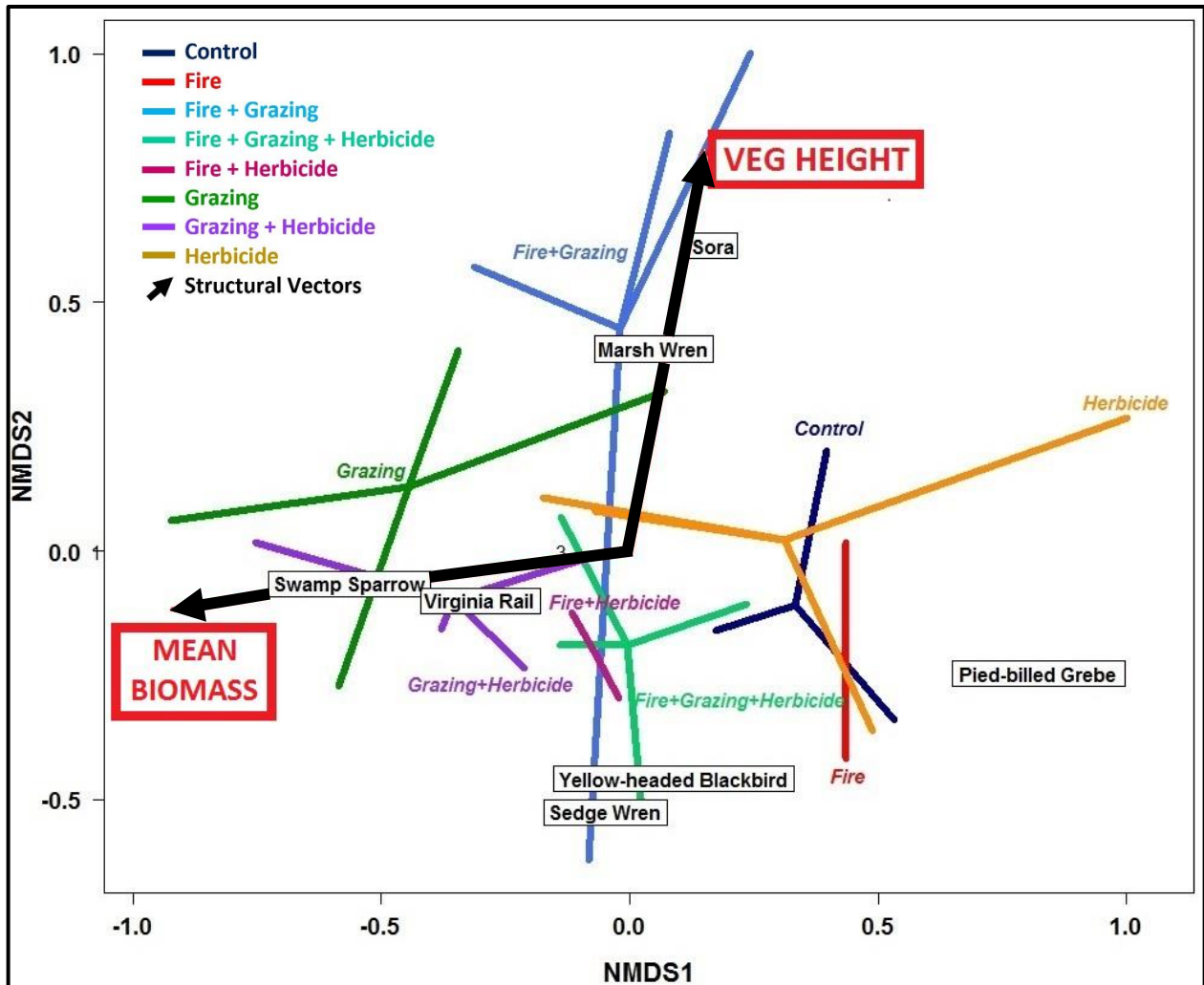
Figure 1.1: Neither passerine nor marsh bird abundance varied by treatment types after two years of management, however vegetation height and biomass are significantly correlated with variation in bird community composition ($P = 0.02$ and $P < 0.01$, respectively) so effect of treatments on bird communities might be seen if treatments eventually affect vegetation structure. Ordination shows relationship between variables relative to each other over a gradient of similarity using distance. Here, the Yellow-headed Blackbird and Sedge Wren are more similar to each other than they are to the Marsh Wren who is most similar to the Sora. All treatments overlap and are in close proximity indicating that they are not different from each other. Biomass and vegetation height are depicted as vectors radiating from the center whose lengths indicate their relative importance in the ordination. Here, the Sora and Swamp Sparrow are most correlated with vegetation height and mean biomass, respectively, while the Pied-billed Grebe shows no similarity to either environmental variable, other species, or treatment.

The Pied-billed Grebe prefers large wetlands and is negatively associated with the

perimeter area so it is possible that we had few detections for this species and saw no

relationship between it and our significant environmental variables because the surveyed wetlands failed in these selection criteria (Figure 1.1) (Fairbairn and Dinsmore, 2001). Alternatively, and more likely, we suspect that we detected few Pied-billed Grebes because this species prefers wetlands with areas of open water and cattail invasion negatively affects the amount of open water present in wetlands (Solberg and Higgins, 1993). Ultimately, the heavily-invaded wetlands that we surveyed might not have had sufficient interspersion of open water areas for this species to be abundant (Solberg and Higgins, 1993).

Our community analysis showed variability among passerine species. Swamp Sparrows and Marsh Wrens were correlated with mean biomass and vegetation height, respectively, while Sedge Wrens and Yellow-headed Blackbirds showed no association with either (Figure 1.1). Sedge Wren and Marsh Wren patterns are consistent with past research indicating that Marsh Wrens are positively associated with emergent vegetation (cattails > 14%) while Sedge Wrens are not (Panci, 2013). In our ordination the Sedge Wrens and Yellow-headed Blackbird were not associated with cattail density or vegetation biomass, which contradicts past research that positively correlates these species with cattail density (Linz et al. 1996; Gabrey et al. 1999). Wren and blackbird densities are typically lower in wetlands treated with herbicide and vary in response to burning such that blackbirds prefer recently burned marshes while sparrows and wrens will avoid them (Linz et al. 1996; Gabrey et al. 1999). For this reason, we suspect that Sedge Wren and Yellow-headed Blackbird populations were affected by these relationships with fire and herbicide treatments but vegetative biomass and height were not the explanatory factor. Our results suggest that although these species do use invaded cattail marshes as habitat they are selecting for something other than stem density or height (Figure 1.1). Swamp Sparrows are positively associated with vegetation density and litter depth in wetlands invaded by reed

canarygrass *(Phalaris arundinacea)* (Kirsch et al., 2007), and so their positive relationship with cattails here stands to reason.

Overall, marsh bird community composition appears to vary along wetland vegetation gradients although there is no evidence that these gradients are being driven by the current management regime after two years of treatment. The effect of treatments on bird communities might be seen if treatments eventually affect vegetation structure.

Effect of Cattail Density on Species Abundance

Our results indicate no relationship between mean cattail biomass and bird abundance however a few trends emerge. For example, Soras and Virginia Rails were less abundant when the amount of cattail biomass was low (<1400 kg/ha) (Figure 1.2). This result was expected considering their established positive relationship with the amount of emergent vegetation in a wetland (Johnson and Dinsmore, 1986). A similar relationship is also observed with the Yellow-headed Blackbird whose abundance increased as cattail biomass increased, which is a trend that has been observed in past studies (Linz et al., 1996). For these three species there may be a minimum threshold for the amount of emergent vegetation that is required for an invaded marsh to be used by them. No trends emerge for the Pied-billed Grebe, Swamp Sparrow and Marsh Wren as they were detected consistently at all biomass densities, exhibiting no observable relationship between mean biomass and abundance. This suggests that mean biomass is not driving their use of wetlands, which corresponds to the results in our community analysis (Figure 1.1).
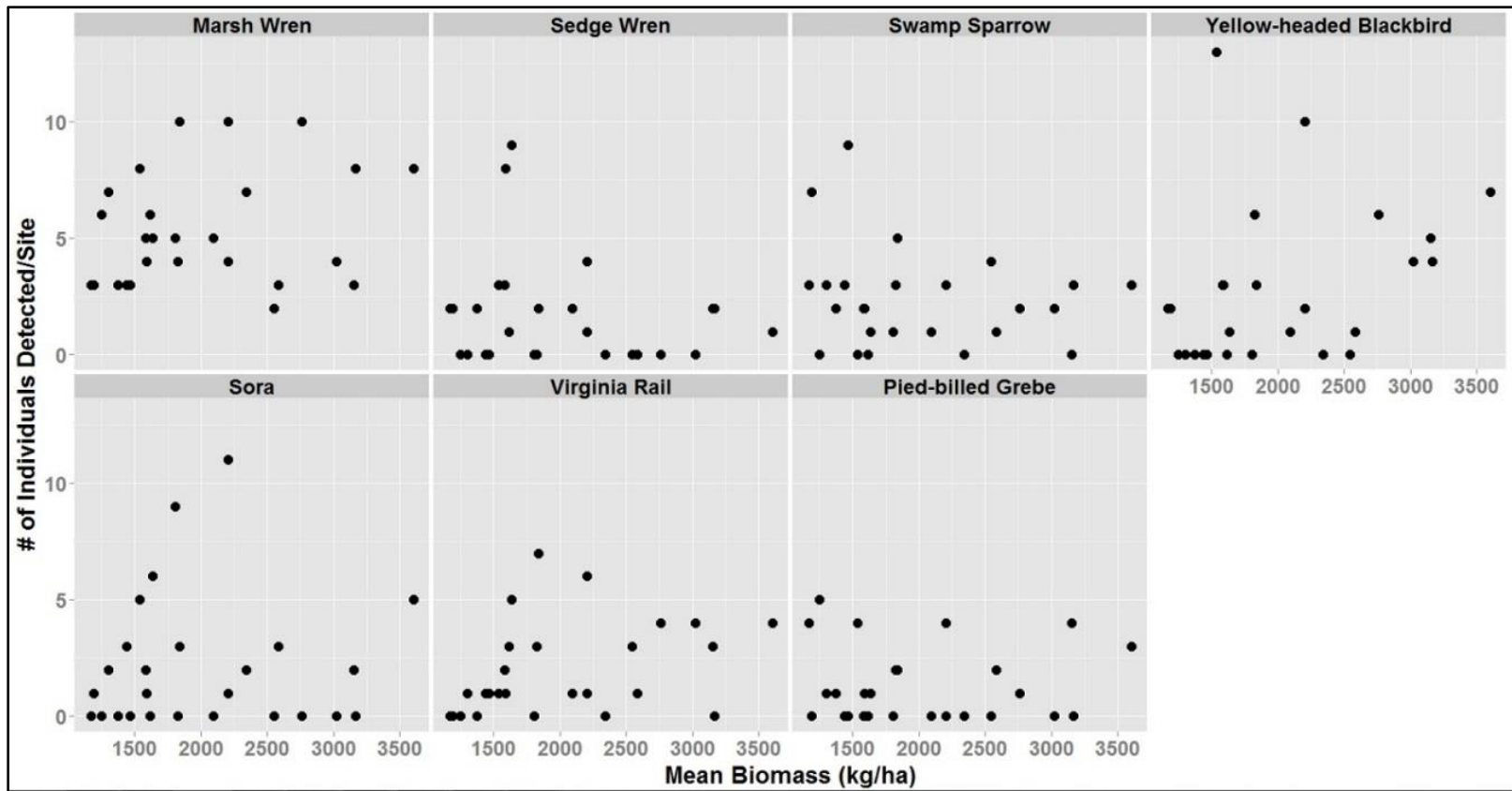
Figure 1.2: Mean biomass versus birds detected for each species indicating no relationship between the two variables for any of these populations.

Species-level Abundance and Density

Marsh Wrens and Sedge Wrens were detected 159 and 47 times, respectively. Soras and Virginia Rails were each detected 57 and 54 times, respectively. Yellow-headed Blackbirds were detected 76 times but assumption violations related to grouping observations in distance sampling precluded population estimates for this species.

Two passerine species with sufficient detections included the Marsh Wren and the Sedge Wren. Our model for the Marsh Wrens predicts 0.37 individuals per hectare (95% CI = 0.16—0.85) (Table 1.2) for a total of 519 Marsh Wrens (95% CI = 227–1186) (Figure 1.3) in the 1,395 hectares of emergent vegetation or cattails at Sherburne. Sedge Wrens, by contrast, are less abundant with a predicted 0.11 individuals per hectare (95% CI = 0.02–0.61) (Table 1.2) and a total population size of 154 (95% CI = 28–848) (Figure 1.3) Sedge Wrens on SNWR. Our results are not consistent with expected results as we anticipated a larger Sedge Wren population commensurate with how common this species is in this area.

Two secretive marsh bird species, Sora and Virginia Rail, had sufficient detections to - estimate abundance and predict total population. Our best model for Soras predicts 0.09 Soras per hectare (95% CI = 0.04–0.23) (Table 1.2) and 125 total individuals (95% CI = 48–324) (Figure 1.3) at Sherburne. Virginia Rails are more abundant than Soras at 0.14 individuals per hectare (95% CI = 0.04–0.44) (Table 1.2) and 193 birds (95% CI = 61–619) (Figure 1.3) predicted to be at SNWR.

Table 1.2: Density estimates and total abundance (with 95% confidence intervals) for two species of passerines and two species of secretive marsh birds at Sherburne NWR.

Model	K	AIC	Density	CV	Total Abundance
Marsh Wren					
Half-normal Cosine	2	496.94	0.372 (0.163-0.851)	43.77	519 (227-1,186)
Sedge Wren					
Half-normal Cosine	2	158.6	0.111 (0.020-0.609)	101.95	154 (28-848)
Sora					
Half-normal Cosine	2	204.86	0.089 (0.035-0.233)	50.62	125 (48-324)
Virginia Rail					
Half-normal cosine	2	199.82	0.017 (0.011-0.028)	63	196 (61-619)
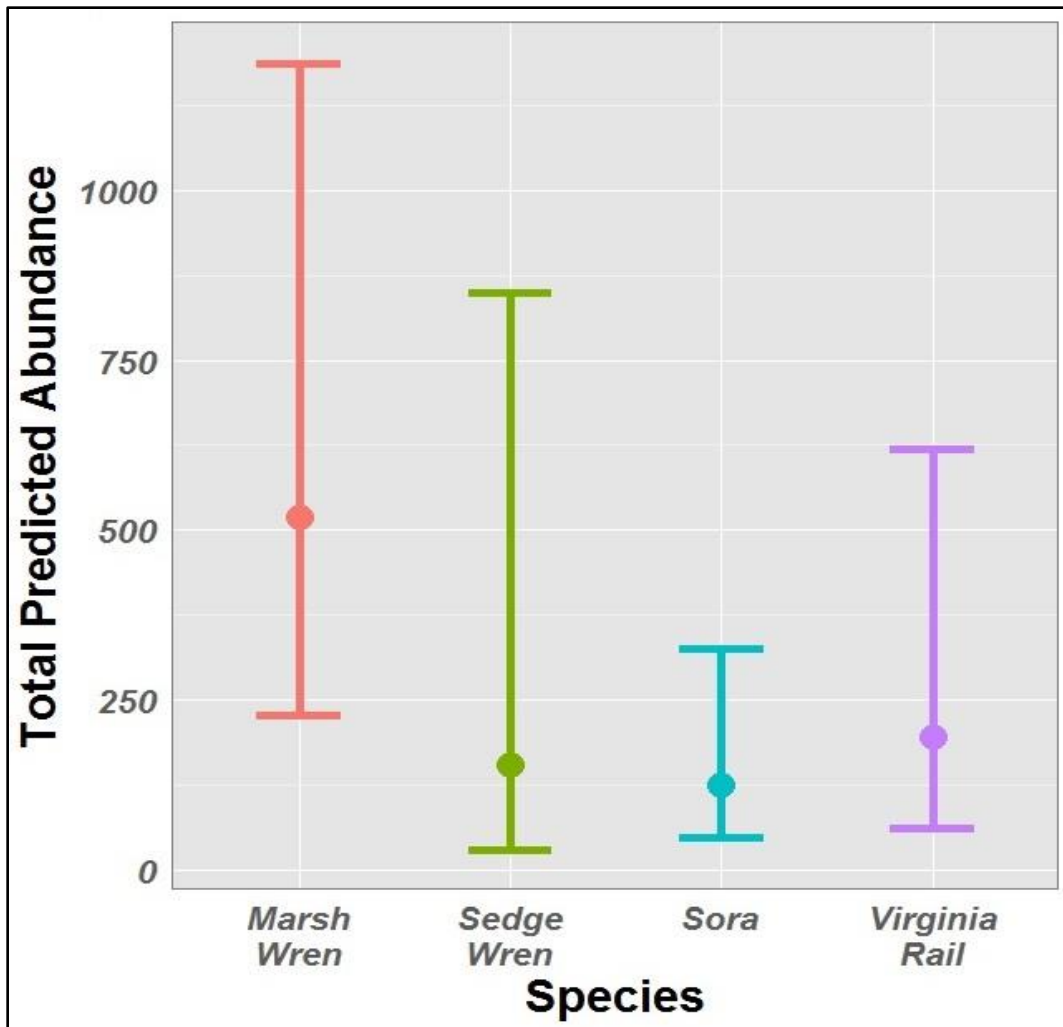
Figure 1.3: Populations estimates of surveyed species for all marshes within Sherburne NWR. Estimates were calculated using the total area of emergent vegetation, cattails or non-cattails, in marshes on SNWR (1,395 ha).

Population estimates for all species, especially under-surveyed ones like secretive marsh birds, are an important step towards understanding population sizes and trends yet there are few studies that attempt to obtain this information (Manci and Rusch 1988; Harms and Dinsmore 2012). Population estimates for the Marsh and Sedge Wrens, Soras, and Virginia Rails are valuable because they offer baseline population estimates for species that will be critical to their future management on both local and regional scale.

20

Our population estimates for all species are low compared to population estimates made in past studies by as much as 4.7 bird/ha for the Marsh Wren, 1.7 birds/ha for the Sedge Wren, 1.2 birds/ha for the Sora, and 1.4 birds/ha for the Virginia Rail (Manci and Rusch, 1988). However, our estimates are similar to those made by Harms and Dinsmore (2012) and only differ by 0.071 birds/ha and 0.083 birds/ha for the Sora and Virginia Rail, respectively. Harms and Dinsmore (2012) suggested that their population estimates were low because they sampled wetlands that were less-than-ideal habitat for secretive marsh birds. Similarly, we suspect our population estimates are lower than both studies because we surveyed heavily invaded marshes that are not ideal habitat for these species (Figure 1.4), although they do offer some characteristics that make them suitable as low-quality habitat. Additionally, we suspect that the applied treatment combinations could have had an indirect or long-term effect on populations outside the scope of this study as nest densities and adults are lower in wetlands post-burn and post-herbicidal application (Grant et al., 2011; Linz and Homan, 2011). This could contribute to our low population estimates as fire and herbicide were applied in half of the treatment combinations and pre-treatment, pre-cattail invasion population estimates do not exist for our study sites. Additionally, all wetlands sampled were heavily-invaded by hybrid cattail and we believe that sampling only low-quality habitat contributed to the low population estimates in this study.

Treatment Effects within Dynamic, Managed Landscapes

Water control structures are often used on refuges to manipulate water levels in wetlands and achieve specific management goals. These manipulations can affect the presence of some species of birds, especially marsh birds, who are dependent upon non-seasonal marsh habitat (Budd, 2007). Sherburne NWR uses water control structures to manage the water levels of

wetlands at SNWR and although these effects of management cannot be quantified within the constraints of this study they may have affected detection and bird abundance within sampled wetlands. In the first year of our study some sites had high water levels while in the second year there was little or no water present. Surveys at those points with little or no water were still conducted as vegetation/cover was still present and marsh birds do occasionally use dry areas for nesting (Meanley, 1953).
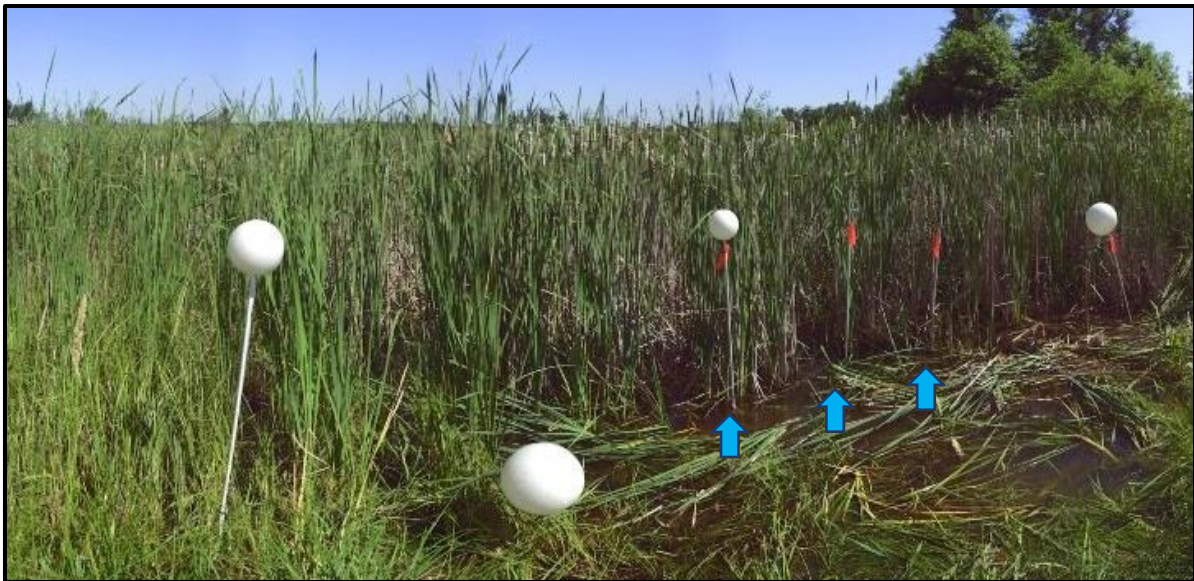
Figure 1.4: An example of a sampled wetland that shows the typical density and height of the heavily cattail-invaded wetlands that we sampled. The visible sections of the white posts are approximately 1.5-m tall. The blue arrows indicate the three posts marking the location of the quadrats that were destructively sampled.

Our data did not indicate a significant relationship between vegetative biomass and bird abundance at SNWR, but trends in the community-wide ordination suggest an association between some species and vegetation structure. These relationships could have been confounded by our small sample size and the possibility that past management altered the wetland dynamics in ways that are not obvious to the observer. The application of treatments like prescribed burning, grazing and herbicide might have a larger effect on a landscape that is not constantly

managed like that of national wildlife refuges. A recent or focal event on a natural landscape that has not been managed might experience a greater impact than one applied to a landscape like Sherburne NWR, which has experienced regular and constant management and disturbance. Managers at Sherburne have been managing this landscape as a prairie-oak savanna wetland complex for decades and the effects of past management efforts possibly obscured the immediate effect of treatments on cattail stands. Additionally, the effects of cattail invasion on heavily-invaded marshes are not reversed after cattail litter and live cattails have been removed from the wetland so it is possible that the effects of treatments on passerines and marsh birds might not be seen because of these long-term consequences to wetland dynamics (Farrer and Goldberg, 2014). In this sense, the effects of cattail treatment and removal might be better observed and quantified in the long-term so baseline population estimates such as these are critical to the management of these species.

Ultimately, the wetland complex at Sherburne NWR is part of a constantly managed landscape and the invasion of hybrid cattails is altering its structure, composition and hydrology, which might reduce the capacity of SNWR to mitigate impacts of global climate change (Euliss Jr. et al., 2006). SNWR lies in an area that is facing increasing demands from urban development, a factor shown to negatively affect marsh bird populations, as the Minneapolis-St. Paul, metropolitan area expands (DeLuca et al., 2004). Wetland complexes in agricultural and urban matrices offer respite and resources for migratory waterfowl and song birds to use as rest stops during migration or as habitat for nesting and breeding. For this reason, the restoration and management of the wetland complex on SNWR is critical on a local and regional scale where the effect of treatments to control invasive cattail on bird community dynamics and population sizes is unknown.

While the results of this study suggest no relationship between treatment and bird communities there is still great value in population estimates for conservation and management purposes. The effect of invasive cattails on habitat quality and quantity is negative and the short-term effect on bird populations is inconclusive; future research into the long-term effects of these treatments on bird communities might be more appropriate, especially if such research uses mean biomass and vegetation height to measure effect (Kostecke 2002; Kostecke et al. 2004). The first step to understanding the influence of management action on secretive marsh bird populations is to conduct more surveys that focus on these difficult-to-detect and work-intensive species.

Conclusion

We found that all treatments individually tended to reduce biomass however grazing alone is the most effective and its effect is dampened when combined with herbicide treatments. Despite reducing vegetative biomass the community dynamics at our sites are not affected. Ineffective management aimed at controlling the hybrid cattail and moderate to high levels of cattails does not positively or negatively affect the abundance of all marsh bird and passerine species surveyed. Clearly, there are structural and compositional factors at play that encourage some species to occupy invaded wetlands while discouraging others from occupying the same area. Further study could be done to delve into whether these stands could completely exclude specific species after a certain density threshold has been reached or whether niche usage by various species could be altered as a result of diminished wetland biodiversity and hydrological alterations. Also, the study of cattail treatment might be more suitable for long-term study as no effect was observed in this short term study and cattails continue to affect wetlands years after their removal.

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CHAPTER 2: POTENTIAL AND CHALLENGES OF ESTIMATING BIOMASS AND MODELLING VERTICAL STRUCTURE IN TEMPERATE SAVANNA AND WETLANDS WITH TERRESTRIAL LASER SCANNING

Abstract

Woody encroachment and invasive species reduce herbaceous vegetation, decrease biodiversity, homogenize landscapes, and alter ecosystem processes such as fire by altering the structure of wildland fuel beds. Structural degradation negatively impacts ecosystems around the world. In the Northern Great Plains, ecosystems like Prairie-Oak Savanna—include a mosaic of temperate savanna and wetlands, both of which are defined by structural characteristics. Prairie-Oak Savanna is a globally-unique endangered habitat type characterized by old-growth Bur Oak (*Quercus macrocarpa*) stands, expansive lowland, upland tallgrass prairie habitat, and wetland complexes. Previous land-use and management have allowed American Hazel (*Corylus americana)* and invasive hybrid cattails (*Typha x glauca)* to encroach upon the understory of oak savannas and marshes and are targeted by fire and grazing management. Conventional biomass harvesting is destructive, laborious and time consuming; technology like terrestrial laser scanning (TLS) has the potential to non-destructively sample with greater efficiency and finer resolution than conventional, destructive techniques. TLS has been used in other ecosystems to estimate woody plant biomass and model vegetation structure, but has not yet been applied in Prairie-Oak Savanna. We use TLS to model biomass in understories and wetlands and the vertical structure of savanna. Within each habitat type, TLS data are calibrated with clipped biomass across gradients of shrub and cattail abundance. We discuss the relative effectiveness and merits of TLS measurements as non-destructive estimators of woody biomass and vertical forest structure and our results indicate that TLS can be used to estimate biomass in some

ecosystems and that it can accurately model the vertical distribution of biomass that characterizes savanna structure.

Introduction

Ecological degradation occurs globally and takes many forms that are ultimately determined by the characteristics of the ecosystem and the influence of many factors so it is critical that management actions and project goals be tailored to specific ecosystems (Scholes and Archer, 1997). Some ecosystems are defined by their structure, which is characterized by the distribution and density of the vegetation; changes to these structural properties can often indicate ecosystem degradation (Scholes and Archer 1997; Brudvig and Asbjornsen 2009).

In the Northern Great Plains, ecosystems like Prairie-Oak Savanna include a mosaic of temperate savanna and wetland, both of which are defined by structural characteristics. Structural changes in these habitats is an indication of degradation and often occurs as a result of altered land-use and previous management (Archer et al., 2001). Savanna habitats are often degraded by woody encroachment that of woody vegetation moving into, and taking control of, the understory while wetland habitats are degraded by invasive species like hybrid cattail (*Typha x glauca)* (Langevelde et al., 2003; Zedler and Kercher, 2004). Ultimately, woody encroachment and invasive cattail cause rapid and drastic structural changes that reduces habitat quality and species diversity suggesting that structure determines savanna quality (Ratajczak et al., 2012).

Keystone structures are critical structural characteristics of an ecosystem that are associated with increased species diversity and their degradation causes a corresponding negative effect on biodiversity (Tews et al., 2004). Shrub encroachment degrades savanna's keystone structure (two-tiered vertical distribution of biomass consisting of scattered trees and continuous herbaceous cover) by increasing the shrub layer and homogenizing the understory. Wetland

37

vegetation structure is an important component of wetland habitat quality and is degraded when the invasive hybrid cattail (*Typha x glauca)* homogenizes structure by establishing monocultures that increases wetland biomass density (Zedler and Kercher, 2004). These changes effectively limit waterfowl access to an important food source and reduce the use of wetlands by waterfowl that select habitat based upon resource availability (Kosteche et al., 2005; Kostecke, 2002). Management in these ecosystems should focus on restoring the integrity of savanna structure by reducing the shrub layer and monitor changes in the vertical distribution and density of vegetation biomass to determine if objectives are being met (Elzinga et al., 1998; Langevelde et al., 2003).

Terrestrial laser scanning (TLS) is a high-resolution, ground-based remote sensing method. It can provide accurate structural information on trees and individual shrubs (height, volume, biomass, and crown area) and has demonstrated an ability for field applications (Adams, 2014; Lefsky and McHale, 2008; Richardson et al., 2014; Vierling et al., 2013). TLS collects data in the form of "point clouds", a collection of data points with three-dimensional coordinates that model the scanned area. The scanner collects these data by measuring the angle and rate of return of laser pulses that reflect off encountered surfaces (Figure 2.1A). The scanner uses the angle of the emitted pulse and the time it takes for that pulse to return to calculate the orientation of that object is relative to the scanner and these data points are recorded as unique entities on an x, y, z coordinate system (Figure 2.1B) (Shan and Toth, 2008). The final product is a scan with millions of points that can be interpreted as vegetation biomass and accurately depicts plant structure, volume and height, and structural forest variables (Figure 2.1C) (Huang and Pretzsch, 2010; Lefsky and McHale, 2008). TLS can sample data at a fine scale so, given a clear field-of-view, it can collect hundreds of points returning from even small surface areas such as a twig or

38

leaf. When several scans of the same area from different angles are combined the TLS scanner

has estimated biomass, such that each returned pulse is hitting biomass while pulses that are sent

out but not returned are not.
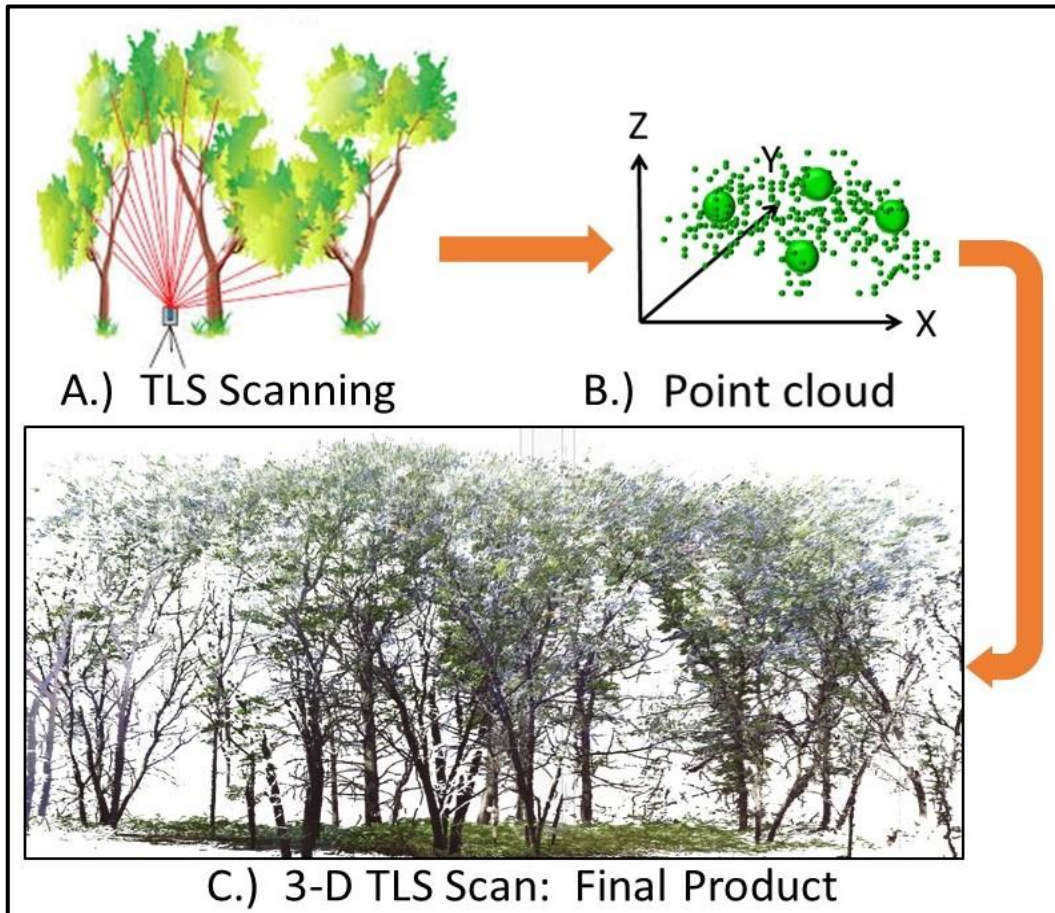
Figure 2.1: **(A)** TLS scanner collecting data (Lutz 2011), **(B)** Representation of
TLS point cloud, with reference spheres and points as discrete entities on x, y, z
axis (Olsoy 2013) and, **(C)** Final product of a TLS scan showing a 3-D
representation of the scanned site.

There are two types of laser scanning, Lidar and TLS, that both use concentrated light

pulses to collect data on scanned areas (Shan and Toth, 2008). Lidar is aerially-based and

commonly used in ecology for measuring characteristics such as canopy height in forested areas.

TLS uses similar technology as Lidar however it cannot cover the same large area as Lidar and

instead covers a smaller area with much finer resolution. Previous work using aerial Lidar underestimated variables such as shrub canopy height and volume due to its aerial-based design and technological limitations that restricted the density of the point clouds to less than 10 points/m^2 (Glenn et al., 2011; Streutker and Glenn, 2006). Comparatively, technological advances associated with TLS allow it to gather high density point clouds (thousands of points/m^2) using a ground-based design that offers a horizontal view of the forest and isn't obstructed by the canopy (Olsoy, 2013) (Figure 2.1).

TLS has the potential to non-destructively sample more efficiently than conventional sampling techniques because each scan models a larger area in less time than conventional sampling and from that scan several samples can be extracted. While TLS has been used in other ecosystems such as Sagebrush Steppe, deciduous woodlands, peatlands, and conifer forests to estimate woody plant biomass and model vegetation structure it has not yet been applied in all ecosystems (Anderson et al., 2009; Dassot et al., 2011; Lefsky and McHale, 2008; Vierling et al., 2013; Yao et al., 2011). Because data quality declines in high density forests, TLS might be most appropriate for relatively open or low-density landscapes like savannas and wetlands that are defined by structural characteristics (Adams, 2014; Huang and Asner, 2009; Lefsky and McHale, 2008; Lettow et al., 2014; Tuchman et al., 2009; Vierling et al., 2013; Watt and Donoghue, 2005).

TLS is economical, efficient, and accurate so developing methodologies for using this technology in the field is imperative for today's managers who are often responsible for monitoring the effect of management action and ecosystem restoration on a landscape-level scale. Ultimately, this technology has the potential to offer managers something that they desperately need: an alternative to conventional monitoring techniques without the associated

limitations. TLS has the potential to gather data that gives a full and accurate picture of the effects of management on landscape structure by combining the fine-scale and precision of conventional sampling methods with the rapidity and large scale of aerial Lidar; all of which are vital components to effectively monitor landscape-scale restoration and management efforts. We sought to determine whether (1) TLS can accurately estimate herbaceous and woody aboveground vegetation biomass in savanna and wetlands habitats by developing TLS methodology and testing it against conventional sampling techniques; and (2) TLS can model the vertical distribution of biomass over a gradient of both understory and canopy density in savanna habitats.

Materials and Methods

Study Area and Site Selection

This study was conducted during July and August, 20l5-2016 at Sherburne National Wildlife Refuge (SNWR) in Sherburne County, MN (lat 45.483951, long -93.711645). SNWR lies within the Mississippi Headwaters/Tallgrass Prairie ecosystem and is part of the Anoka Sandplain, situated in the transitional zone between tallgrass prairies to the West and boreal forests to the East (USFWS 2005). It has sandy soils, many wetland complexes and features 732 acres of a globally-endangered habitat type, Prairie-Oak Savanna that is characterized by old-growth Bur Oak (*Quercus macrocarpa*) stands, expansive lowland, and upland tallgrass prairie habitat (Leach and Ross, 1995; USFWS, 2005). All savanna habitat at SNWR is experiencing some degree of degradation and woody encroachment, primarily composed of American Hazel (*Corylus americana*), is increasing the shrub layer. The wetland complexes at SNWR are negatively affected by hybrid cattail (*Typha x glauca*), an invasive aquatic plant that

homogenizes wetland structure and increases canopy density by forming dense monodominant stands that exclude species.

Wetland Habitat: Calibration of TLS with Harvested Cattail Biomass

Twenty-one cattail plots comprised of three, 0.5-m^2 quadrats (n=63) were sampled in July 2015 and 2016. Quadrats were placed on the edge of the wetland 0.5-m apart and all vegetation within the quadrats was clipped at water level and bagged; harvested biomass was oven-dried at 60 degrees Celsius for 48 hours and massed.

Savanna Habitat: Calibration of TLS with Harvested Shrub Biomass

17, 1-m^2 quadrats were established across SNWR along a gradient of shrub and canopy density in August, 2014. All woody shrub vegetation within the quadrat was clipped at ground level and bagged; harvested biomass was oven-dried at 60-degrees Celsius for 48 hours and massed.

TLS Methods: Wetland and Savanna Calibration

In both calibration projects, quadrat corners were marked with white fiberglass posts and several reference spheres, white spheres in various sizes that the scanner identifies and uses as markers to sew multiple scans together, were placed at varying heights and depths around the quadrat to facilitate rapid post-processing scan alignment (Figure 2.1). Quadrat locations were chosen such that no downed logs or tree trunks (DBH > 2.5 cm) were included as all scanned biomass needed to be harvestable for analytical comparison. Each quadrat was scanned from 5-m with a FARO Focus 3D S terrestrial laser scanner and the field-of-view between the scanner and the quadrats was cleared of all obstructions. Three scans were taken from three different angles at each quadrat, one low-resolution 360-degree scan and two high-resolution partial scans that covered between 120 and 360-degrees (Figure 2.2A, B). The high-resolution scans require more

time for the scanner to complete than the low-resolution scans so a smaller area covered was more time-efficient; however, the degree chosen depended upon site characteristics that determined reference sphere placement. All reference spheres must be within the scanned area to facilitate post-processing scan alignment so at sites that had few places near the quadrat to hang reference spheres we had to place them far apart or behind the scanner, which required that the high-resolution scans cover a greater area. The scanner required between 3 and 8 minutes to complete a single scan so the amount of time required to prepare and scan each quadrat three times was variable and depended upon several things including: the resolution and degree of the scan, how difficult a site was to navigate and prepare (understory especially dense or tall, insufficient spots to place reference spheres, etc.), how much vegetation needed to be cleared from the field-of-view between the scanner and the quadrat, and placing reference spheres such that they were visible from all scanner angles. Ultimately, each site could be scanned in 22-45 minutes.

Processing and alignment of the scans was completed using FARO Scene 5.1.3 (FARO Technologies Inc. 2012). All three scans were preprocessed and aligned using the artificial reference spheres that were placed around the quadrats in the field. All quadrats were marked in the field with white fiberglass posts that are identifiable in Scene so we used them to isolate the points associated with the biomass that was destructively sampled (Figure 2.3). The maximum height of all destructively sampled wetland and savanna vegetation (height $\leq$ 2-m) was used to determine the height of the column that was isolated and exported from Scene such that it included all sampled biomass at the wetland and savanna sites. We summed all points within the exported column using MATLAB 8.4 (The Mathworks Inc. 2014)
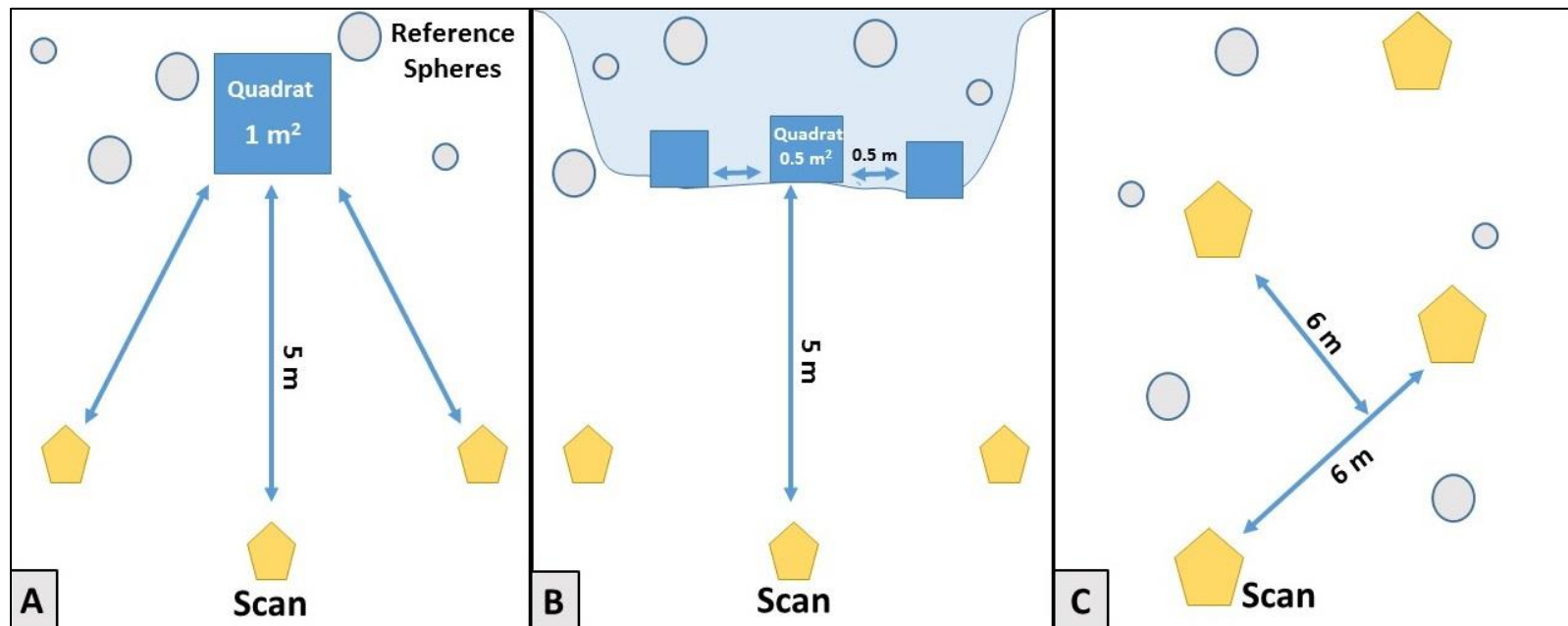
Figure 2.2: A) Destructive shrub sampling design, B) Destructive cattail plot, C) Non-destructive plot design. Circles are reference spheres, hexagons are scan locations.
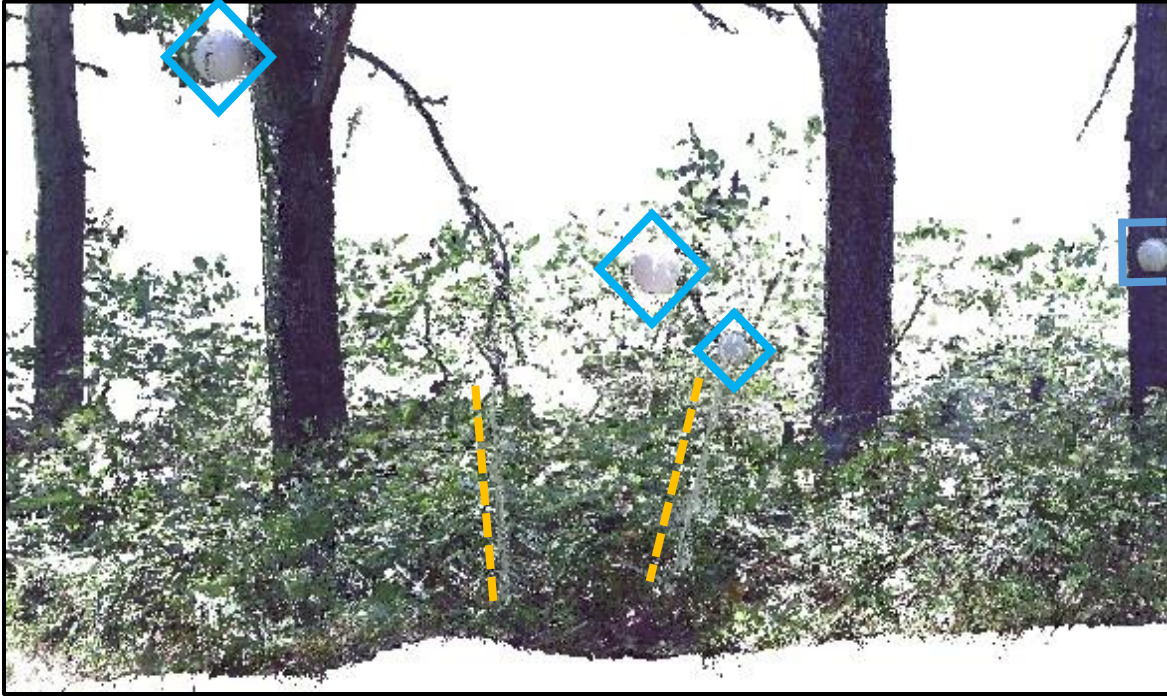
Figure 2.3 A: An example of the 3-dimensional TLS representation of a woody shrub quadrat (A) with white posts (to the right of ▬▬) and reference spheres (◆) visible.

Savanna Habitat: Modelling Vertical Structure

To model the vertical structure of savanna we used TLS as a non-destructive sampling technique at ninety-seven sites that had a gradient of both understory and canopy density and height. We visited these sites in July-August, 2015-2016. At each site reference spheres were placed around the scanned area at varying heights and depths. Then, four high-resolution, 360-degree scans were taken with the scanner locations being 6 meters apart in a zig-zag pattern to cover a 9-m by 6-m area (Figure 2.1C). This pattern was used to ensure scan overlap and allow for the maximum potential of scan alignment, even at sites with dense or tall understories that could obscure reference sphere detection. In the lab, the scans were preprocessed and combined using the reference spheres and FARO Scene 5.1.3 (FARO Technologies Inc. 2012).

For each site, 5, 2-m^2 subsamples (n=485) that extended from ground level to the maximum canopy height were exported from Scene (Figure 2.4). All points within the exported column were summed in vertical 0.1m bins along the Z-axis using MATLAB 8.4 (The Mathworks Inc. 2014).

Figure 2.4: TLS scan with area of interest isolated for exportation.

Analysis

Calibration of TLS Point Data

We tested for a relationship between TLS point count data and destructively sampled shrub biomass with a mixed-effect linear regression (LME) model in the R statistical

environment (RStudio Team 2015). An LM model compared the response variable, total point count, to our predictor variable, the log-transformation of total biomass. We then compared our model against the null using anova and estimated the R^2 value for our model in the R Statistical Environment.

Savanna Habitat: Calibration of TLS Point Data

We tested for a relationship between TLS point count data and destructively sampled cattail biomass with mixed-effect linear regression (LME) modelling in R. An LM model compare the response variable, total point count, to our predictor variable, the log-transformation of total biomass. We then compared our model against the null using anova and estimated the R^2 value in R.

Savanna Habitat: Modelling Vertical Structure

We tested the capabilities of TLS to model the vertical distribution of biomass and estimate transition points between vegetative layers with an accumulation function and segmented regression in R. We fit an accumulation function to the 0.1 meter bins from ground level to maximum canopy height and then used a linear model to compare our response variable, accumulated point count, to our predictor variable, height. Next, we estimated an unbroken-line segmented regression model with two breakpoints for each site using the segmented package in R (Muggeo, 2008, 2003).

To model the woody understory we isolated all sites where the first break point occurred at, or below, 2-m. We used biomass estimates from TLS to determine three levels of understory density using the density function in R, assigning each site a categorical variable (low, medium, high). We tested for a relationship between our response variable, the slope of the first segment (the accumulation of biomass in the understory), with our biomass levels, our predictive variable

47

using generalized linear mixed-effect regression models (GLMM) models using the glmer function in the lme4 package in R (Bates et al., 2015), with plot and quadrat as random effects. Then, we tested for model significance using the anova function in R (Bates et al., 2015). Finally, we conducted a pairwise comparison of all treatments using the glht function with a Tukey-test in the multcomp package in R (Hothorn et al., 2008).

Additionally, we use linear mixed-effect regression models (LME) to investigate the relationship between the slope of the first segment and biomass. Then, we calculated the R^2 value for our LME models, and tested for model significance using the anova function in R (Bates et al., 2015).

Results

Wetland Habitat: Calibration of TLS against Clipped Cattail Biomass

The results of our linear regression model indicate that no significant relationship exists between TLS point count and harvested biomass (P = 0.14) (Figure 2.5). Our R^2 value indicates that only a small amount of the observed variation is explained by our model ($R^2 = 0.04$).

Savanna Habitat: Calibration of TLS Against Clipped Shrub Biomass

In accordance with our hypothesis that TLS can be used to accurately estimate woody shrub biomass, our results indicate a significant and positive relationship between total point count and harvested biomass for shrubs (P = 0.02) (Figure 2.6). Our R^2 value estimates that twenty-seven percent of the observed variation is explained by our model ($R^2 = 0.27$).

Savanna Habitat: Modelling Vertical Structure

Our results indicate a significant relationship between slope and vegetation biomass levels (P < 0.01, $R^2 = 0.58$). In the pairwise comparisons, all biomass levels showed statistical

significance (Low, Medium and High: P < 0.01). Our results indicate a positive and significant

relationship between vegetative biomass and segment slope (P < 0.01, $R^2 = 0.39$) (Figure 2.8).
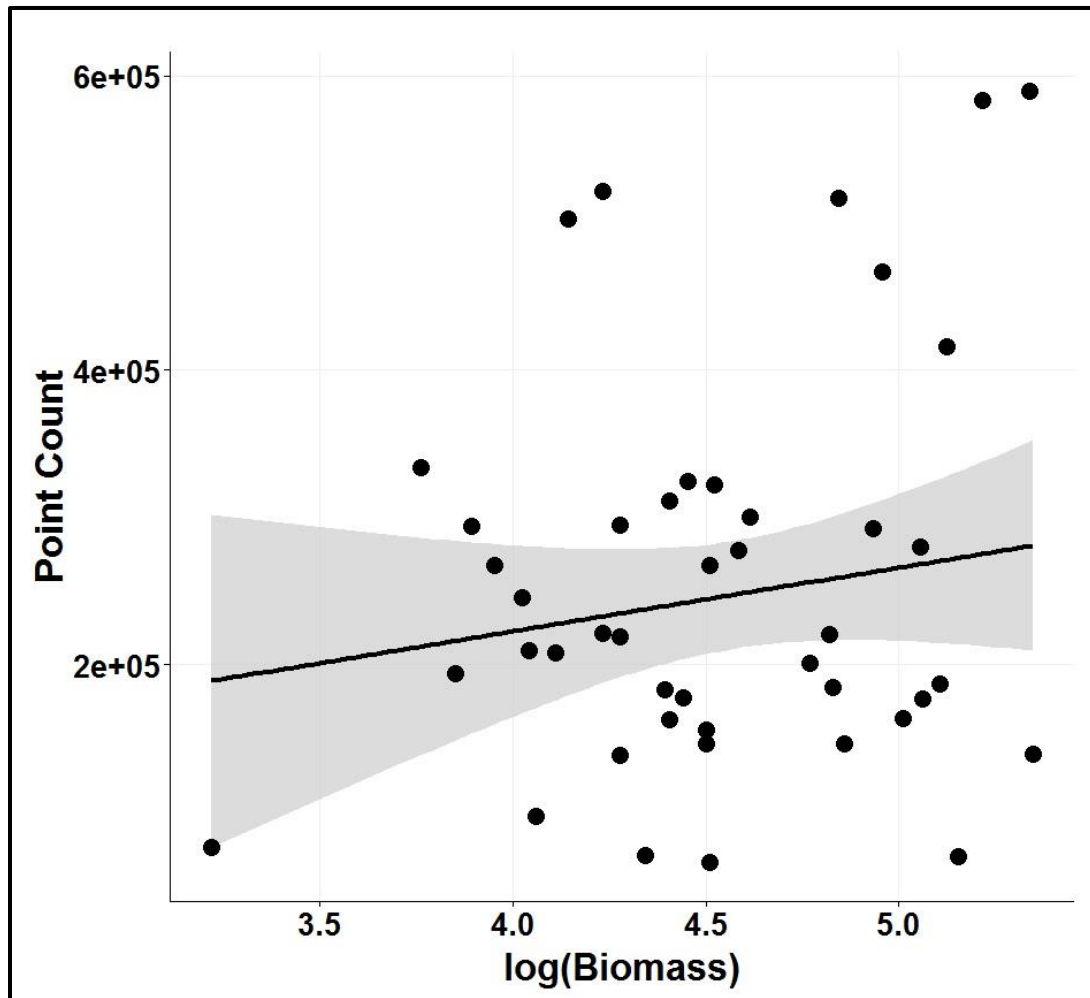
Figure 2.5: The relationship between conventionally-sampled hybrid cattail biomass and TLS point count data is positive but not significant (P = 0.14, $R^2 = 0.04$).
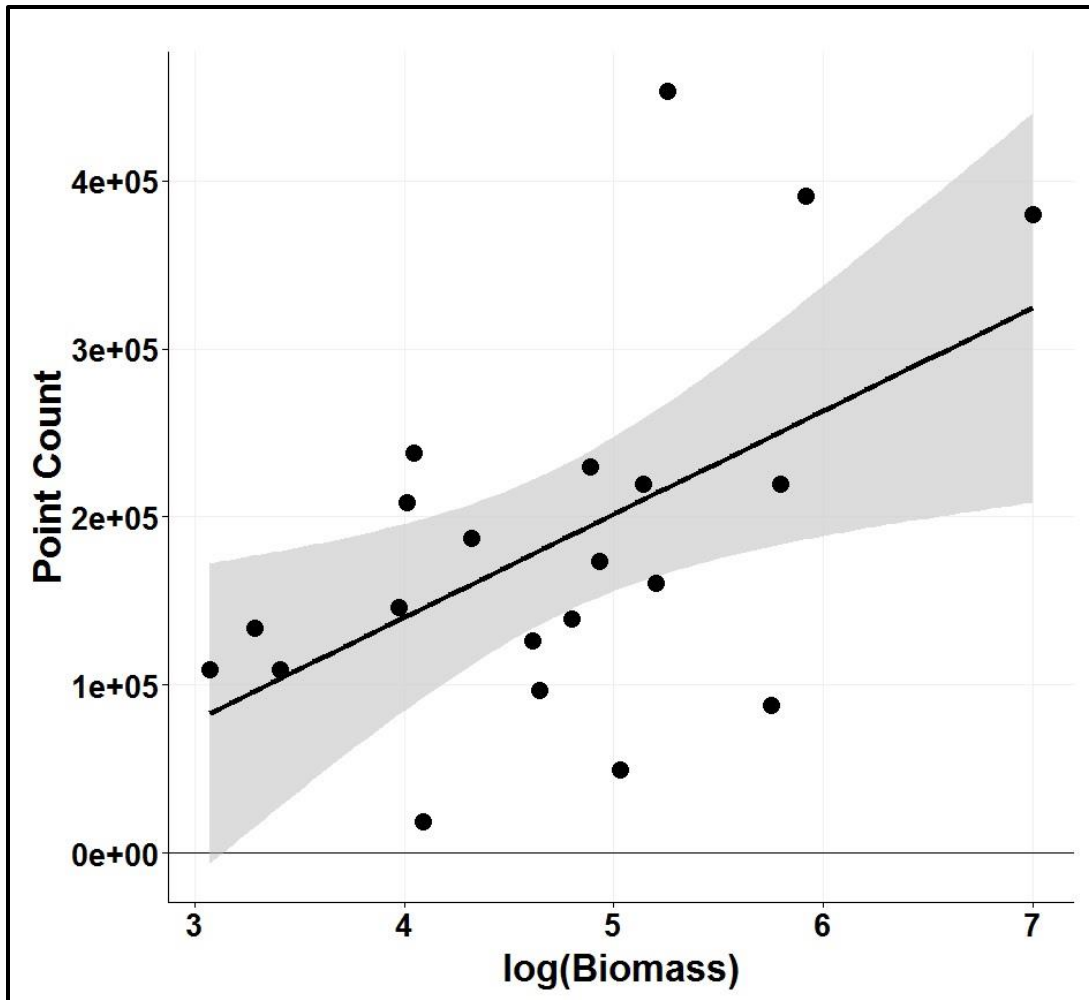
Figure 2.6: The relationship between conventionally-sampled shrub biomass and TLS point count data is positive and significant (P = 0.02) and explains twenty-seven percent of the observed variation ($R^2 = 0.27$).

The results from our segmented regression indicate that the vertical distribution and density of biomass and savanna structure can be modelled using TLS (Figure 2.7). The segmented regression shows how biomass is distributed vertically in a savanna, identifies where structural changes occur, facilitates the comparison of structural characteristics across sites, and allows the comparison of the same site across years so if structural characteristics are altered by management it can be measured statistically to quantify change and measure progress.
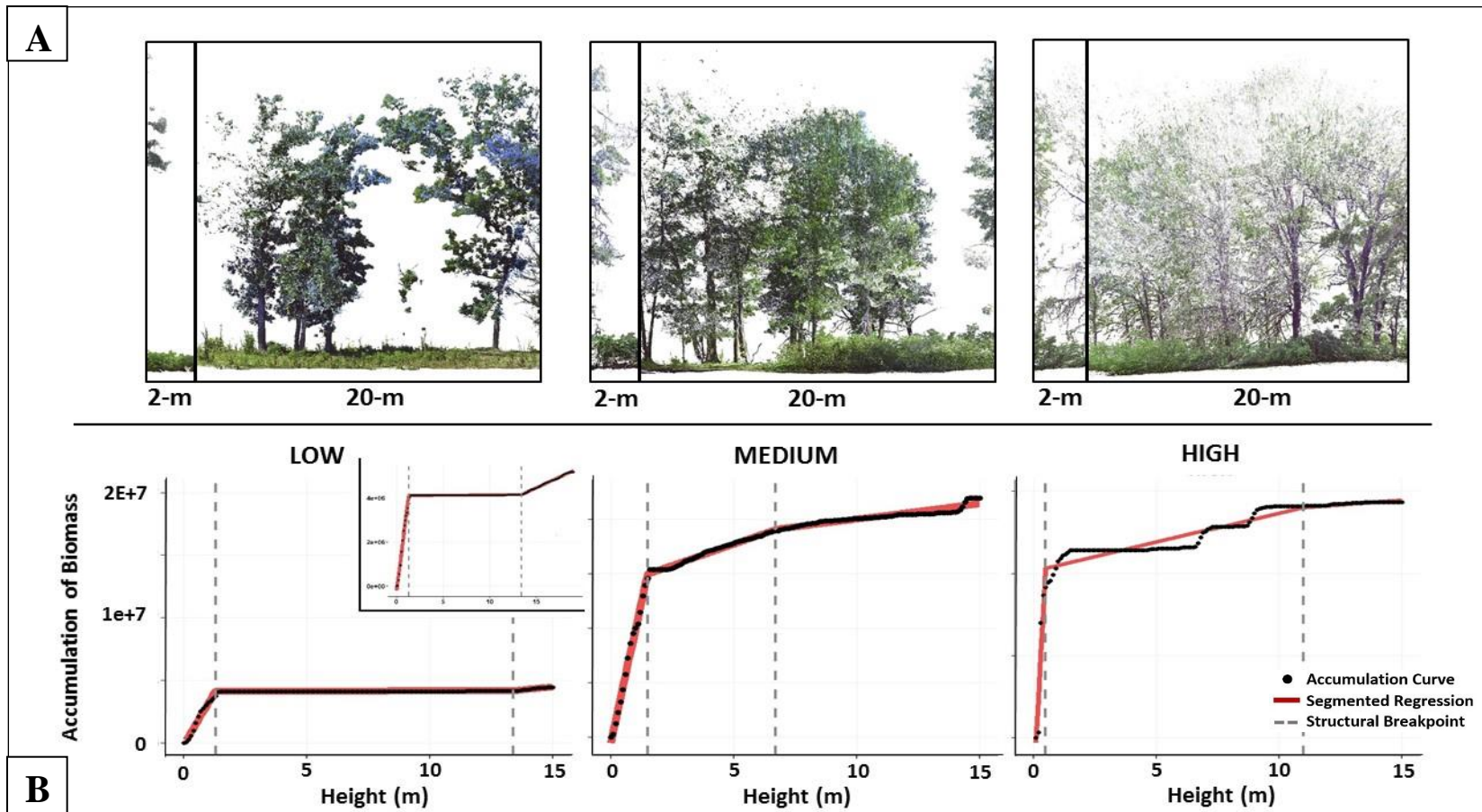
Figure 2.7: Point density accumulation curve for three quadrats with high variability across three vegetation biomass levels, Low, Medium and High. (A) TLS orthophotos of sites across all understory biomass density levels, 2-m exports are the sites exported from scene and used in the accumulation curves and segmented regressions (B) and are next to an orthophotos that shows a 20-m^2 area of that site. (B) Rapid biomass accumulation curves with height (locally steep slopes in curves) show areas of dense vegetation; shallow slopes denote gaps in vegetation and canopy structure. The structural breakpoints in our segmented regression show where structural changes occur. The slope of the understory segment (Height<2-m) shows density of vegetation. Steeply sloped segments show dense understory vegetation. As biomass levels increase from Low-to-High the slope of the segmented regression becomes progressively steeper. All regressions are scaled to the same biomass amount, however the Low level is inset to better illustrate its accumulation on a smaller scale as the increase in the canopy layer is obscured at the larger scale.
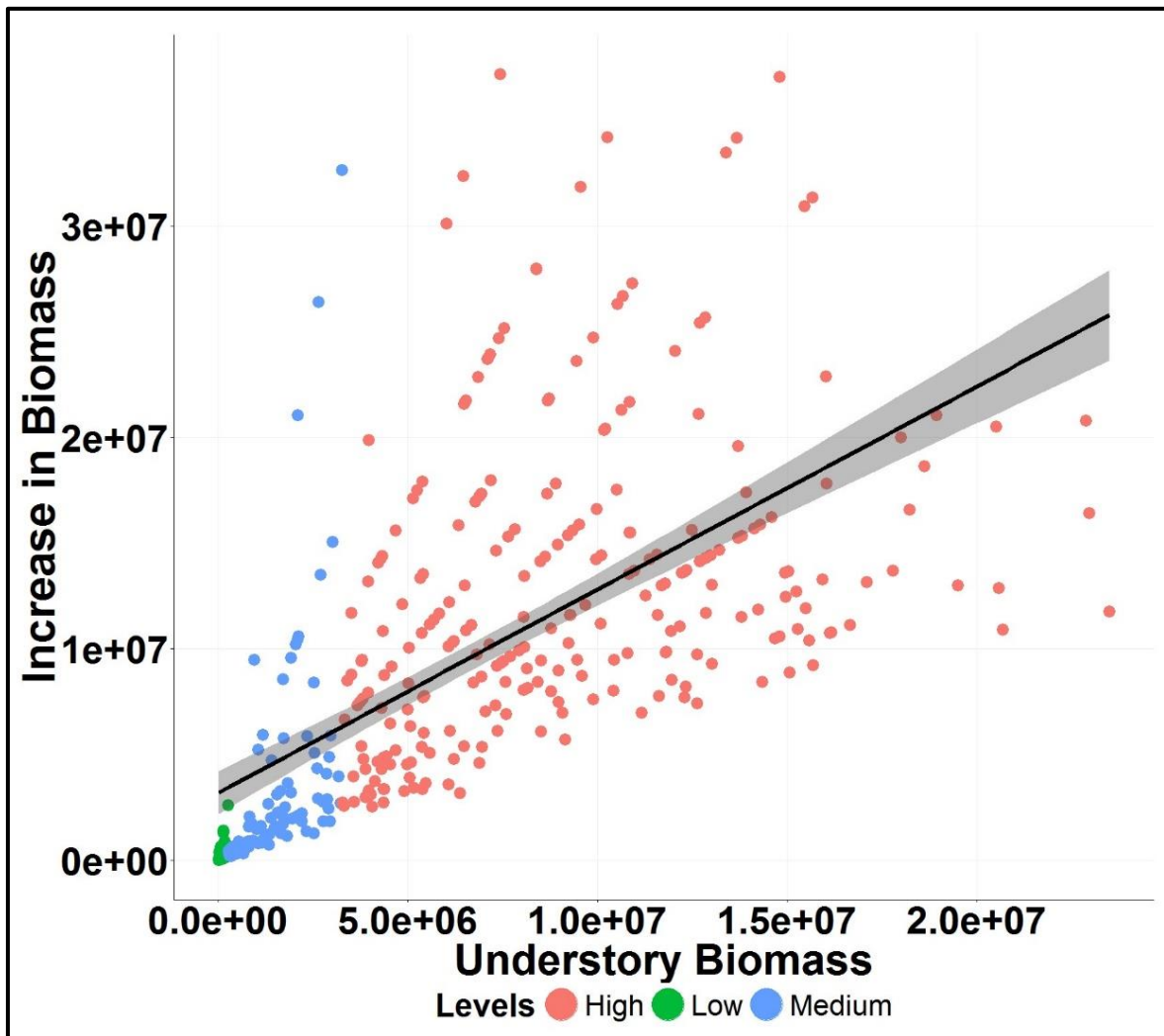
Figure 2.8: Our results indicate a positive and significant relationship between vegetative biomass and segment slope (P < 0.01, $R^2 = 0.39$).

Discussion

Wetland and Savanna Habitat: Calibration

Previous research has demonstrated that TLS has the ability to provide accurate structural and volumetric information on trees and shrubs however these studies were conducted under controlled circumstances and considered only one or two plants at a time (Lefsky and McHale

2008; Olsoy 2013; Adams 2014). This study builds upon this research by taking TLS out of controlled environments and into the field, testing its ability to gather accurate data in a field setting. The methodology presented here could be applied in other biomes to accurately estimate the biomass of a woody understory and structural forest variables but may not be ideal in all settings as we found no significant relationship between biomass and TLS point count for invasive cattails.

TLS cannot collect quality data in dense forest stands and we suspect that this limitation contributed to our insignificant result for cattail biomass as the marshes we sampled were heavily-invaded by dense monocultures of hybrid cattail (Figure 2.9, 1.4) (Watt and Donoghue, 2005). Additionally, structural differences in the sampled biomass, sturdy woody shrubs with broad leaves and greater surface area versus flexible, herbaceous cattails with thin leaves and little surface area, likely contributed to the differences in results seen between the wetland and savanna habitat types. Cattails have long, supple leaves that move even when there is very little wind and this is problematic for accurate TLS sampling as scanning moving objects can lead to both under- and over- estimations. A moving object can contribute to underestimations by being missed entirely or by returning laser pulses being obstructed by vegetation that bends in front of it and effectively stops the laser from receiving the returning pules. Conversely, TLS can overestimate biomass by double-counting a moving leaf that returns pulses from its starting and ending locations.

Our wetland habitat regression shows that there was a broad range of point counts with many sites having higher or lower estimates of point count across all ranges of harvested biomass (Figure 2.5); we believe that this is due to the flexible structure of the cattail plants

Figure 2.9: This sampled wetland that shows the density and height of the heavily-invaded cattail stands present in the wetlands that we sampled.

combined with the unsheltered, open nature of wetland ecosystems that allowed the cattails to bend and move even on days with calm weather and no wind. Comparatively, the woody shrubs are sheltered from wind by the trees around them and can remain still even in moderate wind.

Ultimately, TLS can model the structure and density of savanna understory, woody shrubs and the vertical distribution of biomass that is so critical to a quality savanna. We were able to successfully sample across a gradient of vegetation density, height, and structural characteristics despite variable environmental circumstances and weather. TLS can consistently identify thresholds or points where the vertical structure of the site changes from one vegetative layer to the next and this information would be invaluable to managers implementing management strategies on ecosystems with keystone structures such savanna (Tews, 2004). Changes in the height of these thresholds in the segmented regression and the slope of the regression lines could ultimately quantify degradation across the landscape or quantitatively measure changes that result from management action (Figure 2.6).

Our results show that TLS can be used as a field sampling method to gather data as it is consistent and dependable. TLS can model two sites with the same amount of biomass and, with segmented regression and accumulation curves, show that although the biomass is the same the structural characteristics are different. A good example of this is Figure 2.7B, the medium and high sites are categorized differently yet they have the same amount of biomass. Ultimately, TLS is modelling structural variables that are difficult to gather with conventional sampling methods, and it's doing it quickly and non-destructively. Additionally, fewer resources are needed to sample with TLS as your data is digital and stored on a hard drive; you do not need drying ovens or vehicles to haul samples and you don't run the risk of losing and samples during transport.

The benefits of monitoring with TLS in terrestrial ecosystems are many yet there are some drawbacks that include its inability to collect species-specific data and environmental limitations that restrict sampling to days without precipitation, fog, or wind. Its inability to collect species-specific data means that TLS may only be appropriate for qualitative measurements or for projects that are concerned with changes to structural properties like the vertical distribution of biomass or understory density and not what species are responsible for these changes. Pairing TLS with non-destructive techniques that address composition would be advisable in projects concerned with species composition and would be relatively easy to incorporate during the down-time associated with waiting for scans to run. Despite the limitations presented here the use of TLS as a tool for gathering fine-scale data in the field and for estimating biomass rapidly and accurately is a viable and realistic alternative to conventional sampling methods.

Savanna Habitat: Modelling Vertical Structure

Ultimately, TLS is effective at estimating biomass, modelling shrub density and characterizing vertical structure at savanna sites so could be used as a tool to guide management decisions on a qualitative level. Our results, combined with those of past studies that indicate TLS can accurately gather detailed structural information on individual shrubs and trees along with spatial information on stem location and basal area, implicate the greater ability of this technology to model several forest variables simultaneously (Lefsky and McHale 2008; Yao et al. 2011; Olsoy 2013; Adams 2014; Olsoy et al. 2014). Additionally, a greater amount of data is collected in less time compared to conventional techniques; further justifying its value to field researchers and managers monitoring at a landscape-scale level. In the short amount of time it took to sample one non-destructive plot we collected accurate data on a 20-m^2 area and exported

5, 2-m^2 subsamples from that data. This illustrates the efficiency and scale of TLS data as conventional techniques cannot collect this same data; even if it were possible for conventional methods to collect the same data it would likely require more time and specialization, making it less economical than TLS. Future applications of TLS should focus on developing and implementing this sampling technique in the field along with investigating its ability to characterize fuel beds, estimate non-woody biomass or quantify changes in the spatial and vertical distribution of vegetation resulting from ecosystem management.

Conclusion

Although TLS was not reliable at estimating cattail biomass in invaded marshes it can be used as a tool to estimate biomass and investigate structural characteristics of woody shrubs in terrestrial ecosystems like temperate savanna. We successfully calibrated this technology against harvested biomass, establishing that it can be used to non-destructively estimate shrub biomass in the understory. Additionally, TLS can characterize the structural differences between pristine, degraded and restored savanna sites, showing that it can be used by managers as a tool to monitor structural changes that result from management action. With further study this technique could be developed to quantify how degraded a site is or estimate the change affected by management action over time. Managers are often limited by funds and time and, because TLS is efficient and effective, TLS could advise managers on where to concentrate resources, thereby affecting the greatest change at the sites that need it most. Ultimately, TLS presents a solution for monitoring terrestrial ecosystems that managers require: a non-destructive sampling method that isn't limited by time, project scale, or precision.

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CHAPTER 3: STRUCTURAL RESPONSE TO BURNING AND GRAZING IN WOODY-ENCROACHED TEMPERATE SAVANNA

Abstract

Encroachment of woody plants into savanna understories changes vertical structure of biomass, reduces herbaceous vegetation and alters savanna fuelbeds. Oak savanna occurs globally and is characterized by old-growth oak (*Quercus* spp.) stands and herbaceous understory. Previous land-use and management can allow woody vegetation to dominate the understory and degrade these dynamic ecosystems making control a primary objective of fire and grazing management. We used terrestrial laser scanning (TLS) to model vertical structure across four treatments: burned, grazed, burned/grazed and unburned/ungrazed in temperate savanna. We used TLS because it can quantify changes in understory vegetation and model the vertical distribution of biomass rapidly, non-destructively and in fine detail. Our results indicate the there was no significant structural response to treatments however we successfully used novel TLS methods to model the relative change in understory biomass over a gradient of woody-encroached savanna habitat.

Introduction

Anthropogenic influences degrade ecosystems around the world, oftentimes by allowing new suites of invasive species or natives that were once controlled as the ecosystem maintained itself, to expand into and take control of the landscape. Fire suppression, altered grazing regimes, and fragmentation degrade ecosystem condition and can even drive some ecosystems to transition into another (Archer et al., 2001; Brudvig, 2011; Petersen and Drewa, 2009; Peterson and Reich, 2001). For ecosystems like savannas that are defined by their structure (i.e., the distribution and density of the canopy, understory height, vegetative layers, and the spatial

distribution and ratio of herbaceous, shrub, and tree cover) degradation can be manifested as canopy ingrowth, reduced herbaceous cover, spatial and vertical homogeneity, and increased woody vegetation (Scholes and Archer 1997; Brudvig and Asbjornsen 2009). As a fire-dependent ecosystem maintained through interactive disturbances such as fire and grazing, savannas are especially at risk for degradation as a result of fire suppression and over-stocking (Archer 1995; Scholes and Archer 1997; Archer et al. 2001; Langevelde et al. 2003).

In savannas around the world, reduced herbaceous cover and increased shrub layers are a result of woody encroachment—the of woody vegetation expanding into, and taking control of, the understory (Ratajczak et al. 2012). The causes, implications, and reversal of woody encroachment in savannas have been a management concern for years and while it's causes and mechanisms are well understood, its reversal is not (Fisher 1950; Briggs et al. 2005; Van Auken 2009; Pulido et al. 2010).

Savannas are transitional ecosystems that are inherently structurally heterogeneous, making it difficult to measure the effect of fire and grazing on woody encroachment. For this reason, it is critical to savanna restoration and management that monitoring focus on structural changes across the landscape to determine if restoration objectives are being met. Additionally, measuring savanna degradation and management effect using structural characteristics will increase our understanding of the effect of fire and grazing to reverse woody encroachment (Elzinga et al. 1998; Langevelde et al. 2003).

In a savanna, the accumulation of fine fuels in the herbaceous layer maintains the ecosystem structure by allowing frequent but low- to moderate-intensity fires to carry through the understory, which kill young, fire-sensitive woody vegetation and maintain the open, herbaceous understory. Under long-term fire suppression, the herbaceous community can be out-

competed and replaced by woody shrubs, which reduces fine fuel load and the potential for surface fires (Ratajczak et al. 2011). Luckily, even severely encroached savannas can respond to restoration efforts but require aggressive and continuous management to cross the functional and structural thresholds that would shift a landscape from woody to herbaceous cover (Archer 1995; Peterson and Reich 2001; Freeman and Jose 2009). Fire can effectively maintain savanna structure but should be used in conjunction with other treatments such as grazing to effectively combat woody encroachment (Archer 1995; Briggs et al. 2005; Asbjornsen et al. 2007; Brudvig and Asbjornsen 2009; Pelc et al. 2011; Brudvig et al. 2011).

Terrestrial laser scanning (TLS) is a high-resolution, ground-based remote sensing method that can provide accurate structural information on trees and individual shrubs (height, volume, biomass, and crown area), accurately estimate shrub biomass and model the vertical structure of savanna understory (Lefsky and McHale 2008; Vierling et al. 2013; Adams 2014; Richardson et al. 2014) (Chapter 2). This non-destructive sampling technique is economical as it gathers data more efficiently and with finer resolution than conventional sampling techniques, modelling a larger area in less time and from which several samples can be extracted. TLS has been used to accurately model vegetation structure and biomass of trees and woody shrubs in other ecosystems such as sagebrush steppe, forests, and peatland however it has not been applied in savanna (Lefsky and McHale 2008; Anderson et al. 2009; Dassot et al. 2011; Vierling et al. 2013). The inherent structural heterogeneity associated with savanna creates gradients of canopy and shrub density and distribution making it ideal testing ground for the field application of TLS. TLS excels at modelling structural forest and it offers managers the ability to quantify management effect in large-scale ecological projects and identify structural effects and trends resulting from management action.

We developed TLS methods to model vertical structure across four treatments in temperate savanna: burned, grazed, burned + grazed, and unburned + ungrazed. Fire and grazing have a complex relationship in savanna ecosystems and because of this, designing a treatment strategy to best achieve specific management goals can be complex. This is especially true when the degradation is severe and the transition from savanna into woodland may have started. While the effects of fire and grazing have been well-studied there are no studies that develop TLS methodology for modelling effects of treatments designed to reduce the shrub layer across a gradient of woody-encroached savanna. Understanding how the re-introduction of disturbance regimes affects the structure of savanna could inform management decisions with the primary goal of restoring an ecosystem (increasing native flora diversity, creating structural heterogeneity, reducing the woody shrub layer, and restoring the two-tiered structure characteristic of savanna) to support the greatest diversity of vegetation and wildlife populations.

Objectives of this study were to (1) investigate the structural response to treatments and (2) compare the relative effectiveness of fire, grazing and fire and grazing together in reducing woody understory. We hypothesized that (1) the structural response to treatments would be variable as the dominant woody shrub in this savanna aggressively responds to disturbance through resprout. We expected that fire-alone would be the least effective, grazing-alone would be moderately effective, and the combination of fire with grazing would be the most effective. We expect fire with grazing to be most effective as any regrowth would be trampled or eaten by cows and deplete the carbohydrate stores the dominant woody shrub uses to resprout, and (2) the relative effects of treatment would significantly change savanna structural characteristics across years, by reducing shrub biomass in the understory.

Materials and Methods

Study Area and Site Selection

This study was conducted during July and August, 20l5-2016 at Sherburne National Wildlife Refuge (SNWR) in Sherburne County, MN (lat 45.483951, long -93.711645). SNWR lies within the Mississippi Headwaters/Tallgrass Prairie ecosystem and is part of the Anoka Sandplain, situated in the transitional zone between tallgrass prairies to the West and boreal forests to the East (USFWS 2005). It has sandy soils, many wetland complexes and features 732 acres of a globally-endangered habitat type, Prairie-Oak Savanna that is characterized by old-growth Bur Oak (*Quercus macrocarpa*) stands, expansive lowland, and upland tallgrass prairie habitat (Leach and Ross 1995; USFWS 2005). All savanna habitat at SNWR is affected by woody encroachment, primarily composed of American Hazel (*Corylus americana),* that degrades savanna structure by increasing the shrub layer.

TLS: Modelling Vertical Structure

To model the vertical structure of savanna we used TLS as a non-destructive sampling technique at ninety-seven sites in two replicated blocks with a gradient of both understory and canopy density and height. At each site, several reference spheres, white spheres in various sizes that the scanner identifies and uses as markers to sew multiple scans together, were placed at varying heights and depths around the quadrat to facilitate rapid post-processing scan alignment (Figure 1.2). Then, four high-resolution, 360-degree scans were taken with a FARO Focus 3D S terrestrial laser scanner; each scan was 6 meters apart in a zig-zag pattern to cover a 9-m by 6-m area (Figure 1.2). This pattern was used to ensure scan overlap and to allow for the maximum potential of scan alignment, even at sites with dense or tall understories that could obscure

reference sphere detection. Each scan took approximately 8 minutes so even sites with thick, tall understories that were hard to navigate took less than 50 minutes.

We preprocessed and aligned the four scans using the artificial reference spheres that were placed in the field using FARO Scene 5.1.3 (FARO Technologies Inc. 2012). For each site, 5, 2-m^2 subsamples (n=485) extending from ground level to the maximum canopy height were exported from Scene as Ascii (.XYZ) files (Figure 1.3). All points within the exported column were summed in vertical 0.1m bins along the Z-axis using MATLAB 8.4 (The Mathworks Inc. 2014). In excel, we averages the point count for all subsamples collected in a plot to have one point that represents the sampled area for each treatment (n= 17). We then calculated the change in biomass between year 1 and year 2 for each treatment type.

Analysis

Comparison of Treatments Effects on Vegetative Biomass

We tested for a relationship between changes in vegetative biomass and treatment type with linear mixed-effect regression (LME) models using the lmer function in the lme4 package in the R Statistical Environment (Bates et al. 2015; R Core Team 2015). We sorted our sites into two disturbance groups, maintained and discontinued, that classified disturbance of both fire and grazing independently. Maintained sites had treatments in both years and discontinued sites were treated in year 1 but not year 2. LME models compared our response variable, the change in vegetative biomass from 2015 to 2016, to our fixed-effect predictor variables, treatment combinations in 2015 and 2016, using block as a random effect. We calculated the corrected Akaike's Information Criterion (AICc) for each model using the AICc function in the AICcmodavg package in R (Mazerolle 2016). We selected our best model by lowest AICc value and greatest Akaike weight after ranking our models with an AICc-based model comparison

using the aictab model selection function in the AICcmodavg package in R (Mazerolle 2016).

We calculated 95% confidence intervals for all parameters to model the relative effect of each

parameter using the confint function in the MASS package in the R Statistical Environment

(Venables and Ripley 2002; R Core Team 2015).

Results

We observed no significant relationship between the change in vegetative biomass and

treatment type (P = 0.43), but our model did explain thirty-one percent of the variation in our

data (R^2 = 0.31). In the pairwise comparisons, no variables were statistically significant

indicating that treatments did not affect vegetation biomass.

We tested a full-interaction and full additive model against our null model and AICc

model selection showed that the null model was the best with the lowest AICc value and 0.99

weight. Ultimately, our results indicate that there is no evidence that treatment had an effect on

understory biomass.

Our results indicate that there is no significant relationship between treatment type or

disturbance group (management discontinued or maintained from year 1 to year 2) and changes

in vegetative biomass (P = 0.55 and P = 0.07, respectively). Ultimately, this indicates that

changes in biomass are not affected by treatment combinations or a potential additive effect of

treatments occurring in both years versus a single year of treatment (Figure 3.1).
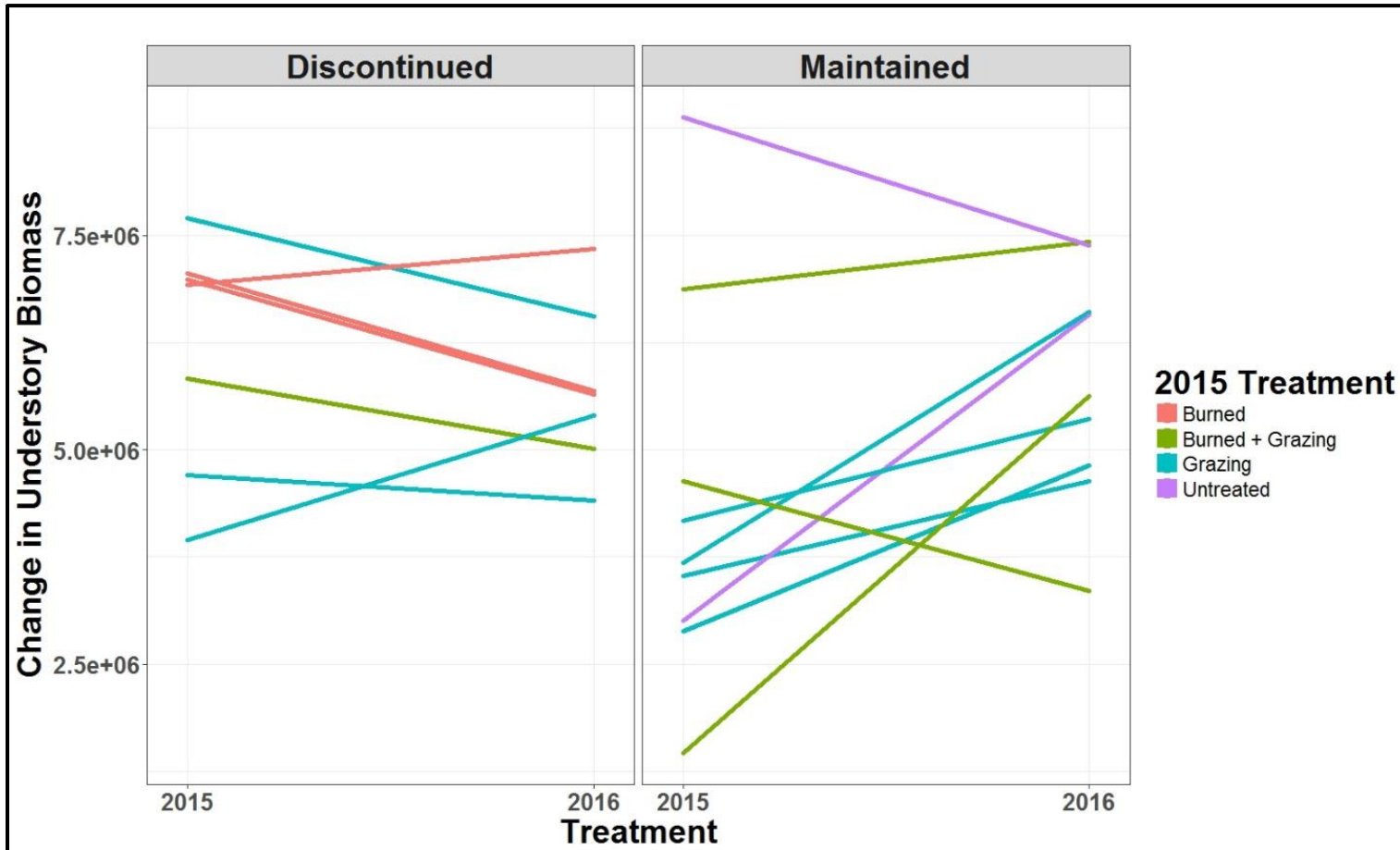
Figure 3.1: The variability in biomass among sites and the variability in response to treatments within and across disturbance types indicates (1) that TLS can estimate biomass across a wide range of structural variability and (2) that treatment did not have effect understory biomass.

Discussion

Savannas are disturbance-dependent ecosystems that require fire and grazing to maintain their species-rich, heterogeneous mosaic landscape and previous research has established that frequent low-intensity fires coupled with moderate grazing can reduce or control woody encroachment in these landscapes (Collins and Wallace 1990; Archer et al. 2001; Fuhlendorf and Engle 2004; Simmons et al. 2007; Ratajczak et al. 2012). This relationship is well-documented yet we did not observe it in our own study and we suspect it is due to (1) the short duration of our study and (2) growth characteristics of the dominant understory species in our study savanna. Although we saw no treatment effect on the understory shrub layer, this study is valuable because it uses TLS in a new ecosystem, develops a non-destructive field sampling technique, and is a demonstration that TLS is a viable monitoring tool that can collect fine-scale data across a gradient of ecological degradation and structural characteristics. Ultimately, the methodology presented here is novel and could be applied in other biomes to accurately estimate the biomass of a woody understory and structural forest variables.

We saw no evidence of treatment on the woody understory and we attribute this to the disturbance-adapted growth characteristics of the dominant woody-encroacher at our sites, American Hazel. This native species is resilient and responds vigorously following disturbance events by extending multiple rhizomatous daughter stems and increasing its patch size within the ecosystem (Ratajczak et al. 2011). It typically takes repeated disturbances, sometimes multiple disturbances in one growing season, to kill this hardy species by sufficiently depleting the glycogen stores it uses to fuel its regrowth (Buckman 1964). We suspect our insignificant results are due to the short duration of our study and the disturbance-adapted, aggressive nature of the

dominant woody shrub at our sites; a long-term study would likely be better suited to observing the effect of these newly-implemented disturbance events on woody-encroached savanna.

Current management strategies for oak savanna often include reintroducing fire regimes and grazing disturbances on the landscape but, as a direct result of its aggressive rhizomatous stems, the effectiveness of such strategies on *C. americana* is difficult to quantify. One study found that although the stem density of *C. americana* increased following disturbance the regrowth was shorter and smaller (1978) ; structural characteristics are critical to savanna and measuring changes in anything else could lead to a misinterpretation of management effect. Our regression shows the variability that is present, even in a degraded savanna, and the differences in the initial vegetation response one year post-treatment; exhibiting the usefulness of TLS to model the same fine-scale structural variability as seen by Axelrod and Irving (Axelrod and Irving 1978) (Figure 3.1). Ultimately, our results suggest that there is no significant short-term effect of fire and grazing on a savanna landscape and this aligns with past research establishing that long-term and consistent management is required to restore a degraded savanna landscape (Archer 1995; Peterson and Reich 2001; Freeman and Jose 2009).

Using TLS to measure changes like these simplifies and expedites the field sampling process as both fire and grazing occur on a large scale and TLS samples more with less effort and greater precision. Managers worldwide face the practical limitations of time and funding so require an economical and effective tool like TLS that can monitor management effect on vegetation structure. This is especially relevant for ecosystems facing similar problems to savanna where restoration results mean quantifying changes across an entire landscape or where perceived results can be confounded by disturbance-adapted responses.

We successfully used TLS to estimate changes in biomass and to model structural variability among degraded savanna sites (Figure 3.1). This technology could be an essential tool to guide management decisions on a qualitative level and our results, combined with those of past studies, show that TLS can accurately gather detailed structural information on individual shrubs and trees across sites with diverse structural characteristics (Lefsky and McHale 2008; Yao et al. 2011; Olsoy 2013; Adams 2014; Olsoy et al. 2014). Additionally, a greater amount of data is collected in less time compared to conventional techniques; further justifying its value to field researchers and managers monitoring at a landscape-scale level. In the short amount of time it took to sample one non-destructive plot we collected accurate data on a 20-m^2 area and exported 5, 2-m^2 subsamples from that data. Future applications of TLS should investigate its ability to characterize fuel beds, estimate non-woody biomass, model fine-scale plant characteristics such as flowers, or quantify changes in the spatial and vertical distribution of vegetation resulting from ecosystem management.

Conclusion

Although we did not find significant relationships in our models we did successfully model variability in savanna site characteristics and shrub density using TLS as a non-destructive sampling technique. TLS successfully characterized changes in biomass and should be used as a tool to monitor structural changes that result from management action in long-term restoration projects. We suspect that our inconclusive results were a result of the short duration of this study and growth characteristics of woody vegetation that can stave off treatment effect for a couple years before being negatively affected. Ultimately, our sites may have been too degraded and dominated by woody shrubs to carry fires with enough intensity to kill woody species and we suspect that in subsequent years the effects of treatment on these hardy plants will be observed.

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