RESTORATION OF MASSASAUGA RATTLESNAKE (*SISTRURUS C. CATENATUS*) OVERWINTERING HABITAT: DESIGN, CONSTRUCTION AND ECOHYDROLOGICAL ASSESSMENT RESTORATION OF MASSASAUGA RATTLESNAKE (*SISTRURUS C. CATENATUS*) OVERWINTERING HABITAT: DESIGN, CONSTRUCTION AND ECOHYDROLOGICAL ASSESSMENT

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A Thesis Submitted to the School of Graduate Studies in Partial Fulfillment of the Requirements

for the Degree of Master of Science

(School of Earth, Environment & Society)

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OF PAGES: 123

ABSTRACT

The eastern massasauga rattlesnake, or massasauga, is threatened in Ontario. Massasaugas overwinter in habitat that is sufficiently moist, above the water table, and below the zero-degree isotherm in a physical space conceptualized as a resilience zone. A development project in the Eastern Georgian Bay subpopulation of massasaugas overwintering habitat necessitated restoration. The Toronto Zoo hibernacula design was deemed inappropriate for use in this bedrock dominated landscape, as the limited deep groundwater percolation would result in flooding of the habitat from the fall to spring. Massasaugas in EGB, overwinter above the water table in Sphagnum peat hummocks. The ecohydrological factors of these hummocks in confirmed massasauga habitat was evaluated in a mixed effect linear model. It was found that taller hummocks, taller shrubs, and less WT variability were the best predictor of suitable conditions. This information was combined into a restoration method that moves surficial peat material from a soon to be impacted wetlands to an adjacent depression with mean depth of 40-80 cm, with mean bottom substrates 15-30 cm, and varying proportions of open water and floating peat with different vegetation communities. This design limits water table variability and allows for the growth of tall shrubs. Unforeseen circumstances meant that peat to be used in the project had to be stockpiled, which increased peat bulk density and limited photosynthesis. Despite this, restored habitat had similar mean lengths of unsuitable conditions compared to confirmed massasauga wetland habitat. The physical size of available overwintering habitat, as well as the total duration of unsuitable conditions was not significantly different between restored wetlands, unconfirmed wetlands, and confirmed wetlands. Amendments to increase cover of live Sphagnum moss will likely increase duration and size of suitable conditions in the restored wetlands. Based on this success with degraded materials this new method of restoration design shows great promise in this region.

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ACKNOWLEDGEMENTS

I would like to thank my supervisor Mike Waddington, for his support during my time with the Mac Ecohydrology lab. I am thankful for his flexibility and guidance as a supervisor, especially with the writing process. I would like to thank Chantel Markle for all her herpetofauna fixing, and finessing. Your expert knowledge has been invaluable, and I am excited to see all you accomplish. To Paul Moore for keeping all my data analysis in perspective, for the MATLAB work and for letting me know when I am looking at upside-down hydrographs. To Sophie Wilkinson for help with all things peat properties and for being a good mentor and friend.

This work would not have been possible without the generous help of some individuals. Firstly, Keegan Smith for sledgehammering frozen peat with me in multiple snowstorms. To Ian Martin, Greg Verkaik, Paul Moore for help setting up the restored site monitoring station, and to everyone that helped in the instrumentation of the confirmed and unconfirmed sites before I was ever here. I am grateful for Alex Tekatch, Hope Freeman, Taylor North, Greg Verkaik, Alex Furukawa, Danielle Hudson, and Madeleine Hayes for help with vegetation surveys, and site downloads. It has been so nice getting to know you all. I am sure you all have many bright things ahead.

To my many different roommates I am thankful for you much needed distraction, and support. To my close friends and family thank you keeping a level head on my shoulder. Finally, to my partner Waverley Birch thank you for being a continuing source of joy and support.

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LIST OF ABBREVIATIONS

Corrected Akaike Information Criteria	AICc
Analysis of variance	ANOVA
Catchment to Wetland Ratio	CWR
Eastern Georgian Bay	EGB
Ecological Land Classification	ELC
Factor Analysis	FA
Gravimetric Water Content	GWC
Hydraulic Conductivity	K _{sat}
Non-metric Multidimensional Scaling	NMDS
Principal Component Analysis	PCA
Polyvinyl Chloride	PVC
Restricted maximum likelihood	REML
Resilience Zone	RZ
Species at Risk	SAR
Variable Inflation Factor	VIF
Volumetric Water Content	VWC
Volumetric water content at 100 hPa	VWC-100
Water Table	WT

DECLARATION OF ACADEMIC ACHIEVEMENT

The written material contained in this thesis dissertation has been prepared solely by this author. The research design is a result of collaborative efforts of the McMaster Ecohydrology Lab and the Global Water Future technicians that collaborated to install weather monitoring instrumentation. The design of the overwintering habitat was co-developed by Dr. Mike Waddington, Dr. Chantel Markle, and this author. Contributions of healthy nearby peat properties data came from Ben Didemus. Dr. Paul Moore was instrumental in assisting with linear interpolation of thermocouples to determine the depth of the 0 degree isoline, Dr. Chantel Markle was instrumental in the utilization of linear mixed effects models, Dr. Sophie Wilkinson contributed to the interpretation peat property data, and Dr. Mike Waddington was instrumental in directing research design. Greg Virkeik, Keegan Smith, Ian Martin, Alex Furukawa, Alex Tekatch, Danielle Hudson, Hope Freeman, and Madeleine Hayes were essential in assisting with installation and monitoring of weather monitoring equipment, and the collection of vegetation and peat property data.

CHAPTER 1: INTRODUCTION

1.1 Threats to the eastern massasauga rattlesnake

The eastern massasauga rattlesnake (*Sistrurus catenatus*, hereafter referred to as massasauga), is a shy and reclusive venomous snake species. It is a threatened or endangered species at provincial/state or federal level across most of its range in the Great Lakes region of Canada and the United States. The subpopulation of massasaugas within Eastern Georgian Bay (EGB) in Canada is classified as Threatened through Schedule 1 of the federal *Species at Risk Act* (SARA, 2002), and listed as Threatened under the *Endangered Species Act* (ESA, 2007), through *Ontario Regulation 230/08: Species at Risk in Ontario List* (current to August 1, 2018). According to the Endangered Species Act (2007) habitat features associated with overwintering and gestation of massasaugas have the lowest tolerance to alteration because of the importance of these habitats for species survival. Massasaugas have demonstrated strong site fidelity to overwintering habitat, with most individuals overwintering within 100 m of previous years overwintering locations (Harvey and Weatherhead, 2006b; Yagi et al., 2020).

It is anticipated that the suitability of massasauga overwintering habitat may be at risk in the future as warm winters and inconsistent snow cover at mid latitudes may expose massasaugas to temperature extremes (Williams et al., 2015). Additionally, changes in precipitation type, intensity, frequency, and duration expected with climate change have the potential to flood overwintering habitat (Markle et al., 2020a). This threat from climate change is compounded by threats to this species that include habitat loss and fragmentation, road mortality, and persecution (Pomara et al., 2014). A development project occured in the EGB region that directly destroyed existing overwintering habitat. The developer was obliged, under section 73 of SARA, to offset the loss in overwintering habitat through the construction of restored habitat. Since characteristics of overwintering habitat vary for different subpopulations of massasaugas (Harvey and Weatherhead, 2006a; Smolarz et al., 2018; Markle et al., 2020a; Yagi et al., 2020), there is a need to understand the characteristics of overwintering habitat for this subpopulation in order to design the restored habitat to be resilient against current and future threats.

1.2 Microhabitat requirements

Poikilothermy, or variable internal temperature regulation, allows snakes to have a narrow form that is suited for ecological functioning, like hunting in mammal burrows (Shrine, 2013). The body temperatures of snakes are nearly identical to a profile of soil temperatures (Brown et al., 1982).. Therefore, massasauga overwintering thermoregulation must occur through habitat selection (Hermes-Lima and Storey, 1993)

Massasaugas overwinter in subsurface microhabitat features that remain above the water table (WT) (Smith, 2009; OMNRF, 2016a; Markle et al., 2020a; Yagi et al., 2020), have sufficient moisture to prevent dessication (Costanzo, 1989), have access to aerial oxygen, and have stable cool soil temperatures that remain above 0 ^oC (Johnson, 1995; Smith, 2009; Parks Canada Agency, 2015). The microhabitat should have physical space for massasaugas to adjust to flooding and temperature fluctuations throughout the overwintering period (Sexton and Marion, 1981). Different subpopulations of massasaugas will meet these microhabitat needs based on the habitat features that are available within the landscape. These landscape features can be deep rock crevices (Harvey and Weatherhead, 2006a), crawfish and mammal burrows (Weatherhead and Prior, 1992),

rotting tree roots (Johnson, 1995), and raised Sphagnum (peat moss) hummocks found in wetlands (Parks Canada Agency, 2015; Smolarz et al., 2018). These raised hummocks are used by the EGB subpopulation of massasaugas. The area within the hummock that remains unflooded and above 0 ⁰C isotherm is referred to as the resilience zone (RZ) (Smolarz et al., 2018). Smolarz et al. (2018) found that taller Sphagnum hummocks maintain RZ longer than shorter hummocks, because these taller hummocks are able to buffer against flooding and the descent of sub 0 °C conditions during the overwintering period. Field evidence from overwintering sites (Markle et al., 2020a; Yagi et al., 2020) has shown that massasaugas might have a short-term tolerance of no more than a few hours of anoxia and sub zero conditions in overwintering habitat. However, a mass mortality event observed in the winter of 2014-2015, that occurred either before the first snowfall or after snowmelt (Ron Black and Mike Colley, personal communication, 2016), shows that overwintering habitat used by massasaugas cannot always provide suitable conditions. Therefore, the overwintering conditions and mortality in a created or restored habitat should always be compared against confirmed overwintering habitat. If unsuitable conditions exist to a similar or lesser degree in the restored habitat than in the confirmed habitat, then the restored habitat met its goals.

Artificial overwintering habitat has been designed and monitored for snake species in different landscapes in Ontario (Toronto Zoo, 2015). This artificial habitat is a mixture of concrete blocks, gravel, stone, and sand in excavated holes in mineral soil (Toronto Zoo, 2015). Little is published about the effectiveness of these structures, but eastern foxsnakes did successfully overwinter at Point Pelee, Ontario (T. Dobbie, personal communication, 2016), and Windsor, Ontario (OMNRF, 2018). This outlined methodology of habitat creation is not appropriate for massasaugas in the eastern Georgian Bay region (OMNRF, 2018). Such a design would be difficult to implement on a large scale due to the difficulties associated with removing bedrock (likely through blasting) at

a larger scale in this landscape. More importantly, the shallow soil depth and impermeable bedrock in the EGB region means there is limited deep groundwater percolation in this landscape. Therefore, any excavated hole will fill with water by the start of winter and during spring melt conditions, and potentially flood the overwintering habitat. Furthermore, the habitat will not resemble the overwintering habitat that the Eastern Georgian Bay (EGB) subpopulation of massasaugas is accustomed to utilizing. As such, there is a need to understand characteristics of natural overwintering habitat in the EGB landscape to design habitat that provides this function in this unique landscape.

1.3 Thesis objectives

It is known that taller hummocks provide more suitable conditions than shorter hummocks (Smolarz et al., 2018). However, hummocks of similar heights occurring in either occupied or unoccupied sites provide different mean heights of suitable resilience zone (RZ) space, for different durations of time over the winter (Markle et al., 2020a). Therefore, there are likely some other micro- or meso-scale variations that explain differences in RZ space and duration between locations. The overarching goal of the thesis is to understand these variations to create and assess the performance of restored massasauga overwintering habitat. Chapter two focuses on how ecohydrological factors (*e.g.* peat depth, water table variability, shrub height etc.) affect peatland WT and peat temperature dynamics, and how these relate to total and mean duration of RZ loss. This includes factors known to affect average peat temperature, such as snow depth and duration (Nichols, 1998; Williams et al., 2015), water table depth (WTD) (Knox et al., 2012; Nagare et al., 2012), and shrub height and cover (Pomeroy et al., 2006). Factors known to affect WT position are also investigated, such as peat depth, the ratio of the catchment area to wetland area, and

hydroperiod (Waddington et al., 2015). Based on the results from chapter two, chapter three presents a design for suitable peatland overwintering habitat for massasaugas. This chapter includes a post-restoration assessment of overwintering habitat using the mean and total duration of RZ loss as a performance metric, and prediction of wetland succession from vegetation surveys. Both the method of restoration and the use of ecohydrological determinants of restoration success are crucial tools that restoration practitioners can use for habitat-focused restoration.

CHAPTER 2: CHARACTERIZING OVERWINTERING HABITAT 2.1 Introduction: Ecohydrological factors related to suitable overwintering habitat

2.1.1 Context for restoration

A development project is occurring in the Eastern Georgian Bay (EGB) region that will disrupt overwintering habitat for the Georgian bay population of eastern massasauga rattlesnake *Sistrurus catenatus* (hereafter referred to as massasaugas). There is a need to offset the loss of overwintering habitat for this threatened species. Massasaugas in the EGB subpopulation overwinter in *Sphagnum sp.* hummocks in peatlands that provide access to sufficient moisture (Costanzo, 1989), aerial oxygen (Smith, 2009; OMNRF, 2016a), and stable above 0 °C temperature (Gregory, 1982; MaCartney et al., 1989). It is known that taller hummocks provide a larger resilience zone (RZ), or space between the bottom of the 0 °C isotherm and the top of the water table (WT) (Smolarz et al., 2018). However, similar hummock heights are associated with different sizes of RZ and durations of RZ loss (Smolarz et al., 2018; Markle et al., 2020a). Therefore, there is a need to understand what other micro and meso-scale factors are associated with confirmed overwintering habitat in order to build suitable habitat.

2.1.2 Fill and spill hydrology

Protection from flooding events is necessary for massasaugas overwintering habitat suitability (Smith, 2009; OMNRF, 2016a; Markle et al., 2020a; Yagi et al., 2020). This can be met with a near-surface water table (WT) if the WT fluctuations do not lead to flooding. Wetlands in the EGB landscape often avoid flooding conditions by shedding excess water. The process of water partitioning completely into storage or runoff is referred to as the fill and spill hydrology (Spence and Woo, 2003). This "spilling", or runoff, occurs when the WT exceeds the storage capacity of the basin, which is determined by the volume of the basin (Brooks and Hayashi, 2002). This

spilling occurs at the lowest impermeable edge along the perimeter in the wetland, which is referred to as the sill. If this sill develops as a narrow gully that only represents a small ratio of the total circumference of the total wetland, then this wetland, which might contain massasaugas, could be vulnerable to winter flooding from snow or ice jams (Markle et al., 2020a). By contrast, a sill that represents a larger proportion of the perimeter of the wetland is less likely to flood massasaugas overwintering habitat and should always spill at the same WTD. Therefore, when restoring massasaugas habitat, a wide sill must be used. Additionally, the top of restored hummocks should be sufficiently above the level of the sill, so that the maximal frost depth does not reach the position of maximal WT rise.

The position of the WT will affect the potential for spring infiltration (Nagare et al., 2012). A nearsurface WT can form an impermeable ice peat cap in late winter (Putkonen and Roe, 2003; Van Huizen et al., 2020). During snow melt, this seasonal ground ice will be closer to the peatland surface in peatland hollows, but further beneath the peatland surface in the hummocks (Nagare et al., 2012). This results in a variable landscape of infiltration capability, where hummocks have a greater early spring infiltration capacity than hollows (Nagare et al., 2012). At the wetland scale, spring infiltration of meltwater and rain is greater at the beginning of winter in sites with water table depth (WTD) and lower hummock moisture values.

A larger WTD will result in a flashier WT fluctuation through the specific yield water table feedback (Waddington et al., 2015). Specific yield represents the water released from storage, per area of wetland, per unit decline of WT (Johnson 1967). When the WTD is greater, the WT will rise more for an equivalent addition of water compared to a smaller WTD. Similarly, denser peat found in degraded peatlands or peatland margins have flashier WT fluctuations for wetting and drying events. The recession of the WT is moderated in peatlands that have sustained groundwater

inputs between precipitation events, compared to wetlands that do not (Devito et al, 1996; Branfireun and Roulet, 1998).

Didemus (2016) observed that sustained inflows to wetlands with a larger catchment to wetland ratio (CWR) moderated the WT recession between precipitation events in the EGB rock barrens landscape. This effect was most apparent in shallower wetlands, and during wetter periods when the WTD was shallower. The thin soil and small catchment areas in the rock-barren landscape mean there is little time between the end of a rainfall event and the end of a runoff response (Buttle and Eimers, 2009; McNarma et al., 1998). When EGB rock barrens dominated catchments are sufficiently saturated, runoff occurs to a lower topographic position.

2.1.3 Peat hydrology

Massasaugas in the Eastern Georgian Bay (EGB) region are accustomed to overwinter in areas with more stable WT fluctuations compared to some of the Carolinian populations in southern Ontario (Yagi et al., 2020). There are depth-dependent differences in peat properties that affect the stability of the WT (Beckwith et al., 2003). Didemus (2016) found that wetlands in the EGB region had a WT present 100% of the time if the average peat depth was greater than 50 cm, but only 29 - 46% of the time if wetlands had an average peat depth less than 40 cm. Shallow wetlands (<40 cm) have low drainable porosity or specific yield (Sy) values. This low Sy will make WT fluctuations more pronounced through time, and act to reduce Sy through the WT-Sy feedback (Waddington et al., 2015). In addition, a long hydroperiod in deep wetlands results in longer pore-water residence times in the autochthonous and humified deeper peat (catotelm) via the pore-water feedback (Furukawa, 2018). This is because the composition of dissolved organic matter prevents further decomposition of deeper peat and makes these wetlands resilient against disturbance. Since the pore-water remains largely stagnant in the waterlogged conditions, the less energetically

favorable end products of decay accumulate and thermodynamically suppress anaerobic decay (Beer and Blodau, 2007; Beer et al., 2008; Morris and Waddington, 2011). Therefore, restoring peatlands to have average depths that are slightly above 50 cm, allows for the longer-term resilience of peatlands, while minimizing the volume of material that must be contributed. Deeper wetlands have pore-water transport that is typically limited by low values of hydraulic conductivity (K_{sat}) within deeper peat layers (Beckwith et al., 2003) and margin peat (Baird et al. 2010). These represent water conservation strategies to minimize water losses. For wetlands that occur over a permeable substrate or fractured bedrock, the K_{sat} of the peat mineral substrate affects the vertical movement of water into or out of the peatland. In four wetlands near the proposed disturbance area in EGB region, the K_{sat} at the peat-mineral soil interface was found to be 1 x 10-6 m s⁻¹ (Vu, 2019). Given that these wetlands occur over fractured granite, there would be some vertical movement of water into and out of the peatland, in addition to water loss through other routes. A restoration practitioner would have to determine whether the WT within the restored peatland was perched and if the movement of water in and out of the peatland would lead to an unstable WT. They would then have to determine whether to use a low K_{sat} mineral organic mix to allow for some movement of water into and out of the peatland, or an impervious concrete liner, or a heavily compressed clay liner at the bottom of the basin to restrict water movement.

2.2.4 Snow

Snow cover provides a subnivean refugia (space below the snow) that provides overwintering organisms like massasaugas a thermally stable environment, relative to winter air temperature. The thermal buffering of the snowpack creates such effective refugia that subpopulations from northern and snowy environments can have less cold tolerance than those from southern regions with shallower and less persistent snowpacks (Williams et al., 2015). The EGB region receives an

average of over 327 cm of snow annually, which represents one-third of the 1090 mm of annual precipitation, assuming a snow water equivalent of 1 kg m² (Environment Canada, 2019). It is expected that climate change will decrease the ratio of precipitation falling as snow, decreasing the subnivean duration from 126 days to 110 days, and increasing the duration of frozen ground without snow cover from 35 days to 45 days at mid-latitudes (Zhu et al., 2019).

The absence of an early snowpack can have a large effect on surface peat temperatures over the entire winter. In a study of six peatlands in Minnesota, Nichols (1998) found over six winters that frost reached a depth of 15 - 25 cm by late February if there was less than 10 cm of snow cover in December. However, when snow depth was greater than 20 cm by the end of December, frost depths were between 3-8 cm by late February (Nichols, 1998). This aligns with the 30 years of frost records made at the Marcell research station by Verry (1991), who found that the end of season frost depth was twice as deep in years with half the amount of snow cover in late December. Ling and Zhang (2003) found in Utqiagvik (Barrow) Alaska, that when snow accumulation was delayed by 10 days, the mean annual ground temperature decreased 0.7 °C, and ground temperatures increased 6.6 °C, and mean annual ground temperature increased 0.2 °C. Given this interannual stochasticity in snowfall, restored overwintering sites must be designed to provide a resilience zone (RZ) that considers a climatic future with less consistent and abundant snow cover.

Even if snow remains present throughout the winter, organisms can be affected by snow densification, accelerated by mid-winter melt and rain on snow events (Williams et al., 2015). The thermal conductivity and thus insulative properties of the snowpack are inversely proportional to the square of snow density (Aggarwal et al., 2009), which increases from metamorphosis caused by rain and melt events. This loss in insulation is particularly problematic when compounded by

the increase in WT that occurs from the combination of rain and melt-water, which reduces the RZ size within a hummock (Markle et al., 2020a). Moreover, the depth and density of a snowpack determine its insulative potential and is spatially variable over small horizontal distances. Sturm and Holgrem (1994), found that snow depth measurements taken only 50 cm apart could vary from 10 - 35 cm because of the influence of the underlying microtopography. These authors found this variation could change the snow ground temperature by more than $10 \,^\circ$ C. Therefore, understanding snow accumulation patterns among wetlands and in locations within them will allow restored sites to be designed such that they accumulate and maintain consistent snow cover.

2.2.5 Vascular vegetation

Greater snow cover has been associated with greater shrub cover in the subarctic (Pomeroy et al., 2006). Several researchers, including Menard et al. (1998), Paradis et al. (2016), and Pomeroy et al. (2006) have observed a connection between shrub expansion and increases in wintertime soil temperature. Sturm (2005) associated positive feedback between soil temperature and snow depth under shrub canopies. This resulted from snow drifting against taller shrubs, and a decrease in the albedo of the surface cover. Paradis et al. (2016) found a difference of 10 °C, where ground cover beneath shrub cover was compared to bare cover. While these studies occurred further north than the EGB region, all locations have intermittent shrub cover and open areas that lead to patterns of snow detachment and accumulation.

The observation of warmer soil temperatures corresponding to greater shrub cover (Menard et al., 1998; Paradis et al., 2016) could help to explain the tendency for massasaugas to overwinter in hummocks beneath shrub cover (Johnson, 1995; Yagi et al., 2020; Markle et al., 2020a). massasaugas have also been found to overwinter in sedge dominated hummocks (Smolarz et al., 2018). To get a better understanding of the association between massasaugas observations and

wetland characteristics, Markle (2019) used linear discriminant analysis to compare a variety of characteristics on 151 wetlands in the EGB region. Markle (2019) found that shrub and tree cover and simpler spatial complexity were associated with wetlands that had spring emergence records of massasaugas.

Although increased shrub cover does lead to warmer soil temperature, this can lead to earlier snowmelt in the spring. Earlier snowmelt might expose massasaugas to snow-free conditions putting them at risk from flooding and freezing. Taller shrubs emerge from the snowpack earlier in the spring and lower the albedo of the snow surface sooner than an unvegetated surface (Pomeroy et al., 2006). Areas with more shrubs can have earlier snowmelt due to increases in radiative and turbulent energy transfers to the snowpack, which would be greater than the reduction in short-wave radiation beneath the shrub canopy (Pomeroy et al., 2006). These same processes can enhance the emission of longwave radiation, and the reflection of shortwave radiation between the tree trunk and snow well (Romanov, 2013). Early snowmelt poses a flooding risk to massasaugas overwintering habitat, while simultaneously decreasing insulation potential.

Later snowmelt tends to occur where shortwave radiation from increased canopy cover offsets increases in longwave radiation, leading to soil cooling (Schelker et al., 2013). Within boreal forests, Metcalfe and Buttle (1998), and Sicart et al. (2004) found that the reduction in shortwave radiation beneath the tree canopy more than made up for increases in emitted longwave radiation. The resulting delay in snowmelt increases the possibility that massasaugas will egress when ambient temperatures remain above 0 °C. The trade-offs with greater canopy closure are the potential loss of total snow accumulation because of interception (Hedstrom and Pomeroy, 1998) and reduction in basking opportunities that are particularly critical for gravid massasaugas habitat (Johnson, 1995). For this reason, if restoration design of overwintering massasaugas habitat

incorporates tree cover, the practitioner would need to incorporate a spatially complex vegetative composition (Markle et al., 2020b), including variable canopy closure and close proximity to open rock to allow for snow accumulation and basking habitat.

2.2.6 Moisture

Snowpack is effective at maintaining heat within wetlands that can continuously provide a sizable ground heat flux (Knox et al., 2012). This flux occurs as the water in the wetland represents a large thermal mass that can remain largely unfrozen provided an early, deep, and consistent snowpack. The large specific heat capacity of liquid water (4186 J kg⁻¹ K⁻¹) means that a large body of water is resistant to temperature change. In addition, since the enthalpy of fusion of water is 3.33×10^5 J kg⁻¹, a large amount of heat is released to the surrounding peat upon freezing. This slows the propagation of ice from the surface. Early season freezing of the soil surface can represent as much as 50% of the energy balance of the snowpack (Sensoy et al., 2006). In a Quebec peatland, Knox et al. (2012) found that the ground heat flux from the unfrozen peatland could be as much as 25% of the snowpack energy balance during the snow accumulation period.

Near-surface WT and high porosity (0.94 to 0.98 in the top 20 cm) of a healthy peatland (Redding and Devito, 2006) allow for considerable gravimetric water content (GWC) and makes peatlands effective at reducing the rate of frost propagation (Knox et al., 2012). Nagare et al. (2012) found that latent heat controls the initial rate of frost propagation until the higher thermal conductivity and lower heat capacity of ice allows for a faster propagation of frost. However, given a typical snow cover and moderate winter air temperatures, it is likely that latent heat will remain the dominant control on frost depth (Knox et al., 2012). When Bueno et al. (2016) removed the top 5 cm moss layer beneath shrubs they found significantly lower winter soil temperatures compared to the intact treatment. Similarly, drained peatlands with greater bulk densities and lower GWC do not have the same buffering capacity to resist the propagation of soil frost (Price, 1996). For this reason Price (1996) and Eskeling et al. (2016) both found colder peat temperatures and longer frost persistence in drained peatland sites compared to control peatland sites. Therefore peat must remain in wet depressions to maintain high porosity and moisture content of the peat material.

In some systems, taller Sphagnum hummocks maintain suitable conditions for massasaugas overwintering longer than shorter hummocks (Smolarz et al., 2018). However, hummocks of similar heights were also found to provide different sizes of suitable overwintering space or resilience zone (RZ) over the entire winter (Smolarz et al., 2018; Markle et al., 2020a). Therefore, there are likely additional micro- or meso-scale variations that contribute to the differences in RZ between these sites. By measuring WT and temperature fluctuations at multiple hummocks in multiple wetlands, one compare ecohydrological variables to conditions that are suitable for massasauga overwintering. This allows for an understanding of the variables that increase the RZ or the physical space between the WT and the 0-degree isotherm during the overwintering period (Smolarz et al., 2018). It is hypothesized that some combination of the height and cover of shrubs; the difference in canopy closure; CWR, peat depth, mean snow depth, snow duration, hummock height, and WT variability will be related to mean and total RZ loss better than a model that only uses hummock height. It is also hypothesized that these ecohydrological variables will vary more at the wetland scale than between hummocks. This understanding will help direct the design of overwintering habitat outlined in Chapter 3.

2.2 Methods

2.2.1 Site characteristics

All seven sites selected for monitoring in this chapter are located along the eastern shoreline of Georgian Bay within 10 km of open water. This is a landscape of rock barrens, forested valleys, wetlands and lakes, and is differentiated from other regions in central Ontario by the gently rolling ridges of gneissic shield bedrock (Crins et al., 2009). All the seven wetlands in this chapter are formed in peat dominated bedrock depressions, and range in size from 0.37 - 1.24 ha in size with peat depths 0.5 m - over 2 m. These wetlands are either dominated by ericaceous shrubs (*Ericaceae*), or sedges (*Carex*) and can be classified as sedge or shrub dominated poor fens, shrub dominated bogs, or coniferous organic swamp according to criteria in the Ecological Land Classification (ELC) system (Lee et al., 1998).

Three of the seven wetlands are assumed to be confirmed massasauga overwintering as they were confirmed through visual observations (referred to as C3) or were within 1 km of massasaugas observations, were suggested by an experienced wildlife biologist and had vegetation, and hummock height similar to confirmed overwintering wetlands (referred to as C1, and C2). These are referred to as confirmed sites and are assigned the names C1, C2, and C3. Conversely, wetlands that were unconfirmed overwintering habitat did not have massasauga observations. These are assigned the names U1, U2, U3, and U4.

2.2.2 Thermal and hydrological data

The sites C1, C2, C3, U1, U2, U3, U4 were instrumented to collect continuous micrometeorological, temperature and WT data for the winters of 2018-2019 and 2019-2020. WT depth was measured every 15 minutes using Solinst Levellogger Junior pressure transducers (Solinst, Georgetown, ON). Loggers were inserted in a 5 cm diameter groundwater monitoring

well and attached by galvanized steel wire. We corrected for barometric pressure with a Solinst Edge Barometric pressure logger. This barometric pressure correction, the height of the well above the peat surface, and the string length were used to reference WT depth measurements to the moss surface. WT variability was calculated over the October 30 -April 15 overwintering by calculating the standard deviation of the time series in R 3.6.1.

At each study wetland, five *Sphagnum* hummocks were instrumented with Campbell Scientific CS616 moisture probes (Campbell Scientific, Inc., Logan, UT). These 30 cm probes were installed horizontally in hummocks at a depth of 5 cm from the surface and vertically into the hummock. In the same hummocks, a profile of type T thermocouple wire with an accuracy of 0.5 °C (Omega Engineering, CT, USA) was inserted at depth of 1, 5, 10, 15, 25, and 50 cm from the hummock surface. These were set to log temperature average every 30 minutes based on the average temperature of three measurements during this period.

Low pass Gaussian filtering was performed in R.3.6.1 with the "smoother" package, which reduced high frequency noise in thermocouple measurements. Temperatures were interpolated between the 6 thermocouple peat depths to extract the depth of the -0.25-degree isotherm was extracted in MATLAB 9.6. The -0.25-degree isoline was chosen to reduce the noise associated with meltwater and rain on snow that occurs with calculating the 0-degree isotherm. Mean RZ loss and total RZ loss was calculated as the mean and total duration of time that the isotherm depth was beneath the WT for each surveyed hummock location during the overwintering period. A Winter is defined as October 15 to May 15th as this roughly approximates Massasauga ingress and egress (Harvey and Weatherhead, 2006b)

2.2.3 Vegetation surveys

In late July 2019, vegetation surveys were conducted in ten 1x1 m quadrat grids at C3, C2,C1, and U4. The same methodology was used for vegetation surveys conducted at sites U1, U2, and U4 in July of 2018. The location of each sampling quadrat was haphazardly selected. For each vascular species encountered, the scientific name of the species; number of individuals; approximate cover as a portion of the vascular canopy; and height (cm) of 5 randomly selected individuals was recorded. If the number of individuals exceeded 100, the number of individuals was approximated. If the identity of a species could not be confirmed, a sample name was assigned and a representative sample was taken from site. These samples were later identified and properly processed as herbarium samples. Bryophyte scientific name was recorded, and cover (%) was approximated within the grid cell.

Additional vegetation surveys were completed at instrumented hummock locations to determine the shrub cover (%) and height (cm), canopy cover (%), and hummock height (cm) relative to adjacent hollow. Shrub cover was the proportion of shrubs within a 1 m quadrat area. Shrub was taken as the mean height of all woody shrub vegetation within the 1 m quadrat area. Canopy cover was calculated using standard non hemispherical photographs that were taken from 1.3 m above the ground and oriented towards the sky. This used the Gap Light Analysis Mobile App (GLAMA 3.0), which calculates the area of the photo that is obstructed by leaves or tree branches. Hummock height was measured using a SmartLeveler (Digital Leveling Systems, Smyrna, Tennessee, USA) that was referenced to the groundwater monitoring well.

R version 3.6.1 (R Core team, 2020) was used for all processing, analysis, and visualisation. Nonmetric multidimensional scaling (NMDS) was performed with the function "metaMDS" in the "vegan" R package (Oskasen et al., 2013), which reduces dimensionality in the data. The cover of each species was passed through the metaMDS function with a Bray-Curtis dissimilarity index (Faith et al., 1987) and k = 2 dimension. The Bray-Curtis index is commonly used within community ecology as it is suited to handle cover data where species are absent (Oskasen et al., 2017). Restricting the NMDS ordination to two dimensions allows for easier interpretation of figures. The function "Envfit" was used pass an analysis of similarity test (ANOSIM) to determine how ordination of quadrat locations compared to mean RZ loss and land cover type. Land cover classes were based on definitions provided in the "Introduction to Ecological Land Classification Systems" Lee et al (1998). Swamps have > 25% tree cover, fens water source is predominantly minerotrophic, bog water source is predominantly ombitrophic. The significant species (p <0.05) that influence the location of (p < 0.05) of ELC derived land cover type in ordination space were listed on the ordination plot along with mean RZ loss of plotted sites. This allowed for associations between vegetation, community type and habitat suitability to be made.

2.2.4 Catchment to wetland ratio

ESRI ArcMap 10.5.1 desktop was used to manually delineate the wetland boundaries. The wetland boundaries were digitized from 0.2 m resolution Central Ontario Orthographic Project (COOP) 2016 imagery (COOP; OMNRF, 2016). The watershed boundaries were created using a privately acquired light detection and ranging (LiDAR) derived digital elevation model (DEM), and the COOP 2016 DEM, and tools from the hydrology toolbox within ArcMap. The two sets of LiDAR derived DEMs that were used had vertical resolutions of either 5 cm, or 2 m. First, the sink tools were used to create a depressionless DEM, then a flow direction tool was to create a flow direction grid. This was used as the input of the flow accumulation grid using the flow accumulation grid tool. A threshold value of 1000 was chosen to represent the flow accumulation, meaning cells with greater than 1000 upstream cells will be symbolized differently than those with less. GPS points

representing the outlet of the watershed were snapped to the nearest flow accumulation cell using snap pour points. These pour points and the flow direction raster were then used with the watershed tool to determine the area of the watersheds.

2.2.5 Water table recession and rise

To understand the potential for flooding at different peatlands, the linear slope of the WT following WT rise events was calculated in R 3.6.1. This linear recession was used for ease of interpretation and since the groundwater recession appeared to recede linearly (Cuthbert, 2014). These WT peaks could result from melt events, rain on snow, or rain events. The largest 15 peaks over the overwintering period from October 15th – May 1st were manually selected based on graphs of the winter WT. A peak was considered to occur when there was a two-day period after the peak without subsequent WT rise greater than 25 % of the initial rise. The largest fifteen events were selected under the assumption that the same rain and melt events would not occur at sites that are separated by up to 80 km. For these same events, the WT rise was calculated by comparing the WT before a melt or precipitation associated rise occurred, to the WT at the peak.

2.2.6 Snow

Snow depth was determined remotely with time-lapse sequences captured on several Stealth Cam XS16 16 MP wildlife trail cameras (Stealth Cam, 2020). The cameras collected five photos a day hourly between the hours of 10 AM and 2 PM, from December 2019 until April 2020. To have a scale reference, bright orange wooden stakes with scale markings 20 cm apart were placed into the peat directly behind *Sphagnum* hummocks. When snow cover had receded in mid April 2020 daily snow depth was manually determined for seven sites for the period of October 31st – April 17th 2020. In order to determine snow depth to the nearest cm Photoshop (CC 2019) was used to superimpose a scaled semi transparent 2 m stick on the trail camera imagery. The markings on the

snow stakes were used as the scale reference, and the superimposed reference allowed for more detailed manual determination of snow depth. If snow depth varied among images collected on the same day, an average value was taken.

In R 3.6.1, three linear regression models were fit to examine how ecohydrological variables related to differences in snow height and duration. The first model had a log transformed mean cover of snow at snow stake locations as the response variable against the product of height and coverage of shrubs in the snow survey location. The second model had the duration of snow cover at a location as the response variable compared to the canopy closure at that location, and the third had the duration of snow cover as the response variable compared to the product of height and coverage of shrubs in the snow survey location. In each model standardized quantile plots of the residuals were plotted to determine heteroskedasticity of the residuals.

Snow surveys were conducted with a meter stick on Feb 12, 2020 to assess how snow depth varied across a site. A rip cutter (Snowmetrics, Fort Collins, CO) snow corer and electronic scale were used to assess density, as the quotient of snow mass and corer volume. The wetland microform type, snow depth and any visible vegetation above the snowpack were recorded, while taking 100 snow depth measurements and 12 snow density measurements. Wilcoxon signed-rank tests were performed in R 3.6.1 to evaluate if there were significant differences in the mean values of snow depth in areas with and without vegetation visible above the snowpack, and if snowpack density varied with microform type.

2.2.7 Model selection

To understand if ecohydrological variables had significant differences between sites and if this was related to mean and total RZ Loss, a factor analysis, principal component analysis, and

multiple linear regression in R. The factor analysis, and PCA was done to eliminate variables that were multicollinear or that were not significantly related to the mean or total RZ loss. The ecohydrological variables included the % cover of shrubs in a site, shrub height, a shrub factor that equaled the product of shrub height and cover, hummock height relative to mean annual WT, canopy closure, annual WT variation, mean winter snow depth, number of days with greater than 10 cm of snow, catchment to wetland ratio, and peat depth. The response variables were total and mean RZ loss for the winter of 2019-2020. Total and mean RZ loss were transformed with a "Yeo Johnson" and "order norm" transformations respectively using the bestNormalize package (Peterson and Peterson, 2020) in R 3.6.1 and normality was confirmed with Shapiro-Wilk tests.

To account for multicollinearity, the 10 ecohydrological variables were first centered, scaled, and combined into two linear mixed effects models. These ecohydrological variables were the fixed effects, mean or total RZ loss as the response variable, and site and hummock location as a nested random effect using the function "lme4" in R 3.6.1 (Bates et al., 2015). The variance inflation factor (VIF) was calculated and several ecohydrological variables were found to be multicollinear (Table 2.2), since they had VIF values > 10 (Paul, 2006; Kim, 2019; Daoud, 2017). The ecohydrological variables were reorganized into a new regression, removing variables to form the largest combination of ecohydrological variables, where the VIF of all values are below 5 (Kim, 2019).

The uncorrelated independent variables were run through a Principal Component Analysis (PCA) (Figure 2.8) to understand if and how sites differed with respect to independent ecohydrological variables. This PCA plot also allowed for the interpretation of how different hummock locations varied within and between wetland locations. The proportion of variance explained by each PCA axis was used to determine the number of factors to use in a subsequent factor analysis (Figure

2.9). If there was greater than 10 % variance explained by a PCA axis, then it represented an additional factor required (Cattell, 1966; Williams et al., 2010).

Two factor analyses were run with the function "factanal" in R 3.6.1. The first included the independent ecohydrological variables and the mean RZ loss as the response variable. The second had the same independent variables and total RZ loss as a response variable. If the loading of the variable in any of the factors was found to be above 0.75, then the variable was considered to be significant, because of the small <50 observation sample size of the data set (Hair et al., 1998). All significant independent factors were inserted in every possible combination of linear mixed effects models with site and hummock as nested random variables.

The second order corrected Akaike Information Criteria (AIC_c) was used to compare the performance of models, where values of delta $AIC_C < 2$ between models implies parsimony (Burnham and Anderson, 2002). If delta AIC_C was less than 2, then model averaging was done with the function model.avg in R 3.6.1. Restricted maximum likelihood (REML) was set to false, because multiple models with differing fixed effects and the same random effects structure were compared (Harrison et al., 2018). The normal distribution of residuals was confirmed with a Kolmogorov Smirnov test, and standardized quantile plots of the residuals.

2.3 Results

2.3.1 Site properties and multi-year resilience zone data

Wetland	Hummock	Canopy	Shrub cover	Shrub	Days with over	Peat Depth
	neight (em)	(%)	(70)	neight (em)	10 cm	(CIII)
U1	21.5 ± 0.7	4.3	36 ± 1.7	29.1 ± 0.6	33	97.9 ± 3.3
U2	22.7 ± 1.0	0.5	26 ± 1.3	26.7 ± 0.4	27	127.2 ± 3.3
U3	26.8 ± 2.8	0	25 ± 1.1	21.8 ± 0.7	33	121 ± 3.3
U4	28.14 ± 1.0	15.0	32 ± 1.5	40.1 ± 0.7	57	74.2 ± 3.3
C1	16.4 ± 1.6	13.1	7.5 ± 5.5	43.36 ± 4.8	71	139.2 ± 3.3
C2	33.5 ± 3.0	38.6	36 ± 4.5	64 ± 5.4	59	156.3 ± 3.3
C3	37.82 ± 1.4	56.2	49 ± 5.3	34 ± 0.4	55	51.2 ± 3.3

Table 2.1: Wetland properties $(\pm SE)$ at the five hummock locations for all sites

The wetlands were found to be have many overlapping ecohydrological properties. The confirmed sites C1 differed in many properties compared to C2, and C3. The site C1 was more of a poor sedge fen, whereas C2 and C3 can be defined as organic coniferous swamps due to the greater than 25 % canopy cover of trees over 5 m according to the Ecological Land Classification Framework (Lee et al., 1998). The Shrub cover was noticeable higher in wetlands C2 and C3 (42.5 %) than in the unconfirmed sites (mean 32.3 %). Similarly, when comparing between confirmed sites (C2, C3) and unconfirmed sites, it is found that mean values for hummock heights (35.6 vs 24.8), canopy closure (47.4 % vs 4.95 %), shrub height (49 cm vs 29.4 cm) days with over 10 cm of snow (57 days vs 38 days) were all higher in in C2 and C3 than in all unconfirmed sites.

At all these sites, the winter of 2019-2020 was warmer and had a higher average air and peat temperature and WT than the winter of 2018-2019 (Table 2.1). This meant that, despite having 5 more snow free days with mean temperature < 0 °C, the 0 °C isotherm remained higher in the hummock profile in the winter of 2019-2020. However, the winter of 2019-2020 also had longer

resilience zone loss events, which are periods when the isoline is beneath the water table (Figure 2.1). These RZ loss events occurred across multiple sites earlier and more often in the winter of 2019-2020 than in the winter of 2018-2019 (Figure 2.2). It was found that the timing of RZ loss events roughly corresponds with the timing of WT rise events over the winter (Figure 2.2).

Table 2.2: comparison of two winters (period between Oct 15^{th} – April 17^{th}). Variables were averaged across all 7 surveyed peatlands.

Winter	2018-2019			2019-2020		
	Q 0.25	Median	Q 0.75	Q 0.25	Median	Q 0.75
WT (cm below hollow surface)	-9.3	-5.7	-2.0	-8.7	-4.1	-0.6
Air Temperature °C	-9.7	-2.5	1.2	-2.4	0.8	4.8
0 °C Isotherm depth	8	18	27	5	11	21
Snow free days with < 0 °C air temperature	-	8	-	-	13	-
Hours of RZ loss	3.75	7	25	6	14	42.5



Figure 2.1: An example of WT position versus 0-degree isotherm graph for the winter of 2019-2020. A RZ loss is the period that the WT is above the 0-degree isotherm. A negative depth value indicates the WT is above the hummock surface.


Figure 2.2: a) and c) Count of daily resilience zone loss events for 2018-2019 and 2019-2020 respectively. b) and d) The mean WT position across all monitored sites for 2018-2019 and 2019-2020 respectively.

2.3.2 Vegetation

In the surveyed peatlands, species richness ranged from 16 - 70 per peatland. Using an analysis of similarity, species could be separated into groupings based on ELC Ecotype (R(11) = 0.44, p = 0.003), mean duration of RZ loss (R(11) = 0.75, p = 0.003), and total duration of RZ loss (R(11) = 0.51, p = 0.01). Wetlands classified with the ELC framework as swamps were associated with the lowest mean and total RZ loss times (Figure 2.3). The significant species associated with swamp communities were *Gaultheria procumbence* (wintergreen), *Galium sp.* (bedstraw), and *Kalmia angustifolia* (sheep laurel). *Mainthemum canadensis* (Canada mayflower), *Vaccinium angustifolia* (lowbush blueberry), and *Gayluccasia baccatta* (huckleberry) are species that exist somewhere between a bog and a swamp. Conversely *Carex sp.* (sedges), *Vaccinium oxycoccos* (bog cranberry), and *Alnus incana* (river alder) were associated with poor fens that had greater RZ loss times.



Figure 2.3: Non metric multidimensional scaling of vegetation surveys with significant species (p < 0.05) colored to display ELC class by color and mean RZ loss time in hours by point shape type. Species that had significant (p < 0.05) influence on the location of points are displayed over the plot.

2.3.3 Catchment to wetland ratio

A linear model showed that for the 15 largest WT rise events of winter 2018-2019, peatlands with larger CWR were significantly associated with lower rates of WT recession (T(104) = 4.87, p = 3.67 x10⁻⁶, R² = 0.18). This includes melt, rain, and rain on snow events for the periods between October 15 2018 – May 15 2019. The WT rise from these same events was not significantly related to the CWR (Figure 2.4).



Figure 2.4: a) Water table (WT) rise versus the Catchment to Wetland Ratio (CWR) b) WT recession slope vs CWR for the winter of 2018-2019

2.3.4 Snow depth

Across 7 peatlands there were a total of 18 trail cameras that measured daily snow depth. The standard deviation of within peatland snow depth was 1.9 - 9.8 cm, and the standard deviation of snow depth between all wetlands was 10.9 cm. Mean snow depth was significantly positively related to the proportion of tall shrubs within the peatland (t(17) = 2.82, p = 0.01, R² = 0.31). Duration of snow cover at locations was significantly related to canopy cover (t(17) = 3.41, p = 0.003, R² = 0.40) (Figure 2.5). Standardized quantiles plots of the residuals demonstrated heteroskedasticity for both these regressions. The duration of snow cover was not significantly related to the proportion of tall shrubs.



Figure 2.5: a) Mean height of shrubs multiplied by coverage of shrubs at hummock vs mean snow cover, b) canopy cover vs days of snow cover for the winter of 2020.

For the snow survey performed on Feb 12th, 2020, there was a significant difference in snow depth between randomly selected points (U(1, n = 99) = 385, p < 0.001) (Figure 2.6 a), with or without vegetation visible above the snow surface. There was not a significant difference in snow density by microform (U(1, n = 12) = 26, p = 0.24). However, there was a 100 kg m⁻³ difference in snow density over short spatial distances in the same peatland (Figure 2.6 b).



Figure 2.6: a) A comparison of snow depth at locations where vegetation was or was not visible above the snowpack b) snow density versus microtopographic position on Feb 12th 2020

2.3.4 Model selection

When the 10 ecohydrological variables were initially inserted in a mixed effect model, the VIF for several variables was too high to assume independence as values varied from 4.02 - 1824 This represents multicollinearity, which was most pronounced in variables that may be measuring similar metrics. The multicollinearity between CWR and peat depth (57, 58) indicates high linear correlation between peat depth and hydrological connectivity within this landscape. In addition, the rotational overlap of the shrub index and canopy closure within the PCA (Figure 2.8) and high VIF values (1824) may demonstrate multicollinearity between these variables.



Figure 2.7: PCA for 10 ecohydrological variables for the winter of 2019-2020 showing how ecohydrological variables and site locations relate.

The largest combination of ecohydrological variables with VIF below 5 resulted in the 6 variables (Table 2.9). When a PCA was conducted with these 6 variables, the sum of the first two eigenvectors was 73.56 (Figure 2.9). The sum of these eigenvectors was above 70 % and was sufficient to explain most of the variability (Rea and Rea 2016). To interpret the PCA (Figure 2.9), sites that are clustered together are correlated for many of the ecohydrological variables. If ecohydrological variables are positively correlated, they have the same rotational angle and negatively correlated would be closer to 180° away. Therefore, environmental variables closer to 90° would be uncorrelated. The length of the arrow signifies the effect this variable has on plotting data in 2-dimensional space. The PCA plot shows that hummocks within the same site cluster close together and thus more variation can be explained by the site than the hummock location. The unconfirmed sites all cluster closely together, which indicates that they covary with the independent variables.

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Figure 2.8: PCA for 6 independent environmental variables for the winter of 2019-2020.



Figure 2.9: Scree plot for the reduced variable PCA. This plot indicates 3 factors are required for FA since PCA3 variance is > 10%.

It was found that the WT variability, hummock height, mean snow cover and shrub height were significantly related to the mean RZ loss according to the factor analysis (Table 2.3). In the model of mean and total RZ loss, the three fixed effect variables found to produce the model with the lowest AIC_c were hummock height, shrub height, and WT variability (Table 2.3). Though mean snow cover was found to be significant within the factor analysis for mean RZ loss, it did not occur in the model with the lowest AIC_c. While hummock height and shrub height have similar model coefficients in the mean RZ loss model (-0.49, -0.48 respectively) (Table 2.3) hummock height has considerably higher coefficient than shrub height in the total RZ loss model (-0.84, -0.26). Similarly, WT variability has decreasing importance in the total RZ model compared to the mean

RZ loss model. Therefore, these two models with differing response variables are both needed to understand overwintering suitability in terms of mean and total threat to suitability over the winter.

Table 2.3: Reduced model Variable inflation factor (VIF), resultant factor analysis (FA) scores where values above 0.75 are considered significant, model weights from model averaging and model coefficients from averaged models. CWR = Catchment to Wetland Ratio, WT = water table.

Variable	VIF		Largest FA		Sum of Model Weights		Model Coefficients	
	RZ Mean	RZ Total	RZ mean	RZ total	RZ mean	RZ total	RZ mean	RZ total
Canopy closure	3.3	3.3	0.71	0.7	-	-	-	-
Hummock height	2.7	2.7	0.89*	0.96*	1	1	-0.49	-0.84
Shrub height	3.2	3.2	0.93*	0.94*	1	1	-0.48	-0.26
Mean snow	2.2	2.2	0.75*	0.74	-	-	-	-
CWR	2.1	2.1	0.59	0.55	-	-	-	-
WT variability	1.7	1.7	0.95*	0.95*	1	0.61	0.27	0.16
Intercept] -	-	-	-	-	-	0.09	-0.06

2.4 Discussion

2.4.1 Multi-year resilience zone data

Differences in the timing and amount of RZ loss over the two monitored winters show drivers of RZ loss. The depth of the 0 °C isotherm might have been more affected by the position of the WT and median air temperature than the number of snow free frost days and total snow cover (Table 2.1). The winter of 2018-2019 had a colder median air temperature but had a greater total snow cover and total duration of snow cover. This winter of 2018-2019 had a snowpack of 26 cm in late December, compared to a snowpack of 18 cm in late December 2019-2020. The result that the 0 °C isotherm was deeper in the winter with greater December snow cover does not align with the December snow cover trend that Verry (1991) considers to be crucial for predicting frost depth over the winter in peatlands. The shallower isoline in the winter 2019-2020 could have been due to the increased prevalence of mid-winter rain events, which are known to increase sub-snowpack soil temperature (Putkonen and Roe, 2002; Rennert el al., 2008). The depth of the 0 °C isotherm is responsive to WT position (Figure 2.1), which could indicate the importance of this thermal mass.

Despite the shallow 0 °C isotherm depth in the winter of 2019-2020, the higher mean WT indicates that there were far more RZ loss events than the winter of 2018-2019 (Figure 2.2). These RZ loss events overlap considerably with WT rise events (Figure 2.2), and result in a greater duration of unsuitable conditions. Mid-winter rains can also form an impenetrable ice cap that restricts infiltration of future snowmelt and rain on snow events (Putkonen and Roe, 2002). It is not known how snow cover, the position of the WT, and presence of an impenetrable ice cap affect the availability of oxygen within the overwintering space (Yagi et al., 2020).

We have assumed that cutaneous respiration (breathing through the skin) in massasaugas is not possible or very limited, and that access to aerial oxygen increases as the water table drops within unfrozen peat (Markle et al., 2020b). It is critical that restored sites in this region, and other similar ones, are designed to provide suitable habitat immediately and as climate change intensifies. This includes predicted warmer winters that have a greater proportion of precipitation occurring as rain (Williams et al., 2015). Having a sufficiently deep peatland to accommodate space between the 0-degree isoline and the WT can help buffer against this future. It is also important to have high drainable porosity to dampen WT rise events, and a sill that can efficiently spill mid winter rain events.

2.4.2 Vegetation

Sites classified as swamps had lower durations of mean and total RZ loss, similar to observations made by Markle (2019, unpublished). Markle (2019) used linear discriminant analysis on 151 wetlands and found that shrub cover, tree cover, and simpler spatial complexity were associated with wetlands that have spring emergence records of massasaugas. ELC classification determines that wetlands with tree cover between 25% - 75% are classified as swamps. These wetlands may differ from the typical swamp and may instead be thought of as very treed bogs, or very treed poor fens. Swamps in the study sites contained tamarack (*Larix laricina*), jack pine (*Pinus banksiana*), white pine (*Pinus strobus*), and red maple (*Acer rubrum*), and had peat depths between 40 - 150 cm. The greater canopy cover in these swamp wetlands is associated with certain trade-offs for suitable massasaugas habitat. One trade-off is increased canopy cover, which is known to increase the amount of snow interception in the boreal (Hedstrom and Pomeroy, 1998). This could decrease the amount of midwinter rain that reaches the peat surface (Hedstrom and Pomeroy, 1998), but can also decrease the amount of early season snow. Greater canopy cover was associated with a longer

duration of snow cover in our surveyed peatlands but has been known to decrease the amount of pre- and post-emergence basking opportunities that are critical for gravid massasaugas females (Johnson, 1995).

A longer duration of snow cover would shift spring emergence later (Figure 2.5), when there are less likely to be spring air temperatures below 0 °C. Despite this, Johnson (1995) found that massasaugas have avoided overwintering in areas with greater canopy cover. A lack of basking habitat in forested areas has led to recommendations for prescribed burns (Pratt et al., 1999; Schaefer, 1996) and mowing (Johnson, 2000) to increase basking massasaugas habitat. It is possible that the finding that massasaugas in the EGB region select overwintering hummocks with greater canopy closure arises from spatial variability of wetlands in EGB, and the existence of nearby patches of bedrock with minimal vegetative cover. This would contrast with other areas in massasauga's southern range, where wetlands may be more homogeneous. An analysis of multiple massasaugas populations spatial variability of overwintering habitat would be necessary to answer why variation exists in the outcome of canopy cover for massasaugas.

The swamp sites had a high proportion of taller woody vegetation, including generalist species like *Chamadaphne calyculata*, fen indicators species like *Alnus incana*, and those that overlapped with swamp and bog species such as *Gaylussacia baccata*, and *Vaccinium angustifolia*. Provided shrubs species are able to reach a similar height and cover in differing wetlands, it is likely that these species would have comparable ability to increase surface roughness. This increased surface roughness can allow for greater snow accumulation (Figure 2.5 and 2.6), and Pomeroy et al. (2006), as well as provide structural support for microtopography (Johnson, 1995).

The presence of shrubs in different systems, as seen in the NMDS (Figure 2.3), is determined by negative species interactions and abiotic conditions (Götmark et al., 2016). Species like *Chamaedaphne calyculata* are not present on this NMDS, as they are generalist species that are present across multiple ELC classes. This generality makes *Chamaedaphne calyculata* an excellent candidate for restoration where there are multiple possible trajectories for succession.

Species like *Chamaedaphne calyculata* (Wilcox et al., 1989) and *Vaccinium angustifolium* (Kender and Walter, 1967) will often establish well from cuttings after a disturbance. This means that if fragments of these species were present in the upper layers of stockpiled peat material, they will likely naturally recolonize restored peatlands. Additional transplants of cutting from on-site material could easily be added through live stake transplants if a greater proportion of shrubs is required.

Other shrub species, such as *Gaylucassia bacatta*, tend to be later successional species, however, if established, this species can quickly recolonize after a disturbance and will move into anthropogenically disturbed areas (Fimbel and Kuser, 1993). It also establishes taller and more abundant growth with diffuse light around 33% canopy cover (Henderson et al., 1984). This association of *Gaylucassia bacatta* with canopy cover was observed in the NMDS (Figure 2.3), as this species plotted closer to the more treed swamp ELC class. Once trees in the restored peatland have established, it would be an ideal species to allow to naturally revegetate restored peatlands that are connected to adjacent peatlands. Alternatively, this species is well adapted to vegetative transplant via rhizome and root transplant (Camp, 1941) if intervention is desired.

A vascular overstory can act as "nurse plants" that hasten the growth of *Sphagnum* in restoration projects. This facilitative effect occurs through the reduction of temperature extremes,

modification of irradiance, and change in moisture availability (Groeneveld et al., 2007). While *Sphagnum* needs a minimum amount of photosynthetically active radiation (PAR), once it is past saturation levels, its growth can be inhibited (Murray et al., 1993). *Sphagnum* growing on bare peat in a restored peatland is likely to be loosely packed and will lose moisture through the large capitula surface area exposed to air (Groenveld et al., 2007). Therefore, species that can shade the new *Sphagnum* without competing for limited resources, will allow for the facilitation of *Sphagnum* growth.

Several studies have demonstrated that *Eriophorum sp.* functions as a nurse plant for *Sphagnum* (*e.g.* Rochefort and Bastien (1998); Tuittila et al., (2000)). Similarly, several studies have found that *Polytrichum strictum* aided in the growth of *Sphagnum* (*e.g.* Groeneveld and Rochfort, 2002; Gonzolaz et al., 2007), especially on bare peat surfaces. These species moderate harsh growing conditions, without limiting photosynthetically active radiation and moisture availability.

Problems occur when competition between vascular plants and *Sphagnum* sp. reduce PAR beyond the light compensation point (Kangas et al., 2015), and moisture beyond the 100 mb of pore water pressure when hyaline cells begin to drain (Thompson and Waddington, 2013a). This can occur via the shrubification-WTD feedback (Waddington et al., 2015) where shrub canopy interception results in greater transpiration, and the increased aeration of the root zone results in greater transpiration demands. Both processes feedback to lower the WT over time.

Several studies of restored peatlands have observed the establishment of dense communities of shrubs that prevent *Sphagnum* accumulation over time (González and Rochefort, 2014; Strobl, 2019). Since shrubs are desired for massasaugas overwintering habitat, the maintenance of a shallow WT is necessary to prevent the shrubification feedback from occurring while *Sphagnum*

is establishing. By restoring to a floating fen system, the WTD is minimized and the possibility of a shrubification feedback occurring would be reduced.

2.4.3 Catchment to wetland ratio

The maximal WT rise, and rate of recession are associated with the likelihood of massasaugas habitat flooding, and the duration that the flooding will persist. The finding that rate of water table recession in peatlands is significantly related to the catchment to wetland ratio (CWR) during the overwintering period agrees with Didumus (2016), who found a noticeable pattern between CWR and WT recession in shallow depressions in this landscape during wet periods. In the winter, the WT is very close to the sill, and many upstream wetlands are spilling for each WT rise event. Therefore, it would be anticipated that the inflow of water would continue for a longer period in wetlands that are more connected to their contributing areas (Devito et al., 1996).

Despite this slower rate of WT recession, wetlands with greater CWR were not related to greater rates of WT rise for winter WT rise events (Figure 2.4). Instead, wetland characteristics that include initial WT, size of sill, and site drainable porosity, are likely to affect the water rise for WT rise events. Lower Sy or drainable porosity values represent flashier WT and would be associated with lower peat porosities from decomposition (Waddington et al., 2015) or with lower proportions of open water areas within the peatland. Since drainable porosity changes with depth, the magnitude of winter WT rise would be dependent on the initial WT (Markle et al., 2020a). In addition, peatlands with smaller sills can have prolonged flooding when ice or snow dams prevent the spilling of water over the sill (Markle et al., 2020a).

Both the maximum value of WT rise, and the rate of WT recession, would be affected by characteristics of the catchment. These include the average soil depth and proportion of open rock

(Buttle and Eimers, 2009; McNarma et al., 1998). Therefore, locating the restored wetlands adjacent to disturbed areas with impermeable surfaces could allow for large quantities of inflow over a short duration. This would increase the likelihood of flooding in the restored peatland, especially if the CWR was large and the catchment area was largely disturbed. Even without disturbed impenetrable area within the catchment, we recommended that site locations and wetland sizes that result in a CWR of 11 or less should be used. This is the maximal value of CWR that results in resilient hibernacula (Figure 2.4). If flooding risk is a concern, changes to the site location, depth, or size can mitigate this risk.

2.4.4 Snow depth

The EGB region has several mid-winter snowmelt and rain on snow events. These types of events can cause sudden differences in the density of the snowpack (Romanov, 2013). In addition to temporal variation in snow density, snow density can vary over short spatial distances (Figure 2.6). The moisture contained in peat at a depth of 1 cm, the energy associated with phase changes, and changes to thermal conductivity with phase changes would introduce variability not present when monitoring basal snow temperature like Marchand (1982). A model that incorporates meteorological conditions, or regular density measurements, paired with basal snowpack temperature measurements could likely quantify the insulative potential of the snowpack. This could give the depth and density of snow that is associated with no further decrease in temperature variability. It could also be used to model past mortality events, or model potential climate change impacts on massasaugas hibernacula.

2.4.5 Model selection

The findings from the model selection help determine which environmental variables are important for massasaugas overwintering restoration design. However, the inclusion of only seven sites in the analysis means that this analysis might only consider a subset of the possible overwintering habitats. Therefore, other functional relationships might not be covered within the analysis and the magnitude of the ecohydrological variables might be exaggerated or dampened. For example, the low value of CWR within the FA might be exaggerated because of site selection for this analysis.

All the surveyed peatlands have peat depths > 50 cm. This is above the 50 cm threshold that Didemus (2016) found to be related to lower porosity and greater WT variability. Shallower sites were also found to be more associated with slower rates of water table recession after a storm event during the wet season (Didemus, 2016). While variability below 50 cm of peat depth does not exist in the FA, it does on the rock barren landscape and is still a critical component for restoration. This means that the reference ecosystem that practitioners should model their restoration towards should have peat depths greater than 50 cm.

The PCA plot (Figure 2.9) shows that replicate hummocks within the same site cluster close together, indicating that important variables may differ more at the meso than microscales. While shrub cover, hummock height and canopy closure can vary considerably across a wetland, WT variability, snow cover, and CWR varies far more between wetlands than within. This helps contextualize the observation that massasaugas have fidelity to return to overwinter within 100 m of the previous year overwintering location (Harvey and Weatherhead, 2006b; Yagi et al., 2020), rather than in the same hummock. This highlights the importance of not disturbing wetlands with interventions that would disrupt regular WT fluctuations like drainage, improper culvert installation and modification to upstream habitat (Vu, 2019). Site level variation also highlights

the importance of restoring multiple suitable hummocks to increase redundancy within a restored overwintering wetland.

The finding that taller shrubs, taller hummocks, and a more stable WT are associated with more suitable overwintering conditions for massasaugas allows for planning on how best to design overwintering habitat. It is not known whether it is shrub cover associated with greater snow cover (Figure 2.5) (Strum, 2005), or the increase in average soil temperature (Menard et al., 1998; Paradis et al., 2016; Pomeroy et al., 2006) that leads to an increase in RZ size and duration. Within the seven sites included in the analysis, the shrub height and cover were greatest when there was greater canopy closure (Figure 2.5). According to the model, a shrub dominated swamp community (as defined by ELC criteria) with > 25 % tree cover greater than 5 m in height is the goal restoration community. Despite this finding, shrub dominated swamps are not the only system that provides resilient overwintering habitat in the rock barren landscape. Smolarz et al. (2018) found that sedge dominated poor fens with tall hummocks were able to provide resilient overwintering conditions for the winter of 2015-2016. Markle (unpublished) also found several examples of sites not dominated by shrub cover that were associated with spring emergence massasaugas records, though wetlands were more often associated with shrub and tree cover in the linear discriminant analysis mentioned in 2.2.5.

There is more than one way to meet the requirements of massasaugas habitat, and multiple factors that can be involved to meet those requirements. For example, the lateral movement of groundwater and/or surface was not measured. Therefore, this analysis could have missed quantifying the potentially important advective heat transfer (Carey and Woo, 2005) that occurs in wetlands with greater connectivity. The information presented in this chapter will be incorporated in the design of overwintering habitat in Chapter 3.

Overall, recommended characteristics for restoring overwintering habitat for massasaugas in eastern Georgian Bay from this chapter include the list below. These recommendations are based on the results of this chapter. Individual recommendations may require more testing to establish causal links and recommendations may not apply to other landscapes.:

- An absolute cover of > 20 % of shrubs that are > 25 cm in height to allow for *Sphagnum* growth support, and snow accumulation;
- Interspersed areas of open water and floating peat to dampen water table fluctuations, maintain near-surface water table to allow for a sufficient ground heat flux, and near-surface WT to prevent the shrubification-WTD feedback;
- An eventual canopy cover between 25-75% over hummock locations from trees > 5 m in height to prolong snow cover;
- Areas with open canopy throughout the site, and adjacent to the site provide fall and spring basking habitat;
- 5) Average peat depth greater than 50 cm.
- 6) A sill that is ~20 cm below the top of the hummock and spans 25% of the circumference of the wetland, to prevent snow and ice from blocking an outlet
- 7) A catchment to wetland ratio below 11, to allow for more rapid WT recession.

CHAPTER 3: RESTORATION OF MASSASAUGA RATTLESNAKE (SISTRURUS CATENATUS) OVERWINTERING HABITAT: CONSTRUCTION METHODS AND ECOHYDROLOGICAL ASSESSMENT

3.1 Introduction

3.1.1 Context for restoration

The Eastern Georgian Bay (EGB) region is situated on the east shore of Lake Huron in the Great Lakes Watershed. This Ecoregion is distinguished from the Lake Simcoe-Rideau Ecoregion to the south by the frequency of bare rock ridges made of predominantly gneissic rock (Crins et al., 2009). The substrate mineralogy is primarily acidic, with podzolic and rock cover as the primary and secondary substrate types respectively (Crins et al., 2009). The EGB region has many peatlands contained in bedrock depressions (basins) within the Precambrian shield. Water contributes to these wetlands in the form of precipitation, snow melt, surface water inflow, overland flow, and limited groundwater inflow until the storage capacity of the basin is exceeded and water runs off ("spills") to a lower topographic position, in a process known as "fill and spill" hydrology (Spence and Woo, 2003).

The EGB region is a biodiversity hotspot supporting over 50 species at risk (Georgian Bay Biosphere Reserve, 2018), including the eastern massasauga rattlesnake (*Sistrurus catenatus,* hereafter referred to as massasauga). Massasaugas require overwintering habitat that provides stable, cool, and above 0 °C temperatures to maintain a reduced metabolic rate (Gregory 1982; Macartney et al., 1989), protection from desiccation (Costanzo, 1989), and protection from drowning during flooding events (Smith, 2009; OMNRF, 2016a; Markle et al., 2020a; Yagi et al., 2020). These overwintering requirements are met with different habitat features for different subpopulations of massasaugas. These habitat features include crawfish and mammal burrows, rotting tree roots, deep fractures in the bedrock, and *Sphagnum* peat hummocks (Weatherhead and

Prior, 1992; Johnson, 1995; Harvey and Weatherhead, 2006b; Smolarz et al., 2018; Markle et al., 2020a) In the EGB region, massasaugas overwinter in *Sphagnum* peatland hummocks (Smolarz et al., 2018; Markle et al., 2020a). The volume of suitable overwintering space that these *Sphagnum* peatland hummocks provide for massasaugas can be conceptualized as a resilience zone, which is defined as the unflooded space where temperatures remain above 0 °C (Smolarz et al., 2018).

A development project occurring within known massasauga and freshwater turtles overwintering habitat has necessitated the construction of restored overwintering habitat for multiple reptile species. Artificial overwintering habitat was developed by the Toronto Zoo (Toronto Zoo, 2015), with a mixture of concrete blocks, gravel, stone, and sand. The Toronto Zoo habitat design has been found to be an effective overwintering habitat for eastern fox snakes (Pantherophis gloydi) at Point Pelee in Ontario (Dobbie, personal communication), but was not deemed to be suitable for massasaugas in Eastern Georgian Bay (OMNRF, 2018). The shallow soil depth and impermeable bedrock in this landscape allow for very little deep groundwater percolation. Therefore, any created hole will fill with water by the beginning of winter or during spring melt, and potentially expose massasaugas to flooded conditions. Additionally, the Toronto Zoo (2015) habitat design does not resemble the wetland features that the EGB subpopulation of massasaugas uses for overwintering. Massasaugas have been observed to return to overwintering habitat within 100 m of the previous years overwintering location (Harvey and Weatherhead 2006b; Yagi et al., 2020). Therefore, restored overwintering habitat for massasaugas in the EGB region should be built close to impacted overwintering locations and should have several Sphagnum hummocks that remain above the water table rather than deep rock crevices.

Meeting the demands of multiple reptile species requires overwintering habitat to be designed with characteristics of wetlands with co-occurring massasauga and turtle species observations (Markle

et al., 2020b). Anoxia-tolerant turtles like snapping turtles (*Chelydra serpentina*), painted turtles (*Chrysemys Picta marginata*), etc. can overwinter in bog and fen-like environments beneath the water table (Markle et al., 2020b). These species require cool, but above 0 °C temperatures to avoid freezing (Litzgus et al., 1999; Thienpont et al., 2004; Markle and Chow-Fraser, 2017), and to lower metabolic rate and increase tolerance to anoxic conditions that occur with complete ice coverage (Herbert and Jackson, 1985). Turtle overwintering habitat would have areas of shallow water with sufficient substrate for temporary refugia to avoid the complete freezing of the water column and/or predation (Brooks et al., 1991; Brown and Brooks, 1994; Thienpont et al., 2004). This can occur in a floating fen system where the requirements of massasaugas are also met (Markle et al., 2020b). These floating fens develop slowly over time through the process of territorialisation (Tsyganov et al., 2019). Conversely, the restoration approach outlined by this thesis is designed to restore functional overwintering habitat within a period of months before winter begins, which is what is required to conserve reptiles at risk when a disturbance occurs.

3.1.2 Co-occurrence of reptiles

To understand which wetland characteristics will make wetlands suitable for co-occurrence of overwintering reptiles, species location data of SAR reptiles paired with environmental variables should be used. Markle (2019) used spring emergence data from 151 wetlands and dendrogram analysis to separate wetlands into semi-aquatic, aquatic, and co-occurring wetland types. Semi-aquatic wetlands did not have open water all year and were associated with massasaugas. Aquatic wetlands had open water all year and were associated with several turtle species e.g. snapping turtles, painted turtles etc. Co-occurring wetlands has species observations that would be present in both aquatic and semi aquatic site types. Linear discriminant analysis was used to predict species occupancy and it was found that the percentage of open water, swamp, tree, shrub, and marsh

cover as well as the complexity and dispersion of each patch could be used to predict occupancy. Aquatic assemblages associated with turtles were associated with marsh and open water cover; semi-aquatic assemblages were associated with trees, shrubs and swamps; and co-occurring assemblages were associated with a high complexity of all habitat types interspersed. The cooccurring wetlands had twice the number of habitat patches per hectare compared to the other two wetland types (Markle 2019). The mosaic of vegetation types, floating peat, and open water in these co-occurring wetlands likely has the highest long-term resilience to external drivers that will allow overwintering wetlands to persist into the future (Markle et al 2020b). There is also redundancy in co-occurring sites whereby shifts towards a wetter or drier state would still result in a suitable wetland type for SAR reptiles in the region.

3.1.3 Wetland restoration techniques

Sphagnum sp. is a keystone species in peatlands (Quinty and Rochefort, 2003). Therefore, the reestablishment of the mechanisms that lead to peat accumulation such as near-surface water table (WT), depth-dependent transmissivity, and high drainable porosity are necessary for the continued self-regulation of peatlands (Quinty and Rochefort, 2003). Effective peatland restoration first identifies the functionality in the peatland that has been perturbed and how to re-establish the peat accumulation mechanisms in a way that addresses the perturbation. For this reason, the same restoration technique will not always have the same outcome in different locations (*e.g.* Parry et al., 2014).

A common technique used for the restoration of mined peatlands is referred to as the moss layer transfer technique developed by Quinty and Rochefort (2003). It involves surface preparations of blocking drainage, building berms, and removing non-target species. Peat diaspores are then collected by shredding all parts of the peat acrotelm into particles larger than 0.5 cm. The acrotelm

is a term used to refer to the upper, variably saturated layer of fibric peat (Morris et al., 2011). These plants are then spread out in a homogenous layer over the degraded peat surface to a depth of 1 to 5 cm. Finally, they recommend using 3000 kg ha⁻¹ of fresh straw mulch above the thin moss layer to prevent excessive surface heating and evapotranspiration (Quinty and Rochefort, 2003).

Sphagnum moss recolonization of bare organic soils is often hampered because of a lack of nearsurface moisture (Joosten, 1995; Poulin et al., 2005). Evaporative loss of new Sphagnum growing over bare peat can also be mitigated using nurse plants. A nurse plant is a species that facilitates the establishment of another species, often through improvements in microclimatic conditions (Lozano et al., 2020; Rochefort et al., 2007). Graf and Rochefort (2010) found significantly higher levels of Sphagnum length, biomass, and cover beneath 50% vascular shade cover than 100% open conditions. Similarly, Graf and Rochefort (2008) found in a field experiment that Sphagnum growth was greater beneath cover of Scirpis sp., than beneath Equisetum sp, or Polytrichum sp., because the Scirpis plot reached the greatest percentage of cover. Graf and Rochefort (2010) state that vascular cover facilitates Sphagnum growth by decreasing incoming radiation and increasing humidity and surface roughness. Increasing vascular cover beyond 50 % will increase Sphagnum stem length but reduces stem density and limits further growth (Graf and Rochefort, 2010). The use of the moss layer transfer technique and subsequent use of nurse plants methods serve as appropriate means of restoring peat accumulation but will take between 10 - 30 years before microtopography typically associated with natural bogs form (Pouliot et al., 2011).

An alternative method known as peat block restoration or the acrotelm transfer technique works by extracting the top 0.3 to 0.5 m of the peat surface and temporarily storing it on the uncut surface (Wilhelm et al., 2015). A meter of peat is extracted below the removed surface and the 0.3 - 0.5m of acrotelm peat is set back onto the flooded surface. This encourages peat to re-establish rapidly after harvesting without the need for off-site material like straw mulch or fertilizer. Additionally, the vascular vegetation is still present within the peat to act as a nurse plant after transplant. When conducted properly the acrotelm peat should not have time to desiccate before being placed onto the flooded surface, thus preserving ecohydrological properties required for growth (Wilhelm et al., 2015). In addition, with careful placement Wilhelm et al. (2015) argues that the development of microtopography with the acrotelm transfer technique could occur faster than with the moss layer transfer technique due to partial cover of woody shrubs present, and patchiness of the *Sphagnum* present, but an estimate of how much faster is not specified.

Floating peat techniques have also been used to restore rich and poor fens (e.g. Tomassen et al., 2003; Kooijman et al., 2016). If degraded peatlands began in an unflooded state, there may be a need to change the pH of the restored wetland for the proper vegetation communities to establish. Kooijman et al. (2016) found that the inundation of high pH water into degraded Sphagnum peatlands allowed the peatlands to shift towards the regionally rare rich fens. Instead of adding water, Tomassen et al. (2003) used a lime treatment to increase the pH of a degraded floating fen system to increase the rate of anaerobic decomposition of the peat and increase methanogenesis and thus the buoyancy of the floating mat. These floating mats dampen WT fluctuations and can increase the spatial complexity of surface types and thus the possibility of co-occurrence of reptiles in a restored wetland (Marke et al., 2020b). Other peatland restoration projects (e.g. Sottocornola et al., 2007; van Zuidam et al., 2018) note nutrient deficiencies limiting peatland restoration progress. To increase the nutrient availability in an oligotrophic fen wetland, van Zuidam et al. (2018) added differing concentrations of nitrogen and phosphorus and noted an increase in Sphagnum biomass accumulation within the first few years of restoration. For these different concentrations of nitrogen and phosphorus, van Zuidam et al. (2018) found that increasing the number of planted functional groups would increase the total biomass accumulated within the first couple years of the restoration. Having three different plant functional groups (*i.e.* clonal dominants, clonal stress tolerators, and interstitials) within the same wetland contributed to more abundant rhizome growth and created a wetland that resembled the reference wetland (van Zuidam et al., 2018). This initial rhizome growth can be important for establishing buoyancy in floating mats as they contain highly buoyant gas-transporting aerenchyma tissue (Tsyganov et al., 2018).

Tomassen et al. (2003) found that the mixtures of *Sphagnum* that were sourced from anaerobic locations were able to start methanogenesis sooner and colonize more effectively than the desiccated peatland *Sphagnum*, since the undegraded *Sphagnum* had higher concentrations of methanogenic cultures. Tomassen et al. (2003) found that the pH to bulk density (kg m⁻³) ratio greater than 0.05 was the best predictor of floating mat formation. For example, if wetland pH were 4 then the peat bulk density would have to be below 80 kg m⁻³ to have a pH to bulk density ratio greater than 0.05. It is necessary to understand the feedbacks that control floating fen development in degraded systems, and not just the feedbacks that operate in a healthy system to plan for a successful restoration.

3.1.4 Hummock development and peat properties

Taller hummocks are one of the key factors in wetlands that contribute to providing suitable massasauga overwintering habitat (Smolarz et al., 2018). Given this, it is important to understand the factors that lead to the growth of taller hummocks so that these processes can be incorporated into restoration design. In natural systems, hummock growth is developed and maintained by internal and external feedbacks otherwise known as autogenic and allogenic feedbacks. *Sphagnum* species are considered ecosystem engineers since adaptations maintain autogenic feedbacks for their own growth and persistence (Van Breemen, 1995). In response to drying, *Sphagnum* deforms

and reduces porosity, or increases albedo, allowing the peat wetland to conserve water in response to natural disturbances (Waddington et al., 2015). These northern peat wetlands are not adapted to be resilient against artificial drainage (Waddington et al., 2010). Under conditions of long-term drying, irreversible deformation can take place, which can reduce the compressibility of peat and result in irreversible losses in pore volume (Kennedy and Price, 2004). This permanent deformation and lowering of the peat surface are counterproductive if a restoration goal includes the growth of taller hummocks. Furthermore, this permanent reduction in porosity will decrease drainable porosity (hereafter referred to as specific yield or Sy) and not allow for (or greatly restrict) swelling of the peat when precipitation exceeds evapotranspiration (Kennedy and Price, 2004). This not only reduces the water storage capacity of the peatland but also increases the chance of wetland flooding. Therefore, care should be taken to maintain saturated conditions for any stockpiled *Sphagnum* sp., to maintain the regular functioning of autogenic feedbacks.

Hummock forming *Sphagnum* species have greater shoot densities than hollow species (Elumeeva et al., 2011). This allows for greater capillary rise, which reduces the likelihood of water stress and allows for growth further above the water table than hollow species (Elumeeva et al., 2011). Rydin et al. (2006) found that while hummock and hollow *Sphagnum* species both grow better under wetter hollow-like conditions, only hummock species can photosynthesize under drought conditions. An associated phylogenetic signal among the *Sphagnum* genera can be observed where *Sphagnum* species that differentiated more recently have similar niche differentiation in their microtopographic position (Bengtsson et al., 2016; Johnson et al., 2015). Nungesser (2003) used these species level differences in accumulation to model hummock heights at various wetlands within North America. The model found close agreement between modelled hummock heights and hummock heights in wetland locations in North America. However, a greater heterogeneity of

microtopography exists within the categorized ranges. This heterogeneity may indicate the importance of external or allogenic feedbacks that are associated with increasing and maintaining microtopography (Nungesser, 2003).

An important allogenic feedback found by Pouliot et al. (2011) is that *Sphagnum* stem length and biomass increased when low densities of vascular plants were introduced according to the stress gradient hypothesis (Maestre et al., 2009). The ericaceous shrubs and trees continue to provide physical support to allow for further hummock growth, until vegetation density increases to a point where competition limits *Sphagnum* growth (Pouliot et al., 2011). In recently restored peatlands, vascular plants can have a facilitative relationship with *Sphagnum* by providing shade, stable temperatures, and high relative humidity needed for growth in a stressful period of early peatland growth (Groeneveld et al., 2007). Diamond et al. (2019) summarizes these feedbacks succinctly by stating autogenic processes are largely responsible for the initial hummock formation and that vegetation feedbacks amplified further microtopographic differences.

A restoration practitioner should incorporate both autogenic and allogenic feedbacks into the restoration design. If microtopographic development is a goal, it is useful to understand the microtopographic niche preferences of *Sphagnum* species that are available for the project. Similarly, one must understand how species like *Polytrichum strictum* (Groenveld et al., 2007; Groeneveld and Rochefort 2002) or *Eriophorum sp.* (Rochefort and Bastien, 1998; Tuittila et al., 2000) facilitate the growth of *Sphagnum* over bare peat surface, by modifying temperature, irradiance, and humidity. This understanding is necessary to investigate the nature of species interactions between *Sphagnum* and different vascular species and understand the point at which facilitation turns into competition. While this competition may be beneficial for microtopographic growth when low densities of vascular plants are established in existing peatlands (Pouilot et al.,

2011), if vascular cover becomes too great the accumulation of *Sphagnum* can be reduced (González and Rochefort, 2014; Strobl, 2019). Since it is difficult to gain a complete understanding of all the allogenic relationships between vascular and bryophyte species, it is useful to instead compare plant community composition (presence and abundance of species in a system) of restored wetlands to some target natural systems (Mathews and Spyreas, 2010). In this case, one could compare the plant community composition of the restored wetlands to a target plant community composition of the restored wetlands to a target plant community composition of the restored wetlands to a target plant community composition of the restored wetlands to a target plant community composition of confirmed massasauga and SAR turtle overwintering habitat. These co-occurring systems have a high spatial heterogeneity of surface cover types with sections of open water and floating peat as described by Markle et al. (2020b). When community composition is similar between restored and natural systems, it has been found that autogenic and allogenic feedbacks that maintain the natural system behave similarly in the restored system (Zedler, 2009).

Since it will take years for successions to advance the restored wetlands to the state of the target reference system, a restoration practitioner will have to perform vegetation surveys over a span of years to determine if the wetland is on the correct trajectory. The order in which species colonize at a wetland or the "priority effects" can have a major role in community assembly as they can alter the system for species that arrive later (Fukami, 2015). To ensure the best chance of target species establishing, Hess (2019) recommends using planting to allow natives to establish first, considering the niche overlap between target and non-target species, and understanding the process of allelopathy, herbivory, and disease that can occur between target and non-target species.

Wetland restoration projects, no matter how well managed, are a disturbance that can allow for the establishment of undesirable species and trajectories if not properly planned and managed (Klimkowska et al., 2008; Mathews and Spyreas, 2010). Successful restoration often depends on the restored systems ability to replicate the function of the target natural system by reinitiating

successional processes (Hobbs and Norton, 1996). This requires ameliorating stresses that would prevent or misdirect this pattern of succession. Since Sphagnum is a keystone species in these wetlands, degradation to Sphagnum that prevents growth will often result in unsuccessful restoration outcomes (González et al., 2013). Due to unforeseen circumstances, the Sphagnum used in this restoration project had to be stockpiled for a period. This stockpiling may affect Sphagnum's ability to photosynthesize (Thompson and Waddington, 2008), and may result in denser peat that is more prone to dramatic WT fluctuations (Waddington et al., 2015). This chapter investigates if there are differences between natural and stockpiled peat properties; if there is a difference in vegetation community composition between restored and confirmed massasauga habitat; and most importantly, if there are differences in the total and mean resilience zone (RZ) loss between confirmed massasauga overwintering habitat and restored co-occurring overwintering habitat. If degradation of peat had occurred, the restoration could be planned in a way that minimizes the stress on the restored system. Short term project success could then be determined using mean and total RZ loss as a metric. Finally, vegetation monitoring can be used to anticipate the trajectory of the restored wetland and make recommendations for adaptive management.

3.2 Methods

3.2.1 Study area

The Eastern Georgian Bay (EBG) region is situated within the southern Boreal Shield. This landscape is flat to gently rolling, with ridges of exposed gneissic shield bedrock (Crins et al., 2009). It is a mosaic of lakes, wetlands, forested valleys, and rock barrens. The wetlands form in bedrock depressions with peat depths that vary from tens of centimeters to several meters (Didemus, 2016). The peatlands are dominated by ericaceous shrubs (*Ericaceae*), taller willows

(*Salix*), alders (*Alnus*), peatland forbs, sedges (*Carex*), ferns (*Polypodiopsida*), and trees including black spruce (*Picea mariana*), tamarack (*Larix laricina*), jack pine (*Pinus banksiana*) and eastern white pine (*Pinus strobus*). It is a landscape that experiences winters with an average of 327 cm of snow and 1090 mm of annual precipitation, mean annual temperatures of 5.8 °C, and below 0 °C monthly mean temperatures from December to March (Environment Canada, 2019).

Nine peat wetlands were selected for monitoring within this landscape. Three wetlands with confirmed or presumed massasauga observations, three without massasauga observations and three restored wetlands. The wetlands classified as confirmed are referred to as C1, C2, and C3 and were found to either have overwintering massasaugas (confirmed through visual observations) or were within 1 km of massasaugas observations and had vegetation and hummock height similar to confirmed overwintering wetlands. Conversely, wetlands that were unconfirmed overwintering habitat (U1, U2 and U3) did not have massasauga observations. Wetlands R1, R2, and R3 represent the restored wetlands that were constructed following the novel methods detailed below (section 3.2.5). The use of the term restored in this thesis implies that the habitat is somewhere along a process of restoration, whereas successfully restored denotes that the restoration project has achieved its goals. A list of site-specific characteristics can be found in 3.3.1, and methods for collection are outlined below.

Peat depth was collected with a 2 m rebar and measuring tape at ten 1x1 m quadrats (haphazardly located) within each of the nine wetlands. If hummocks occurred within the 1x1 m quadrat, hummock height was measured with a tape measure relative to the depth of the adjacent hollow. The proportion of ground cover versus vascular cover was visually estimated within each quadrat. After vascular vegetation was identified to species, the proportion of shrubs with the quadrat was calculated. Canopy cover was measured using non-hemispherical photographs taken from 1.3 m

above the ground and oriented towards the sky. These photographs were imported into Gap Light Analysis Mobile App (GLAMA, 3.0), which calculates the percentage of the photo area where the view of the sky is obstructed by branches and leaves (*i.e.*, canopy cover). The height of up to five individuals per species was collected and the average in each quadrat was taken as mean shrub height. The duration of snow cover greater than 10 cm was calculated from digitized trail camera imagery (methods in 3.2.4). The number of days in 2020 where greater than 10 of snow on the ground was filtered for each wetland. The wetland size was calculated by digitizing wetland boundaries by visual inspection using Central Ontario Orthography Project (COOP 2016) and South Central Ontario Orthography Project (SCOOP 2018) imagery with spatial resolutions of 50 and 16 cm respectively in ESRI ArcMAP (ArcGIS Desktop, release 10.4; ESRI: Redlands, CA).

3.2.2 Stockpiled peat properties

To see if stockpiling peat degraded peat properties in a way that would affect restoration outcomes, eight frozen stockpiled peat samples were collected between December of 2018, and January of 2019. A sharp metal wedge and sledge hammer were used to cut frozen peat into 30 x 40 x 50 cm cubes in the field, which were thawed and subsampled into 10 cm interior diameter PVC tubing to a depth of 40 cm. Roots were cut around the edge of the PVC casing to not compress any portion of the sample. This PVC casing was frozen and cut into 5 cm sections or "pucks" with a bandsaw. Cheesecloth was taped on the underside of the peat pucks to allow the free movements of water. An additional 6 stockpiled samples were collected to a depth of 40 cm in July and August of 2019. The stockpiled peat had been stockpiling for 3 months longer at one location in the site than in another location in the site. By sampling these two locations on two occasions, a record of how stockpiling affects peat properties over 3, 7, 11 and 14 months could be retrieved. This stockpiled peat data was compared against six PVC peat core control samples that were the same diameter

and depth as stockpiled peat cores. These control samples were from healthy leatherleaf-dominated peatlands near U1, U2, and U3 that were published in Didemus (2016).

In the lab, cut peat pucks were saturated for 24 hours in deionized water. The volume of the saturated peat was measured with a Vanier Caliper, by measuring the height of the peat at four locations relative to the height of the PVC, and the diameter of the peat relative to the PVC. This method was used to track changes in the peat volume with shrinking and swelling. The peat samples were placed onto a 56 cm diameter porous ceramic pressure plate (Soil Moisture Equipment Corp, Santa Barbara, CA) with an air entry pressure of -1 bar. To determine specific yield, a method used by Thompson and Waddington (2013) was replicated to determine the proportional water loss from saturated conditions after 10 hPa of suction was applied to the ceramic plate from a vacuum reservoir (Thompson and Waddington, 2013). The mass and volume of the peat pucks were measured after 24 hours of suction. Evaporation was limited by encasing the ceramic plates with chamber walls and a removable lid which were both misted with water and covered with plastic garbage bags.

After the 10 hPa pressure step, the pressure plates were re-saturated and ran for another 24 hours of suction at 20 hPa at which time the volume and mass was taken. This step was repeated at a pressure of 30, 40, 50, 75, 100, 150, 200 and 500 hPa to construct the soil moisture retention curve (SMRC). Volume measurements and re-saturation of the pressure plates were conducted at each pressure step as these are major causes of error for accurate determination of the SMRC (Bittelli and Flury, 2009). At pressure steps of 200 and 500 hPa, the samples were placed on the pressure plates for 48 hours before being removed and weighed to account for slow tortuous flow through smaller pores. The bulk density of the peat was determined after all other pressure steps were completed by oven drying samples at 65 °C for 48 hours. The bulk density could then be calculated

as the mass of oven dry peat (not including the mass of the PVC puck), divided by the volume of peat.

A Kruskal Wallis test and post hoc Dunn test were performed in R 3.6.1 to evaluate if the bulk density of stockpiled peat differed between stockpiling durations and between stockpiled and control peat. For each pressure step the gravimetric water content (GWC Kg³ kg⁻³) was calculated as the mass of water per mass of dry soil, and volumetric water content (VMC m³/m³) was calculated as the ratio of water volume to soil volume for both stockpiled and control peat. A log transformation was applied to all GWC and VMC data for normality. A Shapiro-Wilk normality test confirmed normality in all cases. An ANOVA and post hoc Tukey tests were used to compare mean GWC and VMC for different stockpiling time and against control for peat depths of 0-15 cm and 15-40 cm in R 3.6.1. These depth ranges were chosen as they approximate the median °C isotherm depth for the winter of 2018-2019 and 2019-2020 (Table 2.1).

3.2.3 Chlorophyll fluorescence

To quantify the moisture stress and photosynthetic ability of stockpiled samples, stockpile bryophyte F_v/F_m ratio was compared to natural bryophyte F_v/F_m using an OS-30p+ chlorophyll fluorometer (Opti-Sciences, Hudson, New Hampshire, U.S.A.) during drought conditions in the summer of 2019. The F_v/F_m is the ratio of variable fluorescence divided by the maximal fluorescence emitted when light of specific intensity and wavelength is directed at photosynthetic tissue (Hájek and Beckett, 2008). This ratio has been shown to be an indicator of plant photosynthetic performance (Baker and Oxborough, 2004; Hájek and Beckett, 2008). This measurement represents the efficiency of photosystem II and values between 0.79 to 0.89 represent theoretical maximal values for bryophytes in an unstressed state (Adams and Demming-Adams, 2004). Values that fell below the minimum threshold for measurement of approximately 0.3, were
assigned a F_v/F_m value of zero. Before measurement samples were dark adapted for 20 minutes to progressively close photosystem II reaction centers, to have maximal fluorescence upon exposure to light (Baker and Oxborough, 2004).

3.2.4 Snow depth and peat temperature monitoring

Snow depth was determined remotely with the use of photos captured on several Stealth Cam XS16 16 MP wildlife trail cameras (Stealth Cam, 2020). The cameras collected five photos a day, hourly between the hours of 10 AM and 2 PM, from December 2019 until April 2020. To have a scaled reference of height, bright orange wooden stakes with black lines in 20 cm intervals were placed in the wetlands behind *Sphagnum* hummocks. Snow depth was estimated (to the nearest cm) for nine wetlands between 31st October 2019 and 17th April 2020 in Photoshop (CC 2019) where we superimposed a scaled semi-transparent 2-m stick on the trail camera imagery. The lines on the snow stakes were used as the reference. The semi-transparent superimposed reference allowed for more detailed (\pm 1-2 cm) manual determination of snow depth. If snow depth varied among images collected on the same day from snow melt or accumulation, an average value was taken.

To understand the insulation potential of the snowpack, the standard deviation of daily peat temperature at a depth of 1 cm was divided by the standard deviation of daily air temperature (hereafter referred to as peat temperature variability). This peat temperature variability was then compared to daily snow depth and density to see if variability decreased with increasing snow cover. The Sturm et al. (2010) model for snowpack aging using snow depth data and climate classes was used to approximate snow density changes over the 2019-2020 winter. Model coefficients for boreal forest were used as these better represent this landscape than the coefficients meant for tundra, prairie, or maritime systems (Sturm et al., 2010).

At each of the 9 study wetlands, *Sphagnum* hummocks were instrumented with a vertical profile of type T thermocouple wires with an accuracy of 0.5 °C at depths of 1, 5, 10, 15, 25, and 50 cm from the surface (Omega Engineering, CT, USA). These were set to record average peat temperature every 30 minutes using three measurements during this period. The thermocouple at a depth of 1 cm was used as the sub-snowpack temperature and air temperature was measured with Campbell 107 and 109 temperature probes at a height of 1 m (Campbell 2019). A linear mixed model of log-scaled peat temperature variability as the response variable versus snow depth divided by snow density and time as fixed effects and snow depth location nested within wetland as a random effect was created in R 3.6.1. This model was calculated again separately for natural and restored wetlands without time as a fixed effect. The residuals were checked for heteroskedasticity and lack of autocorrelation.

3.2.5 Restoration of overwintering habitat for massasaugas

3.2.5.1 Overview of technique

If a disturbance will pass through known overwintering habitat the following steps can be taken to ensure the continuation of suitable overwintering habitat for co-occurring reptiles at risk in Eastern Georgian Bay. These methods outlined below are a summary of a new restoration method codesigned by Waddington, Markle and Lehan (2019).

Find or create a bedrock depression that will allow for peat depths greater than 50 cm, with a sill
or spill area that represents ≥ 25% of the wetland circumference. This depression should have a
ratio of catchment size to wetland size ratio of 11 or lower (see recommendations from 2.4.5) and
would ideally be temporarily hydrologically connected to an existing peatland, through the
restored wetlands runoff.

- a) Finding a depression assumes that disturbance, whether anthropogenic or naturally caused, has resulted in denuded vegetation and bare, or nearly bare bedrock. There will likely be water pooled in these locations.
- At the existing overwintering habitat that requires offsetting, carefully extract the upper 40 cm of Sphagnum hummocks and surface vegetation with a long arm excavator. Temporarily place this material vegetation side up in a wet depression.
- 3) Extract the lower peat material and mineral soil from the existing overwintering habitat. Transport lower material to the newly created or disturbed bedrock depression. Place and roughly level to a depth of 15 or 30 cm. Maintaining the orientation of this lower peat material is less important.
- 4) Fill the existing depression with nearby water to a level slightly below the sill.
- 5) Use the long arm excavator to gently float the upper peat hummocks patches so that there are patches of open water as well as patches of either shrub, sedge or fern dominated hummocks (Figure 3.1). A skilled operator should be able to gently nest segments of upper peat close enough that gaps between peat blocks will join as *Sphagnum* and the roots of vascular vegetation settle and gradually grow into one another. The mosaic of open water and floating peat type allows for the long-term resilience to external drivers (Markle et al., 2020b), and gives the greatest chance of creating habitat that is suitable for a co-occurring reptile assemblage (Markle, 2019).



Figure 3.1: Example of restoration design. Sedge and peat blocks represent peat material with cover that is majority sedge or shrub cover. The sill area represents the lowest point along the circumference of the wetland where water will spill when storage capacity is exceeded.

3.2.5.2 Location selection for overwintering wetlands

During fall rewetting, a technician examined possible depressions that approximated necessary depth (Table 3.1) and were within the disturbance area, to determine if water was pooling. These possible depressions were digitized as points in ArcMap 10.6. The point distance tool in ArcMap 10.6 was used with the depression locations as the input feature, a database of reptile species at risk observations as the near feature, and a search radius of 2 km. This search radius represents an approximation of the distance massasaugas, and some SAR turtles, may travel to access overwintering habitat (Parks Canada, 2015). The output of the point distance tool provided a list of the distances to SAR from each of the surveyed depressions. At each surveyed depression, two scores were assigned by calculating the square of the count of SAR divided by the average distance

of the depression to either snake or turtle SAR observations. This was done separately as some species require deep overwintering locations (SAR turtles) and others are tolerant of shallow overwintering locations (SAR snakes). These scores as well as the existing depth of the depression were used to select the best potential restored habitat locations. Three of these candidate locations with high SAR scores, that also planned to use stockpiled peat from locations where sample peat cores had already been taken (methods in 3.2.2) were selected for detailed monitoring.

3.2.5.3 Design of surface cover mosaic

The wetlands were designed to vary in depth, peat cover and depth of the lower peat substrate. This high degree of spatial variability (Table 3.1) was chosen to replicate the diversity in the percentage of open water and vascular cover types in co-occurring wetlands within the EGB region (Markle 2019). This diversity of wetland designs provides redundancy as there are multiple successful restoration outcomes if succession transitions co-occurring wetlands towards semiaquatic, or aquatic wetland types.

Approximate Area	% Sill length	Basin depth (cm)	% Sedge peat	% Shrub peat	% Open water	Depth of lower peat material (cm)
300m ²	25%	Mean = 40 cm Max = 80 cm	25	25	50	15 cm
300m ²	25%	Mean = 40 cm Max = 80 cm	35	35	30	15 cm
300m ²	25%	Mean = 40 cm Max = 80 cm	45	45	10	15 cm
300m ²	25%	Mean = 60 cm Max = 120 cm	25	25	50	30 cm
300m ²	25%	Mean = 60 cm Max = 120 cm	35	35	30	30 cm
300m ²	25%	Mean = 60 cm Max = 120 cm	45	45	10	30 cm

Table 3.1: The 6 design specifications of restored wetlands in Eastern Georgian Bay

3.2.5.4 Peat assessment

Before any movement of peat took place, a Qualified Biologist trained to assess peat properties completed an assessment of the wetland or portion of a wetland that was expected to be impacted during construction activities. The biologist made 5 to 7 random measurements of peat profile depth within the expected impact wetlands by inserting the rebar into the peat until it hit the underlying mineral soil or bedrock. At each of these locations an auger was used to establish the depth of the transition point (where fibric peat transitions to humic peat). This transition is marked by a change in colour and humification of peat material. The Von Post decomposition class (Grover and Baldock, 2013) of these two layers was recorded. The percentage cover of hummocks and hummock height relative to an adjacent hollow was also recorded in five sample locations.

After this assessment, a long arm crawler excavator with a large bucket attachment was used to cut a 1-m wide trench along the width of the wetland, on the wetland edge closest to the excavator access path. Then, using the average depth of fibric material from the peat assessment, the excavator extracted the top fibric layer (~50 cm) of peat before separating lower peat and clay organic soil mix. Great care was taken to gently place the surficial peat blocks right side up, and against adjacent peat blocks at a nearby temporary wet storage area. The lower humic peat was loaded onto a dump truck and was stored in a wet depression separately without needing to maintain the initial orientation. The initial protocol for peat extraction had been to move excavated lower peat, and surficial peat material directly into an adjacent wet depression that met the criteria for restored wetlands on the same day as excavation of the impacted wetland (section 3.2.5.1). This process would resemble the block transfer technique and would minimize the changes to ecohydrological properties of the surficial peat (Wilhelm et al., 2015). Unfortunately, operational

constraints meant that the material had to be stockpiled in temporary wet storage depression during the construction process for up to 14 months before restoration could occur.

3.2.5.5 Habitat creation

During the habitat creation phase, all overlying aggregate in the depression area was excavated down to the bedrock using a long arm crawler excavator. Wetland dimensions were chosen to approximate the dimensions in table 3.1, with restored locations closer to SAR turtle observations being longer finger-like basins and locations closer to SAR snake observations being smaller rounder depressions where possible. If portions of the base or sides of the basin were heavily fractured, the excavator bucket was to be used to heavily compact a layer of lower peat and clay. This method of compaction of lower peat and clay material could be used to make slight modifications to extend the size of the sill. Where feasible a jackhammer could be used if needed to lower an edge of the basin to create a larger sill. The sill extensions were done to avoid the potential of outflow being blocked during winter flooding. Where possible, the sill location was chosen to occur next to existing peatlands to allow it to spill into this adjacent wetland during saturated conditions.

Lower peat material depths (Table 3.1) were gently placed and levelled with the long arm excavator. Water from a nearby inlet of Georgian Bay was added to the depression so that the maximum height of the saturated humic peat was a couple cm below the level of the sill. Additional water was added until water was just under the level of the sill. Fibric peat blocks of shrub or sedge dominated peat were gently added to the peatland with the long arm excavator. The open water areas were generally in deeper sections of the basin and generally closer to the center of the peatland.

3.2.6 Continuous data collection

Continuous data collection for all nine wetlands (Table 3.2) follows the same design. A vertical profile of type T thermocouple wire with an accuracy of 0.5 °C (Omega Engineering, CT, USA) was attached to a wooden dowel and inserted to depths of 1, 5, 10, 15, 25, and 50 cm below the hummock surface at five hummock locations in each wetland. Thermocouples were set to record an average temperature onto a Campbell CR10x logger every 30 minutes based on three measurements during this period.

Water table (WT) depth was measured every 15 minutes using Solinst Level logger Junior pressure transducers (Solinst, Georgetown, ON). Loggers were placed into a 5-cm diameter groundwater monitoring well and attached by galvanized steel wire. Barometric pressure was corrected with a Solinst Edge Barometric pressure logger. This barometric pressure correction, the height of the well above the peat surface, and the string length were used to reference WT depth measurements to the moss surface.

Low pass Gaussian filtering was performed in R.3.6.1 with the "smoother" package, which reduced high frequency noise in thermocouple measurements. Temperatures were linearly interpolated between the all thermocouple depths to extract the depth to the closest cm of the -0.25 °C isotherm in MATLAB. The -0.25-degree isoline was selected to reduce the noise associated with the meltwater with near 0 °C temperatures that affects determination of the 0-degree isotherm in the spring. Obvious outliers and spurious values were manually eliminated. This method constrained the rate of temperature fluctuations to values that were thermodynamically possible. Mean and total RZ loss was calculated as the mean or total duration of time that the WT depth was above the position of the 0 °C isotherm depth for each surveyed hummock location during the overwintering period. The number of RZ loss events is the number of RZ events that occurred at

each site. The scaled RZ index is a measure from zero to one indicating the space between the water table and the bottom of the 0-degree isotherm for the entire duration of the overwintering season. A value of 1 indicated that 80 cm of overwintering space was available within hummocks for the entire winter, and a value of 0 indicates that there was no available overwintering space for the entire overwintering season. All the measures of RZ performance were assessed for differences between sites using the Kruskal Wallis test and Dunn post hoc test with bonferroni adjustment in R 3.6.1. Data was available for all wetlands from 31st October 2019 to 17th April 2020 which could have resulted in missing the beginning of massasauga ingress and end of egress (Harvey and Weatherhead 2006b; MNRF 2018).

For each of the restored wetlands, the time when WT reached a local peak position for 15 WT rise events (*e.g.* rain, snowmelt) were manually determined from a high-resolution plot of the hydrograph between 1^s November 2019 and 15th April 2020. The restored wetlands occur within two km of each other. The pattern in the timing of WT peaks was investigated at each restored wetland for paired WT events to account for differences in the timing of WT peak. The magnitude of water table rise at restored wetlands was calculated by subtracting the lowest position of the WT, 24 hours before the peak to the position at the peak for the 15 events at R1, R2, and R3. The rate of water table descent after peak was calculated using linear regression of all WT positions from the WT peak until two days after. This method was used for ease of interpretation and because the groundwater recession appeared to recede linearly (Cuthbert, 2014). The difference in timing of the WT peak between restored wetlands was calculated for restored wetlands to determine the flashiness of the restored wetlands. The values of magnitude of WT rise, slopes of WT descent, and difference in timing of WT peak for 15 paired events in 3 wetlands were investigated for autocorrelation and then checked for normality with a Shapiro- Wilks test in R 3.6.1. If found to

be normal the values were inserted into an ANOVA with post hoc Tukey test, and if transformation could not produce normality then a Kruskal-Wallis test with Dunn test and Bonferroni correction was used.

3.2.7 Vegetation surveys

In July of 2018 vegetation surveys were conducted at wetlands U1, and U2. The same survey design was used in July of 2019 at wetlands U3, C1, C2 and C3 and in July of 2020 for wetlands R1, R2, and R3. At each wetland, all the vascular and bryophyte species were identified within 10 1x1 m quadrats. For each vascular species encountered, the scientific name of the species, number of individuals, approximate cover as a portion of the vascular community, and height (cm) of 5 randomly selected individuals was recorded. If the number of individuals exceeded 100, the number of individuals was approximated. If the identity of a species could not be confirmed, a sample name was assigned, and a representative sample was taken from the wetland. These samples were later identified and properly processed as herbarium samples. Bryophyte and other ground cover scientific names were recorded, and cover (%) was approximated within the 1x1 m quadrat. Bare peat cover was recorded where *Sphagnum* was assumed to be visibly desiccated, brittle, and a capitula could not be identified.

R version 3.6.1 (R Core team, 2020) was used for all processing, analysis, and visualization. Nonmetric multidimensional scaling (NMDS) was performed with the function "metaMDS" in the "vegan" R package (Oskasen et al., 2017), which reduces dimensionality in the data. The cover of each species was passed through the metaMDS function with a Bray-Curtis dissimilarity index (Faith et al., 1987) and k = 2 dimensions. The Bray-Curtis index is commonly used within community ecology as it is suited to handle cover data where species are absent (Oskasen et al., 2017). Restricting the NMDS ordination to two dimensions allows for easier interpretation of figures. Function "Envfit" was used to pass an analysis of similarity test (ANOSIM) to determine if the average of species cover between restored and confirmed wetland as a rank distance was greater than the average of the within wetland type group rank distance (Anderson and Walsh, 2013). The ANOSIM test outputs an R statistic that if greater than 0.1 indicates significant differences (Anderson and Walsh, 2013). The significant species (p < 0.05) that influence other species' location in ordination space were plotted along with ordi-ellipses representing ordination space associated with either confirmed or restored wetlands. Complete overlap of ordi-ellipses would indicate wetlands types were identical with respect to community composition.

3.3 Results

3.3.1 Wetland properties

Restored wetlands were smaller than confirmed and unconfirmed wetlands, had lower values of canopy closure than all wetlands except for C1, and had the tallest hummocks (Table 3.2). Mean shrub cover at restored wetlands was between values of unconfirmed and confirmed wetlands, but shrub height at restored wetlands more closely resembled unconfirmed wetlands than confirmed wetlands. Peat depth at restored wetlands was shallower than all wetlands except for C3. The duration of snowpack greater than 10 cm at restored wetlands was somewhere between confirmed and unconfirmed wetlands.

Table	3.2:	Compariso	n of we	tland pr	operties	between	confirmed,	unconfirmed	and	restored
wetlan	ds. M	Iean values	are disp	layed w	ith stand	lard error	of the mean	n. "~" represe	ents e	estimated
values,	, N/A	= measurer	nent was	s not tak	en.					

Wetland	Humock height (cm)	Canopy closure %	Shrub cover %	Shrub height (cm)	Days with over 10 cm snow	Peat depth (cm)	Wetland size (ha)
U1	21.5 ±0.7	4.3	26.7 ± 1.5	33.9 ± 0.6	33	70.6 ± 3.7	0.37
U2	22.7 ± 1.0	0.5	30.2 ± 1.4	26.7 ± 0.5	27	127.2 ± 4.7	1.24
U3	26.8 ± 2.8	0	20.5 ± 1.2	20.7 ± 0.7	33	121 ± 6.3	0.40
C1	17.5 ± 1.8	13.1	3.9 ± 0.5	56 ± 4.6	71	139.2 ± 3.5	0.56
C2	28.8 ± 3.0	38.6	17.7 ± 1.9	50 ± 4.9	59	156.3 ± 4.0	0.86
C3	31.8 ± 1.8	56.2	60.1 ± 4.7	$36.5\ \pm 0.6$	55	51.2 ± 2.3	0.16
R1	35.9 ±4.4	0	20.3 ± 4.5	37 ± 2.5	40	66.1 ± 10.9	~0.03
R2	30.0 ±7.9	0	9.8 ± 2.3	22.8 ± 1.2	N/A	63.7 ± 7.8	~0.03
R3	38.1 ±4.9	0	30.2 ± 7.9	27.6 ± 1.2	52	54.8 ± 4.9	~0.03

3.3.1 Stockpiled peat properties

A Kruskal Wallis test and post hoc Dunn test revealed that the control peat had significantly lower mean bulk densities for the top 15 cm than the newer stockpiled peat (Z (n=81) = -3.99, p < 0.01), and older stockpiled peat (Z(n=81) = -3.37, p < 0.01) (Figure 3.2 a). These tests also revealed the older stockpile peat had significantly higher bulk density than the new stockpiled peat (Z (n=81) = 2.77, p < 0.05) and the control peat (Z (n=81) = -3.67, p < 0.01) at depths greater than 15 cm (Figure 3.2 b). Bulk density increased with depth from peat surface (Figure 3.3 a) and the Sy decreased with depth from the peat surface (Figure 3.3 b).

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Figure 3.2: Bulk density between control and stockpiled wetlands for the top 15 cm a) and 15 - 40 cm from the peat surface b)



Figure 3.3: a) Mean (\pm SD) bulk density (kg m⁻³) for control wetlands, newer stockpiled peat, and older stockpiled peat b) Mean (\pm SD) specific yield for new stockpile and old stockpile wetlands.

There were significant differences (F(5,184) =31.5, p < 0.001) in GWC of the top 15 cm of peat for the four durations of peat stockpiling (Figure 3.4 a). A post hoc Tuckey test revealed mean peat GWC for 3 months of stockpiling was not significantly different (6.5 ± 3.1 , p > 0.05) than control peat (8.2 ± 2.3), but GWC was significantly different than control peat after 7 months (3.3 ± 0.9 , p <0.001), 11 months (4.4 ± 2.6 , p <0.001), and 14 months (3.5 ± 1.6 , p <0.001)(Figure 3.4 c). At depths of 15-40 cm there were significant differences (F(5,183) =25.3, p < 0.001) in GWC between peat stockpiling durations (Figure 3.4 b). Peat stockpiled for 3 months, or 7 months did not significantly differ in GWC (3.9 ± 1.9 , p >0.05), (4.0 ± 2.0 , p > 0.05) respectively from control peat GWC (5.2 ± 1.31), but did differ in GWC from control peat after 11 months (2.0 ± 0.6 , p < 0.001), and 14 months (1.5 ± 0.6 , p < 0.001)(figure 3.4 d).

There are no significant difference (T(5,148) =0.79, p > 0.05) between mean VMC for control and stockpile peat from 0-15 cm (Figure 3.4 e), however, there was significant difference (T(5,159) =6.4, p < 0.001) between mean VMC for control and stockpiled peat from 15-40 cm (Figure 3.4 f). The mean VMC was not significantly different between 3 months of stockpiling (0.46 ± 0.13, p > 0.05) and control peat (0.53 +0.12), but was different for 7 months (0.40 ± 0.11, p < 0.05), 11 months (0.40 ± 0.14, p < 0.01), and 14 months (0.28 ± 0.08, p < 0.001) of peat stockpiling. VWC₋₁₀₀ increased with depth (Figure 3.5), however VWC₋₁₀₀ was greater at depth in peat locations with lower bulk density (Figure 3.3 a). Mean chlorophyll fluorescence was found to be significantly lower in stockpiled peat than control peat during summer drought conditions (Figure 3.6). Most of the observations of stockpiled peat fell below the 0.3 threshold for measurement for F_v/F_m and can be assumed to be photosynthetically inactive.



Figure 3.4: a & b: gravimetric water content of top 15 cm (a) and 15 -40 cm (b) of increasing stockpiled peat durations. c & d: Gravimetric water content of top 15 cm (c) and 15 -40 cm (d) of nearby control wetland compared to all stockpiled peat. e & f: Volumetric water content of top 15 cm (e) and 15 - 40 cm (f) of nearby control wetland compared to all stockpiled peat.



Figure 3.5: Volumetric water content at soil pressure of -100 hPa for varying peat locations types.



Figure 3.6: Chlorophyll fluorescence for stockpiled peat and natural peat during summer drought conditions.

3.3.2 Snow depth and peat temperature variability

It was found that the restored wetland R3 had greater snow cover than R1 (Figure 3.7). Both wetlands are topographically lower than the surrounding area, but R1 is located to the west of an open area, and R3 is located to the east of an open area. The snow accumulation event that occurred in mid to late February of 2020 is associated with a -8 °C storm that deposited low density snow (Figure 3.7). It is also associated with strong wind speeds measured around 11 m/s at 2 m of height at a nearby partly treed (~2km away) weather monitoring station.

It was found that when time was incorporated as a fixed effect into the linear mixed effects model, there was not a significant relation between peat temperature variability and snow depth/density (T(5) = -0.9, p > 0.05) indicating the change in daily snow depth is not related to the change in daily peat temperature variability. An empirical relation between peat temperature variability and snow depth with a nonzero slope was observed, where restored sites had greater temperature variability for equivalent snow depth values (Figure 3.8). This trend exists across all ranges of snow depth/snow density.



Figure 3.7: Snow depth for two restored wetlands (R1, R3) for the winter of 2019-2020.



Figure 3.8: Variability of peat temperature represented as standard deviation of daily peat temperature divided by standard deviation of daily air temperature versus the snow depth divided by snow density derived from Sturm (2010) model.

3.3.3 Resilience zone loss in restored wetlands

RZ loss events in wetlands characterized as confirmed, unconfirmed, or restored were found to have significant differences in median RZ loss time according to the Kruskal Wallis test ($X^2(2,830)$ = 49.7, p = 1.59 e⁻¹¹). To understand these inter-wetland differences in median RZ loss, the result of the Kruskal-Wallis was inserted into a Dunn test with Bonferroni correction (Figure 3.9). All the unconfirmed wetlands mean RZ loss time (U1 = 42.6 ± 6.7, U2 = 20.7 ± 4.1, U3 = 20.8 ± 4.9) were within the same range as the C1 wetland RZ loss time 32.8 ± 5.2. It was found that the restored wetlands median RZ loss times (R1 = 5.8 ± 7.5, R3 = 5.94 ± 6.7) resembled confirmed wetland C2 median RZ loss time (5.08 ± 14.9). The confirmed wetland C3 had the lowest median (3.5± 44.9) RZ loss than all other wetlands and was found to have fewer total RZ loss events.



Figure 3.9: Mean RZ loss duration versus wetland type. Confirmed peatland wetlands have recorded massasauga observations, unconfirmed peatlands have been surveyed and not found to have massasauga observations, and restored wetlands massasauga occupancy is unknown. Letters above boxplot denote grouping of wetlands according to a Dunn test with p < 0.05. Temperature was not recorded at R2, meaning it could not be included in the analysis.

Wetland	Total RZ loss (hours)	Group	Number of RZ loss events	Group	Scaled RZ index	Group
C1	4925	а	38	а	0.09	d
C2	91.5	с	6	b	0.34	ab
С3	9	с	3	b	0.46	a
R1	420	bc	24	ab	0.23	bc
R3	535	abc	30	ab	0.3	ab
U1	3796	ab	18	b	0.19	с
U2	4947	ab	48	a	0.1	d
U3	3539	a	43	a	0.08	d
Kruskal Wallis test statistic	$X^2(7) = 15.1, p$	= 0.03	$X^2(7) = 15.2, 1$	o = 0.03	$X^2(7) = 26.6, g$	0 = 0.0004

Table 3.3: Kruskal Wallis results comparing wetland type count, total duration of lost RZ, number of RZ loss events and the scaled RZ index.

Other measures of wetland RZ performance varied between the restored sites. R3 had a larger scaled RZ index compared to R1 (0.3 vs 0.23) respectively, but R1 had a lower total duration of RZ loss than R3 (535 hours vs 420 hours) (Table 3.3). While R3 had a higher count of RZ loss events than R1 (30 vs 24) both sites number of RZ loss events were closer to one another than any other grouping of sites. The restored sites were significantly different than the most resilient wetland in terms of the total duration of RZ loss and count of RZ loss events, and R3 was not significantly different than the C3 in terms of the value of the scaled RZ loss index (Table 3.3). However, these sites were also not significantly different from the unconfirmed sites in terms of

the total duration of RZ loss and the count of RZ loss events, but R3 had a significantly higher scale RZ loss index than all unconfirmed sites (Table 3.3).

3.3.4 Water table variability

The five wetlands with the largest snow season WT variability (between 1st November 2019 and 15th April 2020) in descending order were R3, U3, U2, U1, and C1 (Table 3.4). Only wetlands U3 and R3 would be in the top five wetlands of WT variability during the snow free period, and wetlands C3, R2 and R1 would make up the remainder of the wetlands. The depth of the WT below the peat surface did increase WT variability at R1 and C3 as reflected by the difference in standard deviations between periods (Table 3.4). There did not appear to be a pattern of WT variability and WT depth that was useful for differentiating wetlands. This can be seen in Figure 3.10 where R2 has greater WT variability than R1 despite a higher WT position (Table 3.4).

Table 3.4: Standard deviation and mean WT position relative to moss surface during snow coverseason 1st November 2019 – 17th April 2020, versus snow free season 17th April – 17th July 2020.

Wetland	WT SD snow season (cm)	Mean WT snow season (cm from moss surface	WT SD snow free season (cm)	Mean WT snow free season (cm from moss surface)
C1	5.33	4.1 ±0.03	7.57	2.15 ± 0.06
C2	3.97	8.1 ± 0.02	5.66	6.17 ± 0.04
C3	1.56	-15.2 ± 0.01	10.2	-18.2 ± 0.09
R1	3.24	-13.6 ± 0.02	8.71	-16.6 ± 0.07
R2	4.73	24.3 ± 0.03	9.34	20.6 ± 0.08
R3	10.7	-39.7 ± 0.07	9.1	-42.4 ± 0.08
U1	5.52	-27.2 ± 0.04	6.66	-29.6 ± 0.05
U2	6.37	-14 ± 0.04	6.41	-16.5 ± 0.05
U3	7.32	-0.7 ± 0.05	10.3	-15 ± 0.09

R2 is the restored wetland with the most open water and was found to have greater WT variability than R1 (Table 3.4). R2 was not found to be significantly different based on Kruskal Wallis and Dunn tests from R1 in terms of the magnitude of water table rise (Figure 3.11), or based on ANOVAs and subsequent Tukey test in terms of the rate of water table descent after a rise event (figure 3.12), or the timing of WT peak for matched WT rise events (Figure 3.13). The magnitude of water table rise, rate of water table descent, and timing of the WT peak for paired WT rise events were not found to be autocorrelated.

R3 is the only unconnected restored wetland, and it had the largest WT variability of the restored sites. R3 was found to be significantly different based on Kruskal Wallis and Dunn tests from R1 and R2 since it had a greater magnitude of water table rise (Figure 3.11), faster rate of water table descent after a rise event (figure 3.12), and earlier start of WT peak for matched WT rise events (Figure 3.13).

There were significant differences between restored sites in terms of the magnitude of WT rise $(X^2(2) = 25.49, p = 2.91 \times 10^{-6})$, linear slope of recession $(X^2(2) = 24.35, p = 5.17 \times 10^{-6})$, and timing of the WT peak $(X^2(2) = 7.60, p = 0.02)$. There was no significant difference in terms of the magnitude of water table rise between sites R1 (4.3 cm ± 2.5) and R2 (5.0 ± 4.5) (Figure 3.11), the rate of water table descent after a rise event for R1 ($-4.0 \text{ cm} \pm 1.7$) and R2 ($-4.4 \text{ cm/day} \pm 1.8$) (figure 3.12), or the timing of WT peak for matched WT rise events between R1 and R2 (difference of 57 min ± 5 hours 45 minutes) (Figure 3.13). Conversely, the unconnected wetland R3 had significantly higher WT rise ($14.0 \text{ cm} \pm 8.0$) than R1 (p < 0.001) and R2 (p < 0.01), and a greater absolute rate of WT descent ($9.6 \pm 3.4 \text{ cm/day}$) than both R1 (p < 0.001) and R2 (p < 0.001). The

difference in the timing of the WT peat was earlier for R3 than R1 (6 hours 51 min \pm 8 hours 5 mins, p < 0.05), or R2 (7 hours 48 minutes \pm 8 hours 43 minutes, p <0.01).



Figure 3.10: Mean-centered (at 0) water table (WT) for period from 2019-2020 in m for confirmed wetlands (a), restored wetlands (b), and unconfirmed wetlands (c).



Figure 3.11: Water table rise for 12 water table rise events in restored wetlands between 1^{st} November $2019 - 30^{th}$ April 2020



Figure 3.12: Linear water table decline (cm day⁻¹) after 12 winter water table rise events in restored wetlands events between 1^{st} November $2019 - 30^{th}$ April 2020.



Figure 3.13: Differences in hours between water table peak at restored wetlands for 1st November 2019 to 30th April 2020 WT rise events (rain, snow melt, rain on snow). Negative y values indicate the second wetland listed on the x axis WT peak occurred earlier

3.3.5 Vegetation surveys

Wetland	Shannon Diversity Index	Wiener	Species Richness	Sphagnum Cover %	Hummock Height (cm)
C1	1.75		20	57.7 (± 10.1)	17.5 (±1.78)
C2	1.96		29	85 (± 3.50)	28.8 (±3.03)
C3	1.7		19	57.0 (± 9.45)	29.8 (±1.84)
R1	2.07		28	18 (± 5.43)	35.9 (±4.42)
R2	1.78		21	13.5 (± 3.80)	30.0 (±7.94)
R3	1.81		17	2.79 (± 2.22)	38.1 (±4.85)

Table 3.5: Comparison of diversity, richness, *Sphagnum* cover, and hummock height between confirmed and restored wetlands. Mean values are displayed with standard error of the mean.

On average there were similar measures of diversity $(1.80 \pm 0.07, 1.88 \pm 0.07)$ and richness (22.6 \pm 2.59, 22.6 \pm 2.64) between confirmed and restored wetlands, respectively (Table 3.5). The unconnected site R3 had the lowest mean WT and the lowest species richness. The NMDS plotting species position of vascular cover was successful in converging and had a stress level < 0.2 (Figure 3.14). An analysis of similarity R statistic found evidence for slight significant differences (R = 0.085, p = 0.030) for vascular cover between confirmed and restored wetlands during the summer of 2020. It was found that *Calamagrostis canadensis* (Canada bluejoint), *Persecaria sp.* (Redshank) *and Chamadaphne calyculata* (leatherleaf) were significantly associated with restored wetlands. While many species were significantly associated with confirmed wetland types, those with the strongest relationship appear on the far-right side of the NMDS plot (Figure 3.14). These include species found to be associated with swamp ELC types in section 2.3.2 such as *Gayluccasia baccata* (Huckleberry), *Vaccinium angustifolia* (lowbush blueberry), *and Gaultheria procumbens* (Wintergreen) (Lee et al., 1998).



Figure 3.14: Non-metric Multidimensional Scaling for vascular cover. Natural wetlands are wetlands with confirmed EMR observation including wetlands C1, C2, and C3, and Restored wetlands are wetlands R1, R2, and R3. Species that had significant (p < 0.05) influence on the location of points are displayed with short form names with 4 letters from genus, 4 from species on the plot. Table 4.1 lists the full scientific and common name of these species.

The same species and types (*e.g.* bare peat, log) of ground cover was found within the restored wetlands as in the confirmed wetlands. The NMDS plotting species position of ground cover was successful in converging and had a stress level < 0.2 (Figure 3.15). There were differences in the proportions of the vegetation. An analysis of similarity found significant differences (R = 0.29, p = 0.0033) for bryophyte cover between confirmed and restored wetlands. *Sphagnum fallax* and *Sphagnum cuspidatum* were associated with confirmed wetlands, whereas bare peat and log cover was associated with restored wetlands. Ground cover species such as *Polytrichum sp.* and *Sphagnum palustre* do not appear on the plot as they were associated with both restored and confirmed wetland types. The confirmed massasauga wetlands had mean *Sphagnum* cover of 59.8 (\pm 4.1%), whereas restored wetlands had mean *Sphagnum* cover of 11.8 (\pm 2.6%). The lowest *Sphagnum* cover was found in the unconnected wetland R3 (Table 3.5).



Figure 3.15: Non-metric Multidimensional Scaling for vascular cover. Natural wetlands are wetlands with confirmed EMR observation including wetlands C1, C2, and C3 among others, and Restored wetlands are wetlands R1, R2, and R3. Species that had significant (p < 0.05) influence on the location of points are displayed with short form names (4 letters from genus, 4 from species) on the plot.

3.4 Discussion

3.4.1 Peat properties

Denser peat has lower values of specific yield meaning the WT is more variable for the same input of water (Waddington et al., 2015). This can lead to unstable WT fluctuations and the possibility of flooding massasauga overwintering habitat. Furthermore, decreased porosity means that denser peat has less gravimetric water content which will reduce the bulk latent heat source and allow for a faster descent of the 0-degree isotherm (Knox et al., 2012). Increasing the duration of stockpiling increases decomposition and bulk density of the stockpiled peat (figure 3.3 a). As macropores collapse the gravimetric water content GWC decreases over time and reduces the GWC of stockpiled peat compared to control peat.

It would be expected that the increase in decomposition would result in an increase in VMC for the top of the profile as part of the WT – decomposition feedback (Waddington et al., 2015). While there was a similar VMC between depths of 0-15 cm between natural and stockpiled peat, there were lower values of VMC in the stockpiled peat for the bottom 15 cm – 40 cm (figure 3.4 e, f). Additionally, VWC-100 or the VWC at a soil pressure of – 100 hPa was found to be lowest in older more decomposed peat than newer stockpiled peat or the control despite the bulk density being greater in the older peat. This soil water pressure of -100 hPa represents the draining of hyaline cells for *Sphagnum* (Hayward and Clymo, 1982). It is possible that macropores had developed in the stockpiled peat because of the extraction and desiccation during stockpiling. Desiccation of drained peatlands has been associated with the formation of macropores and pipe flow (Holden et al., 2006; Wallage and Holden, 2011). This process would introduce non-functional macropores throughout the peat profile that can increase hydraulic conductivity and affect water retention (Holden et al., 2006). Additionally, Liu and Lennartz (2019) suggest that macroporosity and

hydraulic parameters follow different functions with bulk density beyond a bulk density of 200 kg m⁻³. At this level they suggest that peat begins to behave more like mineral soils with respect to hydraulic properties. Much of the older stockpile peat material had a bulk density that was greater than 200 kg m⁻³ at depth (Figure 3.3a). While the uncharacteristically high Sy for the high peat bulk density will be good for dampening WT fluctuations, it remains to be seen how this may affect the succession of this restored system.

The large difference in bulk density between stockpiled and control peat is most noticeable from 15 cm - 40 cm (Figure 3.2b). It is not known whether this increase in bulk density at depth in stockpile peat is the result of more humified peat, or because of a more rapid transformation to mineral soil. The mean Sy for the entire 0 cm - 40 cm peat profile is similar between old and new stockpiled peat (0.40 vs 0.37). Data is not available for the control peatland wetlands. However, bulk density has been found to explain $R^2 = 0.63$ to 0.7 (Wilkinson et al., 2019), of the variance in hollow and hummock Sy, meaning the restored peat should have a lower specific yield than the control wetlands. Additionally, the excavated lower peat initially excavated from near the bottom of the impacted peatland is present at the surface in some areas of the restored wetland. This lower peat may be partially mixed with mineral soil and has a lower specific yield than the separately extracted hummock peat. However, the open water areas within R2, and R1 allow this wetland to have smaller winter WT fluctuations than all the unconfirmed wetlands (Table 3.4: Figure 3.10).

The greater bulk density in the restored wetland will also influence temperature variability. The decrease in pore space will increase thermal conductivity (Mustamo et al., 2019), and will lower maximal gravimetric water content. Since latent heat is an important control on the depth of the 0-degree isotherm (Knox et al., 2012), a decrease in the total volume of water at the base of the hummock would decrease this buffering potential. This temperature variability can be seen in

Figure 3.8 as the restored wetlands have greater temperature variability at a depth of 1 cm than the unconfirmed or confirmed wetlands for equivalent snow depth divided by snow density values. It would be expected that this relationship would be affected by the depth of the water table (Table 3.3). R3 does have greater distance from the top of the hummock to the mean winter WT position than any of the other unconfirmed or confirmed wetlands. However, R1 ranks far below many of the confirmed and unconfirmed wetlands for distance from top of hummock to mean winter WT position. Therefore, if this distance to mean WT were driving the differences in peat temperature variability between wetlands, one would expect an overlapping range of peat temperature fluctuations between R1 and natural wetlands (Figure 3.8).

If the bulk densities are incorporated into Tomassen et al. (2004) pH to bulk density ratio, values over 0.050 kg m⁻³ should have the potential to form floating mats, whereas those below will not have sufficient buoyancy. While the exact pH at the restored sites is not known, if we assume the addition of water from an inlet of Georgian Bay could have raised the pH as high as 6 (Georgian Bay Littoral Biosphere Reserve, 2004) and we use the median values of bulk density from the drying experiments results, this results in pH to bulk density ratio of 0.043, 0.069 and 0.011 kg m⁻³ for the older stockpile, stockpile and control peat samples, respectively. The pH and thus this ratio is likely actually lower due to the largely ombrotrophic conditions at these wetlands. Since wetlands R1, R2, and R3 are made primarily of the older stockpile peat, one would expect that as a cohesive block, the stockpiled peat should be too dense to float. Field observations made at R2 in April of 2020 found the previously stockpiled peat block to be floating and detached from the subsurface grub material.

It is possible that Tomassen et al. (2004) threshold may mean that the peat block would still float but would have a greater ratio of sphagnum beneath the water line than is optimal for production. However, it appeared on the field visit that most of the peat blocks were above the surface of the water. There are a few possible explanations. The first is that the deeper mineral organic mix from the stockpiled peat became detached and shifted the median density of the upper floating peat to be lower. The second is that the rate of methanogenesis produced in the stockpiled peat was greater than predicted in Tomassen et al. (2004) empirical model. The third is that the stockpiled samples collected were denser than the average peat block, and the fourth possibility is that the stockpiled peat had not gone through enough oxidative stress to reach a point of irreversible compressibility. If this was the case, then upon saturation the macropores swell in response to increased VMC and the true bulk density and thus pH to bulk density ratio would be greater than 0.050. Additional peat sampling and fluxes of methane may be needed to truly investigate any of these hypotheses.

3.4.2 Resilience zone

The finding that mean RZ loss time was significantly lower in the restored wetlands than the unconfirmed wetlands, and C1 (Figure 3.9) demonstrates that the restored wetland design was able to provide suitable overwintering habitat when compared to natural habitat. This is the first use of this restoration technique and these results demonstrate the suitability of this technique for future use in this region. The finding that mean RZ loss was lower in the restored wetlands with degraded peat properties than in unconfirmed wetlands healthy peat, demonstrates how well this technique buffers against temperature and WT fluctuations. The finding that C3 had significantly lower mean and total RZ loss than all other wetlands demonstrates that the restored wetlands could still be improved to better buffer against temperature fluctuation and WT rise. This is made clearer when investigating the total RZ loss, count of RZ loss events and scaled RZ index. In all the metrics of restored wetland performance, at least one of the restored sites is not significantly different from the unconfirmed wetlands (Table 3.3). The total RZ loss metric may be affected by the
performance of one less resilient monitored hummock while the count of RZ loss events may be heavily biased towards recording many short RZ loss events instead of a couple of shorter RZ loss events. For this reason, it is important to have multiple tests of RZ loss success and natural analogues to compare restored habitat against. The use of scaled RZ index also underscores this point. This metric is greater in R3 (0.3) than R2 (0.23). This difference was large enough to differentiate the group membership of these sites (Table 3.3). Since this metric measures the distance between the bottom of the 0 °C isotherm and the top of the WT, a sustained low winter water table at R3 made this wetland appear to be more suitable, despite the greater count of RZ loss events, and greater total and mean duration of RZ loss events (Table 3.3). Therefore, while mean RZ loss may be the best indicator of success of the restored wetlands, it cannot be evaluated in isolation. There is likely a need to improve site conditions to reduce the total RZ loss and number of RZ events at the restored wetlands. It remains to be seen how the growth of *Sphagnum* will change ecohydrological conditions at the restored site over the next couple of years.

Rewetting drained peatlands has found to either decrease mean bulk density of two different drained wetlands by \sim 50 kg m⁻³ over a period of only 2 - 4 years (Wilson et al., 2019), or to result in only small changes (\sim 10 kg m⁻³) in bulk density 7 – 16 years after rewetting (Urbanova and Barta, 2020). This lack of change in bulk density over time was due in part to dramatic WT fluctuations limiting new *Sphagnum* growth and microbial activity (Urbanova and Barta, 2020). Additionally, a lack of litter from vascular plants and *Sphagnum* has been found to limit methanogenesis and the development of new *Sphagnum* (Urbanova and Barta, 2020). The restored wetlands should receive an amount of litter from the current cover of vascular plants (Table 3.5), but wetland R3 may currently have WT fluctuations that are too great to promote the growth of *Sphagnum*. Additional monitoring will be needed to determine if the restored wetlands continue

to provide suitable habitat into the future, and if performance improves as more *Sphagnum* grows at the surface of the wetland.

3.4.3 Water table fluctuations and the resilience zone

Re-establishing healthy Sphagnum growth in the restored wetlands should increase porosity and specific yield (Sy) of the near-surface peat (McCarter and Price, 2014; Gauthier et al., 2018). Sy decreases with depth below the surface and results in greater water table variability with greater WT depth (Waddington et al., 2015). Since the same stockpiled peat was used for R1, R2 and R3 then the variability of the WT should vary most according to the mean WT position among wetlands. This WT depth Sy feedback can explain the greater WT variability between R1 and R3 but cannot explain the WT variability between R1 and R2. R2 was observed to have far more open water than R1 in both April and August visits. This should result in a greater wetland wide value of Sy and should dampen the fluctuation of the WT. However, the magnitude of water table rise at R2 was not found to be significantly different than at R1 (Figure 3.11). If this disparity in WT rise was accounted for by a greater proportion of run-in at R2 than R1, than one would expect R2 to have a slower WT decent, and a delayed WT peak compared to R1 as the wetland remained connected to the contributing area for longer (Figure 2.4) (Devito et al., 1996). There were no significant differences in either the timing of the WT peak, or the rate of water table descent after winter storm events between wetlands R2 and R1. It is possible that the mean Sy of 0.40 for the 40 cm profile of floating hummock peat was large enough, and conditions remained near saturated enough that the difference in the proportion of open water did not account for a large difference in variability between these wetlands.

It is possible that the restored wetland basins are not impervious to the lateral movement of water. The restored wetland may be connected to adjacent wetlands more often than when the restored wetlands are "spilling". This may help to explain why R1 and R2, which are adjacent to existing wetlands are able to maintain near-surface WT, whereas R3 is not. Another possibility is that only R3 has a leaky basin and R1 and R2 are impervious to the lateral movement of water outside of the basin. This second explanation would not explain the difference in WT variability between wetlands R1 and R2. Further investigation on the water holding capacity of these wetlands is needed. Installing piezometer nests between restored peatlands and adjacent wetlands would be difficult as the material is mostly impervious and would flow through fractures in the bedrock. Therefore, even if the piezometers could be installed, it is likely that one would miss the leakage point along the wide sill of the restored peatland. If it is thought to be necessary, one could compare stable isotopes between restored wetlands R1 and R2 and adjacent wetlands and use fingerprinting analysis (Leibundgut et al., 2011) to investigate the lateral movement of water from the restored wetlands during not spill periods.

Despite the greater variability in peat temperature fluctuations in R1 and R3 and large WT fluctuations in R3, the wetlands were still able to provide a RZ for a large proportion of the winter season. This may be in part because of the greater hummock height found in the restored wetlands compared to the confirmed wetlands (Table 3.5). Yagi et al. (2020) found that increasing the distance between hummock surface and WT depth was associated with more suitable overwintering conditions for massasaugas in disturbed peatlands in southern Ontario. The increase in physical distance buffered against greater environmental stochasticity in the degraded wetland (Yagi et al., 2020).

3.4.4 Snow accumulation and ablation and the resilience zone

Another reason that R3 may have had similar mean RZ loss in comparison to the R1 is the ability to accumulate thermally insulating snow. There is a trend seen in Figure 3.7 for restored wetlands to accumulate differing amounts of snow depending on their position relative to nearby open areas. R3 is located to the east of an open area, whereas R1 is located to the west of an open area. The large snow accumulation event that occurred in mid to late February is associated with a cold temperature snow event that would have deposited low density snow. It is also associated with strong wind speeds measured around 11 m/s at 2 m at nearby wetland C3. While the heights and density of shrub cover would be similar between wetlands, R3 is downwind of the prevailing wind direction and would be able to intercept more of the blowing snow from tall vegetation cover. Similarly, de-acceleration and snow deposition that can occur in the localized topographic depression like at the west edge of a wetland (Tabler et al., 1990).

There is currently minimal tree cover adjacent to R1 and no tree cover at R2 and R3. Tree cover is typical of smaller wetlands with confirmed massasauga observations including C3 and C2, and to a lesser extent C1. Additionally, it was determined by Markle (2019) to be among the variables associated with massasauga overwintering habitat. This lack of tree cover may lead to earlier snowmelt (Figure 2.5), which could put massasaugas at risk of freezing if followed by several days of sub 0 °C temperatures. Additionally, microhabitat selection by massasaugas within overwintering wetlands is understudied in the EGB region. As such, it may be beneficial to have restored overwintering habitat appear as similar to natural overwintering habitat for it to be chosen by massasaugas. By this same merit, minimizing the distance to suitable foraging, and basking habitat by creating sites that can be temporarily hydrologically connected to existing peatlands and adjacent to uplands may increase the likelihood that SAR will encounter and use the created

wetlands. Tracking SAR use of all restored wetlands according to methods in (OMNRF, 2018) is necessary to understand the suitability of these wetlands. These surveys are occurring at all restored wetlands for this development, but the results will be part of a separate report.

3.4.5 Vegetation composition

The WT variability at R3 could lead to succession towards an undesirable state as sustained WT fluctuations below 20 cm below the surface has been associated with a lack of bryophyte development and non-target grasses and forbs competing with obligate wetland species (Wilson et al., 2019). Restoration of wetland systems is highly disruptive and can allow for exotic and nontarget species to invade and dominate before the desired wetland functioning occurs (Cole, 1999). Within Ontario the proliferation of wetland invaders Phragmites australis (common reed), Phalaris arundinacea (reed canary grass), and Lythrum salicaria (Purple loosestrife), have been associated with reducing plant community diversity (Schooler et al., 2006; Blossey et al., 2020), reducing habitat for massasaugas (WDNR, 2013), and being resistant to biological control (Robinchaud and Rooney, 2020). The use of large blocks of stockpiled peat with vascular species and roots present likely help to buffer against encroaching non-target species. By ensuring target species establish first, "priority effects" can ensure that these species will alter biotic and abiotic conditions in a way that directs succession to the desired state (Fukami, 2015). These early establishing species will use available niche space, instead of leaving it vacant to disturbance tolerant species (Hess, 2019).

Wetland plant species have an advantage against emergent and terrestrial species in that they can germinate in lower oxygen and lower light conditions (Casanova and Brock, 2000). Wetland species seeds have low persistence within the soil seed bank because prolonged anaerobic conditions slow the decomposition of seeds. Under aerobic conditions, seedling competition between target and non-target wetland species increases as non-target species are better equipped to germinate and persist (Klimkowska et al., 2010). Stockpiling the peat material increased the duration of aerobic conditions, and likely reduced the propagation of target vascular species. Stockpiling organic soil has been found to lead to the degradation of soil seed banks (Buss et al., 2020), reduce species richness, while increasing weedy annual species abundance (Dhar et al., 2019), and reduce mycorrhizal associations associated with greater growth rates in ericaceous shrubs (Fadaei et al., 2020).

The invasive species *Persecaria maculosa* is a facilitative wetland invasive species found in R1, R2 and R3, but not in any of the 40 monitored natural wetlands near the wetland area. It prefers moist to wet conditions but displays an abundance of trait plasticity (Herchel et al., 2004). It is tolerant to periods of dryness and will reproduce vigorously in resource rich environments (Sultan, 1996). In addition, it has a moderate to long seed bank persistence and has been found to persist in the seed bank for 8 or more years (Burnside et al., 1996). Fortunately, this species does not tolerate highly acidic soils (CDFA, 2003). As *Sphagnum* cover increases within the restored peatlands, the pH of upper peat should decrease as *Sphagnum* creates conditions to facilitate its growth (van Bergen et al., 2020). This natural allelopathy could be combined with manual picking of *Persecaria maculosa* shoots within the wetland. Seed set for *Persecaria sp* occurs in late summer, meaning the shoots should be removed in midsummer to avoid spreading seeds and to reduce the amount of seeds that will make their way into the seed bank (DiTomaso et al., 2013).

The other invasive species *Cirsium arvense* is a perennial species that was only found in a small portion of the drier and unconnected restored wetland R3. *Cirsium arvense* is a noxious weed that can be spread by small root fragments or by seed (Stolcova, 2002). These root fragments can withstand freezing, thawing and drying and the seeds are viable for 20 years within the soil (Holm

et al., 1991). *C. arvense* is a facilitative upland species that shows signs of physiological stress during sustained high WT conditions including leaf loss and an inability to germinate (Lenssen et al., 1998). This wetness intolerance may be why *C. arvense* was able to persist only at the driest restored wetland. This also indicates why restoring a near-surface WT is critical not only for the growth of *Sphagnum sp.* but also to prevent any further growth or encroachment of non-target species.

The method of restoration used in this study was not designed to be used after an extended period of stockpiling. It is unclear if the current lack of cover of Sphagnum sp. will prevent the restored wetlands from reaching a greater cover of Sphagnum in all non open water areas. The loss of photosynthetic potential (Figure 3.6), was found to be largely irreversible as only 11.8 (\pm 2.6% s.e.) of ground cover could be classified as living Sphagnum eight months after rewetting. This is five times less Sphagnum cover as the confirmed wetlands $59.8 \pm 4.1\%$ (s.e.). This is potentially problematic as González et al. (2013) found bare peat cover to be significantly associated with failed restoration projects when evaluating species associations in 34 recently restored peatlands in Eastern Canada. While hummocks are physically present in the wetlands, photosynthetically active Sphagnum is largely absent from the wetlands except for in isolated patches. Further intervention to hasten the spread of live Sphagnum may be required. A modification of the moss layer transfer technique (Quinty and Rochefort, 2003) could be used for these restored wetlands. Spreading Sphagnum diaspores over bare peat areas could be conducted in the fall or spring when the WT is high and evaporative stress is low. Areas with existing moderately dense vascular cover may be sufficient to facilitate the high humidity and moisture conditions required for early growth and will likely not need any mulch cover (Rochefort et al., 2007; Rochefort et al., 2003). By contrast, bare peat areas with sparse or no vascular cover could be covered with a light cover of on-site mulch material instead of the off-site straw mulch suggested by Quinty and Rochefort (2003). Given that this method produced habitat with mean RZ loss similar to confirmed habitat with degraded plant material, it should be capable of fewer and shorter RZ loss events with pristine material. Overall this new system of restoring overwintering habitat for reptiles in Goergian Bay showed great promise, and can be incorporated as a standard wetland offsetting methods for this region.

CHAPTER 4: SUMMARY AND CONCLUSIONS

4.1 Summary and conclusions

The Eastern Georgian Bay (EGB) region has large expanses of connected habitat. It is a reptile hotspot within Canada and contains 50 species at risk in Ontario. These species are threatened by habitat loss and fragmentation, road mortality, human persecution, animal trafficking and increased environmental stochasticity because of climate change. The eastern massasauga rattlesnake is threatened or endangered across most of its range, and the EGB region subpopulation is listed as Threatened on both the Species at Risk Act, and Species at Risk in Ontario. Under the Endangered Species Act, habitat features associated with overwintering, and gestation of massasaugas have the lowest tolerance to alteration because of the importance of these habitats for species survival. Overwintering habitat is limited across the landscape and massasaugas have strong fidelity to return to within 100 m of the same overwintering location as the previous year . Overwintering habitat needs to provide sufficient moisture, stable above 0 °C temperatures, and physical space above the water table during flooding events. Different subpopulations meet this requirement by using different habitat features including rock crevices, crayfish burrows, rotten tree roots, and Sphagnum hummocks. The EGB region subpopulation uses Sphagnum peat hummocks for overwintering. Within the hummocks, the volume of unflooded space where temperatures remain above 0 °C can be conceptualized as a resilience zone (RZ). A development with the EGB region has necessitated the creation of overwintering habitat for massasaugas. While an overwintering habitat design for snakes has been created by the Toronto Zoo, this design more closely resembles a rock crevice, than the Sphagnum hummocks that the EGB region subpopulation utilizes. Therefore, this thesis provides an ecohydrological functioning

understanding of the *Sphagnum* hummock features used by the EGB region subpopulation of massasaugas to properly restore this habitat features.

In Chapter 2, ecohydrological factors were compared against the suitability of hummock habitat, to obtain an understanding of the factors that lead to suitable overwintering habitat. It was known that taller hummocks and protection from flooding events are associated with confirmed massasauga observations. However, hummocks of similar heights provide different amounts of total RZ space and remain suitable for differing amounts of time over the winter. Peat wetlands in the EGB region with and without massasauga observations were instrumented to measure WT fluctuation and soil temperature in order to get a measure of mean and total RZ loss. The height and cover of shrubs, difference in canopy closure, catchment to wetland ratio, peat depth, depth and duration of snow cover, hummock height and WT variability were compared to mean and total RZ loss. To account for multicollinearity, the combination of variables that resulted in variable inflation factors below 5 were used in a factor analysis. Significant (p < 0.05) factors were combined into mixed linear models in R with nested random effects as site and hummock location, and response variable as either mean or total RZ loss. It was found that models with the lowest AIC had negative coefficients of hummock height and shrub height and a positive coefficient of WT variability. The model of total RZ, more heavily weighted hummock height in comparison to shrub height or WT variability than the model of mean RZ. Taller shrub heights were associated with greater snow cover from observations in remote cameras and manual measurements. Canopy cover was associated with a longer snow duration at wetlands. Both canopy cover and tall shrub heights are associated with swamp type sites according to the ELC classification framework These sites contain black huckleberry, winter green, taller shrubs, and a greater cover of taller jack pine, and tamarack trees. An increasing ratio of the catchment to wetland size was associated with slower

WT recession time which could increase the amount of run in and possibility of flooding at sites. However, wetlands that can effectively shed this water or have tall hummocks like C2 may still be able to have suitable overwintering conditions despite a large volume of run-in after storm events.

These observations of suitable overwintering habitat from Chapter 2 were incorporated in the design of suitable overwintering habitat for massasauga and other reptiles at risk in the EGB region in chapter 3. The habitat was designed to resemble wetlands with confirmed observations of massasauga and SAR turtle species This included twice the number of habitat patches per hectare of trees, shrubs, open water, and marsh species than habitat associated primarily with massasaugas The restored wetlands had 6 different design specifications. All wetlands were approximately 300 m² and varied in basin depth, depth of lower peat material and the proportions of sedge dominated peat, to shrub dominated peat, to open water. Due to unplanned circumstances the upper peat had to be stored and as a result desiccated over a period of 11 - 14 months. This significantly increased the bulk density for the surficial layer of stockpiled peat which may have resulted in greater temperature fluctuations for equivalent levels of snow depth divided by density in the restored wetlands versus the natural wetlands. After the first winter the restored wetlands were found to have similar levels of mean RZ loss, total duration of RZ loss, count of events and scaled RZ index compared to the confirmed sites. However, in all measures of success aside from mean RZ loss, the restored sites were not significantly different from the unconfirmed sites. The restored wetland R3 had a large WT depth which led to more dramatic WT fluctuations that would flood overwintering habitat. This may be as a result of this site being disconnected from the surrounding wetlands and having a more impervious upland surface than wetlands R2 and R1. The vegetative composition of R1, R2 and R3 were found to be significantly different than in the confirmed wetlands C1, C2 and C3. Within the vascular layer this difference may be due to the marshy species

associated with open water in the restored site and the presence of a couple undesirable species. In the ground layer, the proportion of bare peat differentiates the restored sites from confirmed sites. This is problematic as large cover of bare peat in restored peat wetlands has been associated with unsuccessful restoration outcomes Adaptive management measures like *Sphagnum* diaspore transplants using a modification of the moss layer transfer technique can be used to increase *Sphagnum* cover at these peatlands. When *Sphagnum* cover does increase, then peat specific yield should increase over time which would both reduce water table fluctuations and decrease the thermal conductivity of the upper peat, because of increases in *Sphagnum* macroporosity.

The methodology behind the design of overwintering habitat as well as the ecohydrological determinants of success, represent important tools that restoration practitioners can use in designing other overwintering habitat for species at risk. While having a similar vegetative community composition between restored and confirmed wetlands may ensure that similar feedbacks are occurring, it does not ensure proper ecohydrological conditions for species that thermoregulate via habitat selection. Conversely, if habitat is designed to provide suitable thermal and hydrological conditions necessary for occupancy but does not resemble habitat that the subpopulation is accustomed to, then it is possible that the restored habitat may not be used. The habitat must be designed to resemble confirmed habitat and have ecohydrological properties that resemble confirmed habitat. The habitat should be constructed close to monitored species locations, especially if the species has a high fidelity to overwintering habitat. Since species like massasauga spend up to weeks near the overwintering location prior and after overwintering, the restored locations should be close to suitable basking and foraging habitat. Monitoring for both the ecohydrological conditions of the overwintering habitat as well as for SAR occupancy must be done multiple times after construction in order to evaluate project success and to guide adaptive

management measures. When restoration projects are designed and constructed properly, they have the potential to not just offset, but increase the quantity and quality of overwintering habitat across the landscape. Overall, this method of restoration shows promising results, and has the potential to become standard practice for wetland offsetting in this region.

6. References

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Appendix A

Table A.1: Species short form, scientific and common names, for species in Figures 2.5 and 5.14				
Short form name	Scientific name	Common name		
Alnuinca	Alnus incana	River alder		

Alnuinca	Alnus incana	River alder
Aronmela	Aronia melanocarpa	Black chockberry
Calacana	Calamagrostis canadensis	Canada bluejoint
Carebrev	Carex brevoir	Oval sedge
Careolig	Carex oligosperma	Few seeded sedge
Chamcaly	Chamaedaphne calyculata	Leatherleaf
Cladsp	Cladonia sp.	Reindeer lichen
Eriovirg	Eriophorum virginicum	Tawny cottongrass
Gaulproc	Gaultheria procumbens	Wintergreen
Gaylbacc	Gaylussacia baccata	Black huckleberry
llexvert	llex verticillata	Winterberry
Irisvers	Iris versicolor	Northern blue flag
Kalmangu	Kalmia angustifolia	Sheep laurel
Kalmpoli	Kalmia polifolia	Bog laurel
Maintrif	Maianthemum trifolium	False soloman seal
Onocsens	Onoclea sensibilis	Sensitive fern
Osmurega	Osmunda regalis	Royal fern
Perssp	Persicaria sp.	Ladysthumb
Rhamalnu	Rhamnus alnifolia	Alder-leaf buckthorn
Rubuhisp	Rubis hispidus	Swamp dewberry
Salidisc	Salix discolor	Pussy willow
Spirtome	Spiraea tomentosa	Hardhack
Typhangu	Typha angustifolia	Narrow leaf cattail
Vaccangu	Vaccinium angustifolium	Lowbush blueberry
Vaccoxyc	Vaccinium oxycoccos	Bog cranberry
Woodvirg	Woodwardia virginica	Virginia chain fern