Implications for management of eastern larch beetle and emerald ash borer and their impact on lowland forest types across the Great Lakes region

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Abstract

Climate change and insect disturbances pose a great challenge to forest management in the Great Lakes region. Here, I evaluate some of those challenges with an invasive insect, emerald ash borer (*Agrilus planipennis* Fairmaire[EAB]) and a native insect eastern larch beetle (*Dendroctonus simplex*[ELB]). *EAB* is remarkable in the level, intensity, and extent of mortality it has caused in ash species in North America. Within the Great Lakes, it threatens to induce site conversion in northern black ash wetlands. This research identifies potential replacement species with adequate cold and flood tolerance to plant in such sites by analyzing survival and growth metrics in an experimental greenhouse and nursery study in Wisconsin. The impact of *ELB* on its primary host tamarack has become an emerging concern due to an increase in epidemic level outbreaks, largely attributed to warmer temperatures and longer growing seasons, which allow for increased reproduction. This research sought to explore the detection and dynamics of ELB in north central Wisconsin through observational studies. Ground surveys of two methods of detection, aerial sketch map surveys and a satellite imagery remote sensing algorithm, were conducted to assess forest mortality in these detection areas. It was determined that in this region of Wisconsin, detection methods are likely underperforming due to the variable and heterogeneous nature of tamarack forests. Fixed radius plot inventories were also conducted in randomly selected tamarack stands and stands identified via detection methods to evaluate relationships between ELB and tamarack structure and composition. Several significant relationships were discovered between ELB and tamarack structure and composition, such as between ELB and stand density, percent host species composition, and small tamarack tree

presence. This knowledge can be implemented by forest managers across the region to assist in efforts to manage forests appropriately in the face of climate change and insect disturbances

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Chapter 1: Introduction

Northern black ash (*Fraxinus nigra* Marsh.) and tamarack (*Larix laricina* (Du Roi) K. Koch) wetlands are two forested systems in the Great Lakes region that are facing novel threats from forest insects that may be exacerbated by climate change. In this region, the majority of these forest types are found in the Laurentian mixed-forest ecoregion. This region serves as a transition zone between the boreal forests to the north and the temperate forests to the south. Under current climate change predictions, the northward expansion of the temperate forest, particularly species with a more southern range, could be expected (Janowiak et al., 2014). Both black ash and tamarack are nearing the southern extent of their range here and the USFS Tree Atlas suggests that both of these species have a poor capability to adapt to predicted changes in climate (Iverston et al., 2012; Matthews et al., 2011; Peters et al., 2020). The dual threat of forest insects and climate change poses a potential risk to the persistence of these systems.

Forest diseases and pests are considered one of the major stressors to forest ecosystems in this region (Janowiak et al., 2014). One study has found that even moderate disturbances, like ones from forest insects, could accelerate transition dynamics of forests under a changing climate in this ecoregion (Brice et al., 2020). Forest insect threats here are not limited to invasive insects, but also include novel developments with native insects. While northern black ash wetlands are threatened by the invasive wood boring beetle, emerald ash borer (*Agrilus planipennis* Fairmaire[EAB]), tamarack is threatened by a native bark beetle, eastern larch beetle (*Dendroctonus simplex*[ELB]).

Emerald ash borer poses a significant threat to northern black ash wetlands due to its high mortality rate and black ash's role as a keystone species in that ecosystem (Ellison et al., 2005; Youngquist et al., 2017). A lack of natural regeneration post invasion could result in the loss of forest cover and site conversion to non-forested wetland, posing a need for silvicultural intervention (Palik et al., 2021a; Slesak et al., 2014; Windmuller-Campione et al., 2021). Potential alternative species for artificial regeneration in this system have been identified, but gaps in knowledge remain to the viability of these species to withstand the cold, wet environmental conditions black ash are found in (Iverson et al., 2016; Palik et al., 2021b; Looney et al., 2015).

The threat that ELB poses to tamarack is less understood. A recent, unprecedented outbreak of the beetle in Minnesota has generated concern, as historically, the beetle was not considered aggressive (Langor and Raske, 1989; Mckee et al., 2022). This outbreak is largely attributed to climate changes, which allow for an increased rate of reproduction (Mckee et al., 2022; Venette and Walter, 2012). This unprecedented outbreak creates a need to have accurate methods of identifying and monitoring populations of the beetle across the Great Lakes region. Currently, aerial sketch map surveys are primarily used. However, these surveys have their limitations, and it has been reported that stands identified by these surveys as having no ELB mortality had low levels of mortality, and that stands with ELB mortality often had pockets of healthy tamarack in them (Shaunette, 2022).

Furthermore, relationships between ELB and tamarack mortality, regeneration, structure, and composition need to be further explored. Identifying these relationships may lead to understanding risk factors for ELB infestation and also potential impacts to tamarack systems. While studies in Minnesota have been able to examine some of these relationships, no studies have been conducted in other areas of the Great Lakes region, like in Wisconsin or Michigan. Regional differences in forest landscapes, climate, topography, and other environmental factors could result in varying interactions between ELB and tamarack.

There are three main goals for this research. The first is to identify potential replacement species for black ash in northern black ash wetlands that have adequate cold and flood tolerance. For this, a selection of species were outplanted at three nurseries across a latitudinal climate gradient in the state of Wisconsin, some of which were subjected to flooding treatments. The second goal is to explore methods of detection for ELB by analyzing ground surveys of ELB disturbance detected by aerial surveys and by a remote sensing satellite imagery algorithm. The third goal is to identify relationships between ELB and tamarack mortality, regeneration, structure, and composition in north central Wisconsin. The results of this research will provide insights valuable for managing forest health challenges across the Great Lakes region.

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Chapter 2: Evaluating Cold and Flood Tolerance for Black Ash Replacement Species in the Great Lakes Region

Introduction

Since its introduction to lower Michigan, emerald ash borer ((EAB) *Agrilus planipennis* Fairmaire)) has caused considerable damage to ash (*Fraxinus* spp.) forests across the United States. Black ash (*Fraxinus nigra* Marsh.) forests are particularly vulnerable to this invasive wood-boring beetle, because black ash trees often compose a high proportion of the basal area in stands (>75%) and EAB's high mortality rate could result in a significant loss in tree cover (Ellison et al., 2005; Klooster et al., 2014; Youngquist et al., 2017). Studies have suggested that there may be insufficient natural regeneration from other non-ash species after significant black ash mortality, which poses a need for silvicultural treatments, like artificial regeneration, to maintain forest cover (Palik et al., 2021a; Slesak et al., 2014; Windmuller-Campione et al., 2021). Several potential replacement species have been identified and evaluated through prior studies (Iverson et al., 2016; Looney et al., 2015; Palik et al., 2021b). However, the cold tolerance of many of these species is uncertain. Further, it is less known how the stress of flooding affects cold tolerance, influencing individual species growth and survival. This study observed 18 different tree species across three flood levels and three cold hardiness zones. The goal of this study was to quantify survival and growth for each species over multiple years at each of the three cold hardiness sites to provide insights about the interaction of flooding and climate.

Methods

Starting in May of 2020, 10 seedlings of 18 different tree species (Table 2.1) were received from the Wisconsin DNR nursery (Boscobel, WI) with the exception of American sycamore (*Platanus occidentalis* L.) and bald cypress (*Taxodium distichum* (L). Rich) seedlings, which were sourced from Iowa and Illinois nurseries, respectively. Species were selected by Wisconsin DNR silviculturists for either being known to inhabit poorly drained areas throughout Wisconsin, or being species found in poorly drained areas just south of Wisconsin. The selected nursery species varied in age (1-3 years) between species, but within each species age was the same. In spring of 2020 a set of control seedlings (10 of each species) were out-planted directly into one of three Wisconsin DNR nurseries. Control seedlings did not undergo any flooding treatments. The nurseries where the seedlings were outplanted are located, from north to south, in Hayward, Wisconsin Rapids, and Boscobel in Wisconsin. Average January temperatures for these locations are -11.0°C, -9.55°C, and -6.94°C, respectively (NOAA).

Seedlings for the flood experiments were planted into pots measuring 10 x 10 x 30 cm, using a bagged soil mixture (Berger BM7 bark mix). Potted seedlings were placed into large stock tanks that were assigned one of three different flooding treatments. Flooding was maintained such that trees either had 27 cm, 14 cm, or 0 cm of aerated soil. The trees and treatments were kept inside two large greenhouses that were maintained at 25 ℃ throughout the growing season. This outplanting experiment builds from experimental flooding treatments which are described in detail in Keller 2022 and Keller et al. 2023.

Table 2.1 Species used in greenhouse experiment and nursery outplanting to evaluate cold and flood tolerance for replacing black ash

Scientific Name	Common Name	Species Code	Stock Tyoe
Acer rubrum L.	red maple	ACRU	2-0
Acer saccharinum L.	silver maple	ACSIL	2-0
Acer saccharum Marshall	sugar maple	ACSUG	2-0
Betula nigra L.	river birch	BENI	1-0
Betula alleghaniensis Britton	yellow birch	BEAL	2-0
Carya cordiformis (Wangenh.) K. Koch	bitternut hickory	CACO	2-0
Juglans nigra L.	black walnut	JUNI	1-0
Larix laricina (Du Roi) K. Koch	tamarack	LALA	2-0
Picea glauca (Moench) Voss	white spruce	PIGL	3-0
Picea mariana (Mill.) Britton, Sterns & Poggenb.	black spruce	PIMA	3-0
Pinus resinosa Aiton	red pine	PIRE	2-0
Pinus strobus L.	white pine	PIST	3-0
Platanus occidentalis L.	sycamore	PLOC	1-0
Quercus alba L.	white oak	QUAL	1-0
Quercus bicolor Willd.	swamp white oak	QUBI	1-0
Taxodium distichum (L.) Rich.	bald cypress	TADI	1-0
Thuja occidentalis L.	northern white cedar	THOC	3-0
Ulmus americana L.	American elm	ULAM	1-0

In the fall of 2020, all living trees from the flooding experiment (~73% of the starting trees) were out-planted in nurseries alongside the control seedlings that were out-planted in the three nurseries in spring of 2020. The same process of planting control seedlings in spring and conducting a new series of flooding experiments throughout the growing season was repeated in 2021. Similar to 2020, all surviving flood treatment seedlings were planted alongside the control seedlings at each of the three nurseries in fall of 2021. Seedlings were planted in rows with approximately 0.5 m between trees within row and approximately 4 m spacing between rows.

Each seedling was assigned a metal tag with an identification number. The care of seedlings was at the discretion of each nursery, and thus varied among nurseries and across years (personal communication, D. Bronson). Growth and survival measurements were taken in the fall of 2021 and 2022, allowing for up to two years of monitoring for 2020 planted trees and one year of monitoring for 2021 planted trees. At each seedling, a tag was searched for and height was measured using a meter stick. Measurements were made along the axis of the seedling and measured to the uppermost live woody tissue and rounded to the nearest centimeter. It was also noted at each seedling whether it was alive or dead, and if it appeared browsed.

If a tag was found unattached to a tree one of three courses of action occurred. If it was located near a tree with a missing tag and the tags/trees surrounding it were in numerical order, that tag was attributed to that tree. If it was not in numerical order, the tag number and the height and species of the suspected owner were written down and reviewed later when organizing data, to ensure the missing tag matched the previous data collected for the tree. If a tag was found in a row in 0.5 meter spacing with no tree (where a tree would have been planted), it was presumed dead. If a tree did not have a tag attached or we could not find the tag in a reasonable amount of time, species and height were recorded for later review when organizing data. Summary statistics were created for average height growth and survival of control seedlings at the two (2021 outplanting) and three year marks (2020 outplanting). Each year represents a full growing season. So seedlings of the three year mark went through a growing season in 2020, 2021, and 2022 before the most recent measurement.

Summary statistics were also generated for a subset of eight species for the growth and survival of control and treated seedlings at the 3 year mark. The subset of eight species included red maple (*Acer rubrum* L.), silver maple (*Acer saccharinum* L.), river birch (*Betula nigra* L.), yellow birch (*Betula alleghaniensis* Britton), bald cypress, American sycamore, swamp white oak (*Quercus bicolor* Willd), and American elm (*Ulmus americana* L.). Three of these were chosen because they were identified as the best performers in the original flood intensity treatments, those were river birch, sycamore, and bald cypress (Keller et al. 2022). These also happened to be species with a more southern range than some of the nursery locations. The other species were chosen because they were identified as having a moderate tolerance to flooding conditions in the original greenhouse study (Keller et al., 2022).

Results of percent height growth (**Appendix 2B**) between the treated and control seedlings of each species (subset of eight) at each location were tested for significant differences using one way Anova tests (p-value of 0.05) followed up by Fisher's least significant difference with a bonferroni p-adjustment. Results of survival between treated and control seedlings (subset of eight) of each species at each location were tested for significant differences using Fisher's exact test for count data.

Results

Survival of Control Seedlings

Rates of survival varied considerably across planting years and locations for many of the species in the control group (Figure 2.1). However, each nursery had species with generally high

(≥50%) and generally low (<50%) survival across planting years. Average survival across all species planted in 2021 was 59.9%, 45.4%, and 61.5% at Hayward, Wisconsin Rapids, and Boscobel, respectively. Average survival for species planted in 2020 was 63.7%, 56.7%, and 67.2% at Hayward, Wisconsin Rapids, and Boscobel, respectively.

At the northernmost nursery location in Hayward, silver maple stood out as having consistently low survival in both planting years. Species that had consistently high survival across planting years at Hayward included yellow birch, river birch, bitternut hickory (*Carya cordiformis* (Wangenh) K. Koch), tamarack (*Larix laricina* (Du Roi) K. Koch), white spruce (*Picea glauca* (Moench) Voss), red pine (*Pinus resinosa* Aiton), white pine (*Pinus strobus* L.), and swamp white oak.

The nursery in Wisconsin Rapids was at our middle latitude location. White spruce, black spruce, red pine, white pine, tamarack, yellow birch and river birch had generally low survival across planting years. Silver maple, bitternut hickory, white oak (*Quercus alba* L.), swamp white oak, sycamore, and American elm had high survival in Wisconsin Rapids.

At our southernmost location in Boscobel, yellow birch, tamarack, black spruce, and red pine had low survival. American elm, bald cypress, swamp white oak, white oak, bitternut hickory, river birch, sycamore, red maple, silver maple, and sugar maple (*Acer saccharum* Marshall) all had generally high survival at the southernmost location.

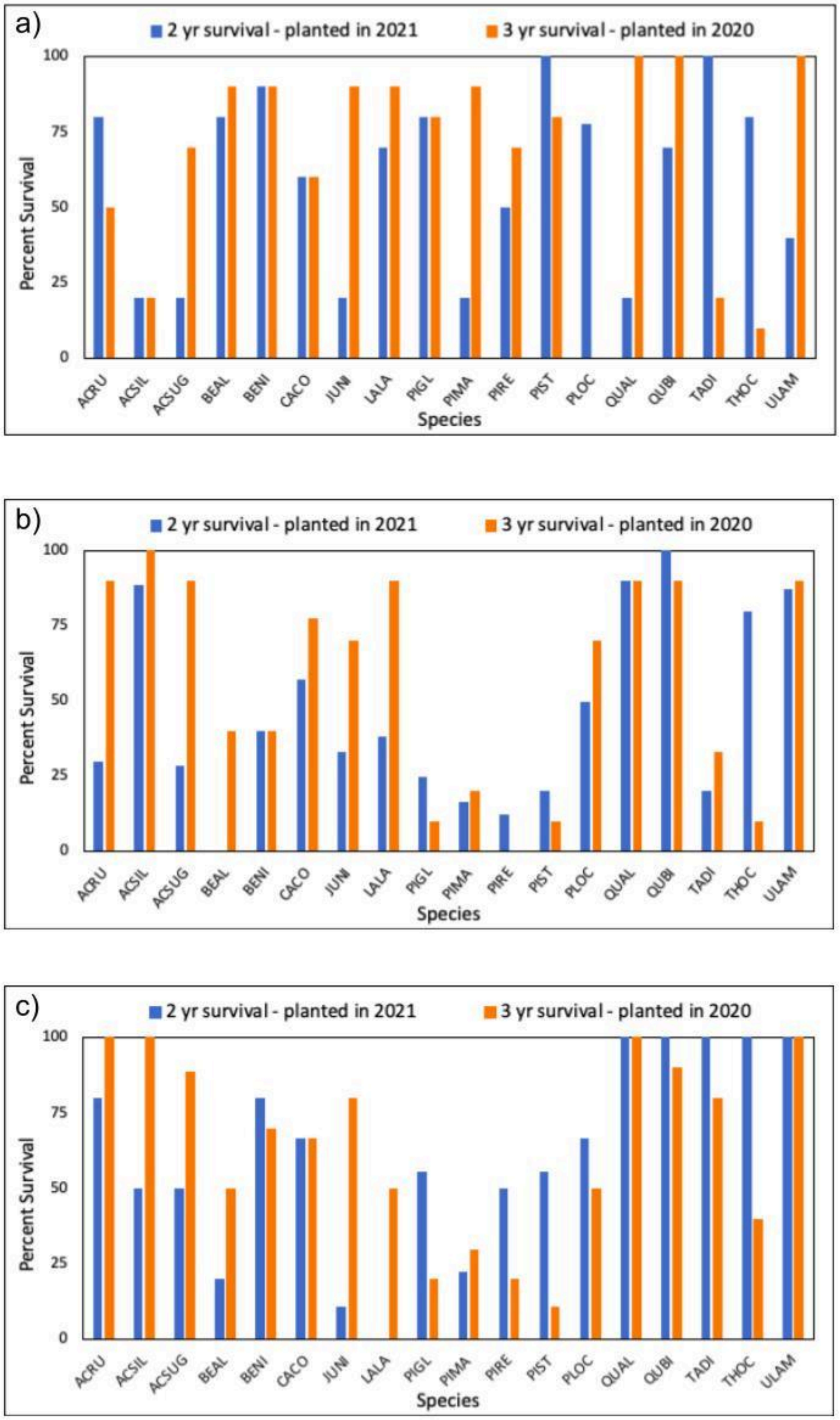

Figure 2.1 Two and three year percent survival of control seedlings planted in 2021 (blue bars)

and 2020 (orange bars) in nurseries at Hayward (a), Wisconsin Rapids, (b), and Boscobel (c)

Height Growth of Control Seedlings

As with survival, there is a similar level of variation with height growth between nurseries (Figure 2.2). Additionally, a species that had high survival might have had limited height growth and vice versa. For example, bald cypress had high survival in the southernmost site, but did not have high height growth. Some species, like river birch and silver maple, had higher growth across all nursery locations when compared to other species. Species like black walnut, black spruce, and bald cypress showed lower growth across sites compared to other species. One species that stood out was sycamore. It had the highest average height growth in Boscobel compared to any other species at any other site.

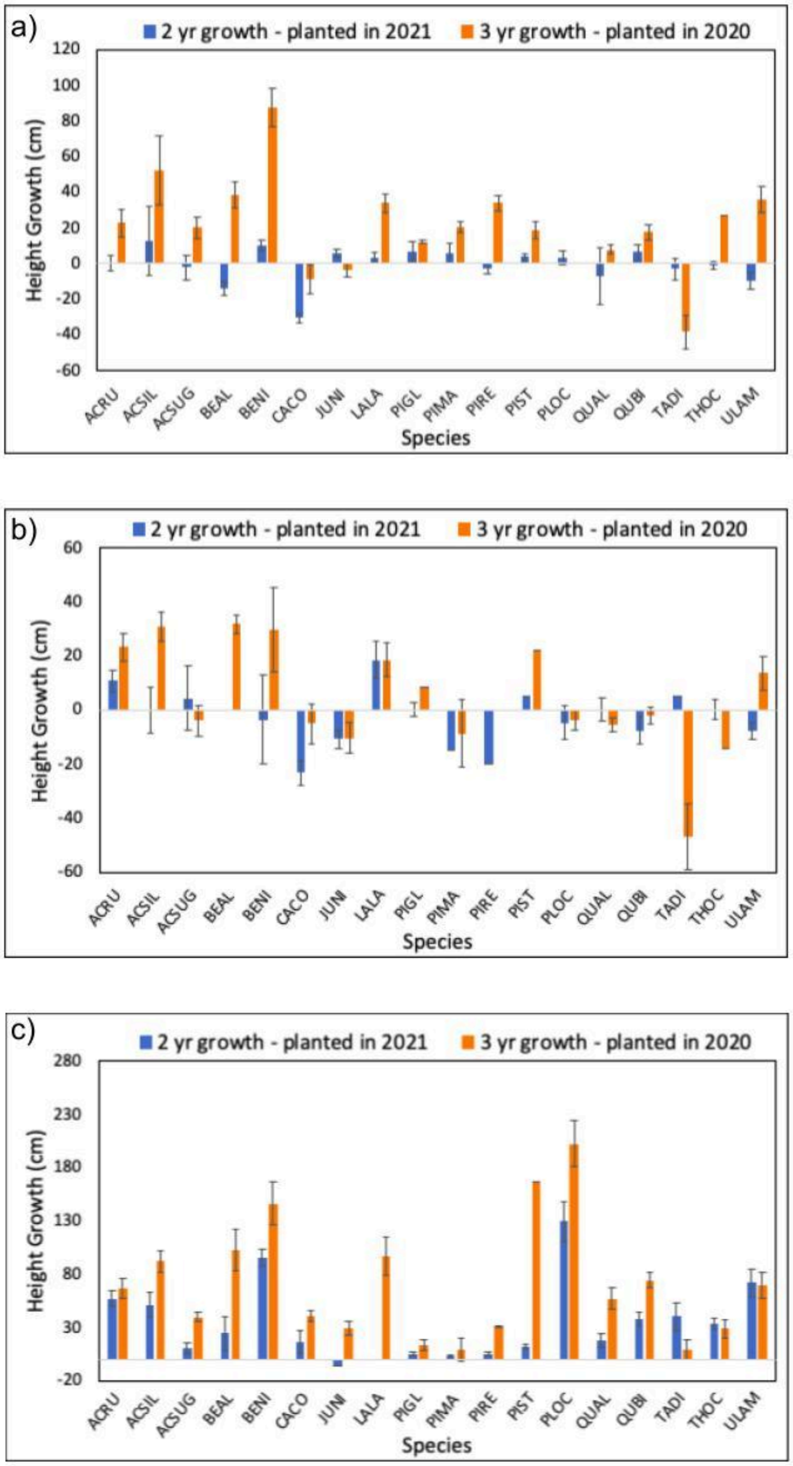

Figure 2.2 Two and three year average height change of control seedlings planted in 2021 (blue bars) and 2020 (orange bars) in nurseries at Hayward (a), Wisconsin Rapids, (b), and Boscobel

(c) with standard error bars.

Survival of Treated and Control Seedlings

When looking at the three year survival (planted in 2020) of the subset of eight species, several, like American elm (Figure 2.3), red maple, and swamp white oak showed generally high survival across nursery locations and treatments. Survival graphs for all species can be found in (**Appendix 2A)**. There were no significant differences between treated and control seedlings at any nursery location for American elm, but there were some significant differences found at Hayward for swamp white oak and red maple and at Wisconsin Rapids for swamp white oak (Table 2.2).

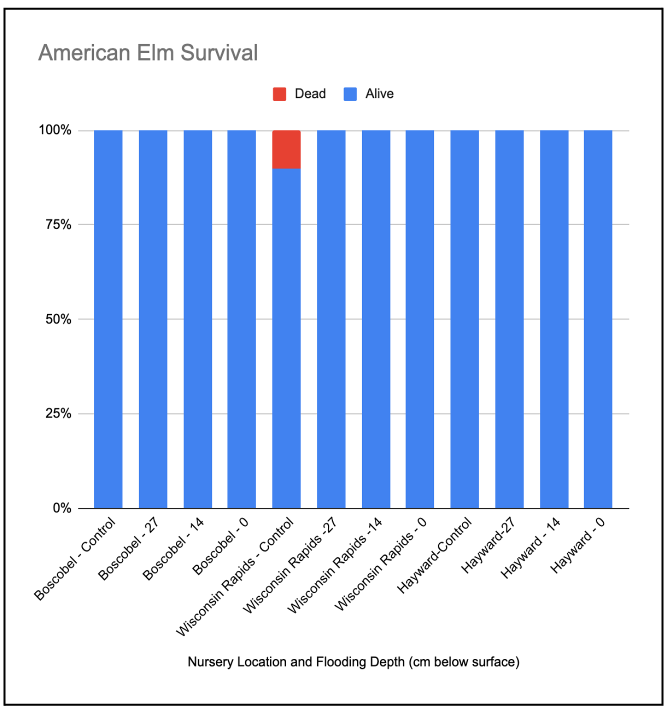

Figure 2.3 Percent survival of American elm at each nursery location and treatment

Table 2.2 Resulting P-values from fisher's exact test for count data for each species and nursery. Survival counts were calculated for each species, location, and treatment. Significant p-values (<0.05) signify whether or not there was a difference in survival between the control and flooding treatments for a particular species and nursery location.

Species	Nursery Location	P-Value
Acer rubrum L.	Hayward	0.6692
	Wisconsin Rapids	1
	Boscobel	1
Acer saccharinum L.	Hayward	0.6992
	Wisconsin Rapids	0.02451
	Boscobel	0.444
Betula nigra L.	Hayward	0.0004748
	Wisconsin Rapids	0.07054
	Boscobel	0.582
Betula alleghaniensis Britton	Hayward	0.2296
	Wisconsin Rapids	0.3719
	Boscobel	0.4425
Platanus occidentalis L.	Hayward	0.04657
	Wisconsin Rapids	0.01565
	Boscobel	0.5636
Quercus bicolor Willd.	Hayward	0.1429
	Wisconsin Rapids	0.4582
	Boscobel	1
Taxodium distichum (L.) Rich.	Hayward	1
	Wisconsin Rapids	0.2781
	Boscobel	1
Ulmus americana L.	Hayward	1
	Wisconsin Rapids	1
	Boscobel	1

Sycamore, river birch, yellow birch, and silver maple, had more mixed results for survival across locations and treatments with river birch having the poorest survival at Hayward. Bald cypress, the species with the southernmost range, had very low survival at the most northern location in Hayward, but had almost 100% survival in the most southern location of Boscobel with just less than 30% mortality in the control treatment. These results suggest that some species may be appropriate for planting across a larger latitudinal range, but others may only be appropriate for specific climate conditions.

Percent Growth of Treated and Control Seedlings

For some species, like sycamore and bald cypress, Boscobel was the only site where they had a positive percent height growth. When looking at percent height growth for some individual species, we can see there can be tradeoffs with survival. For example, American elm, which had the best survival across all species, did not necessarily have the best initial growth. On average, American elm seedlings that underwent flooding treatments experienced dieback at Hayward (Figure 2.4). Out of the three species previously identified as having high survival across treatments and sites (red maple, American elm, and swamp white oak), red maple had the highest average percent height growth.

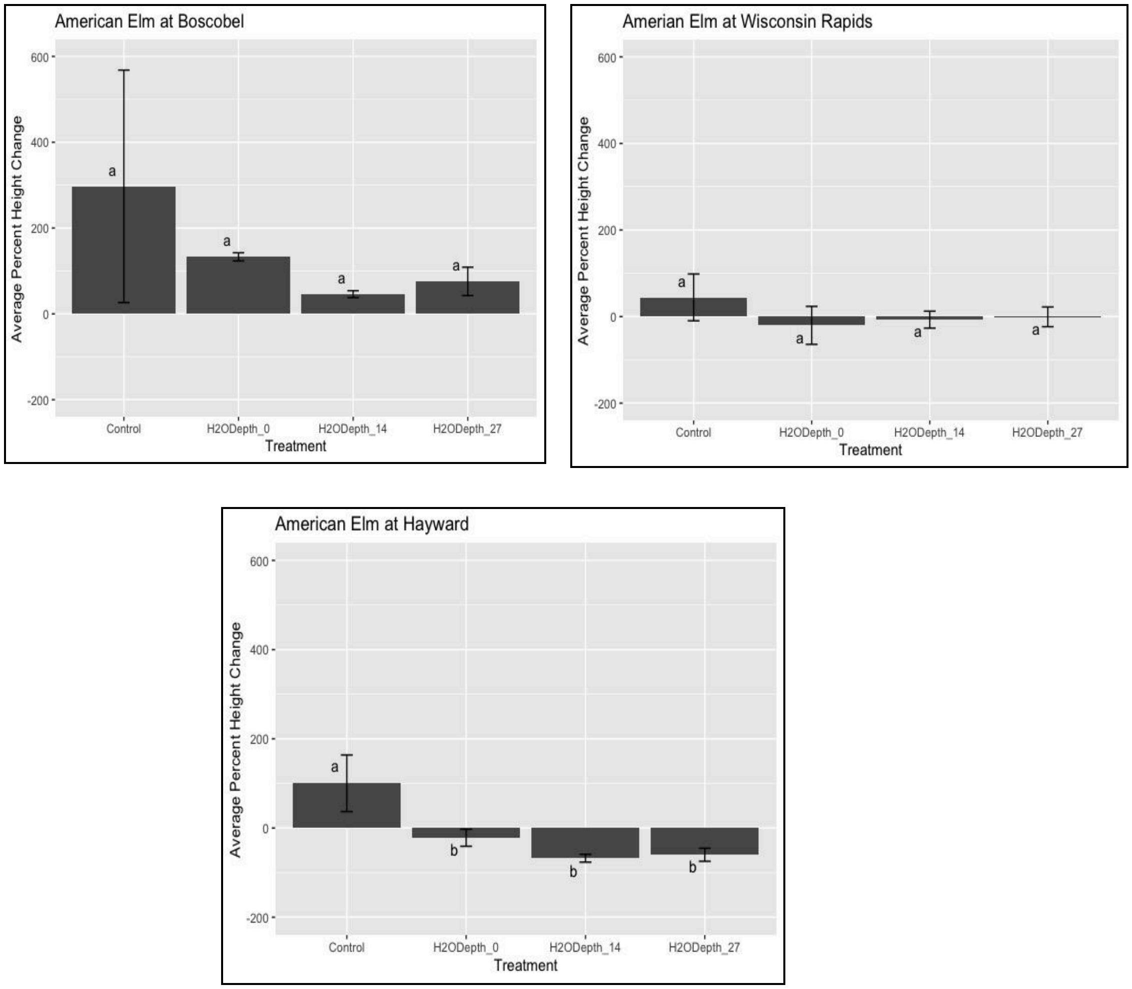

Figure 2.4 Average percent height growth with standard deviation error bars shown for each species and treatment at each nursery location (ordered from south to north). Alphabetical letter notations (a, b) signify significant differences between treatments as determined from least significant difference test with Bonferroni p-adjustment

Conclusion

Overall, there is no one single species that performed the best in survival and growth across the different locations and treatments. For many species, there was a tradeoff between survival and growth as those with the highest survival at any given site did not necessarily have the highest growth and vice versa. This information can be used to aid in the choice of appropriate species for a site in conjunction with information from previous studies like flood tolerance, shade tolerance, predicted climate changes, and in situ planting success. Additionally, foresters and nursery managers will need to work together on species selection, production, and outplanting logistics for the variable scenarios in which replacement species may be utilized. Care should be given to consider climatic regions (i.e winter severity and hydrology of the site) to pick an appropriate replacement species. The level or risk for EAB is also important to consider, for example if you are far from infestations, your stand may be better suited for species with high survival and lower growth. On the contrary, if your stand is immediately threatened you may plant a higher density of low survival, high growth species. Since this growth and survival was observed over a short period of time, there are plans to take additional measurements on these seedlings at year-five. Species from this study that show promise as replacements include American elm, river birch, and swamp white oak.

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Chapter 3: Where's Eastern Larch Beetle? Exploration of Different Detection Methods In Northern Wisconsin

Abstract

Forester and natural resource managers are increasingly exploring opportunities for early detection of emerging forest health concerns. One of these emerging concerns is eastern larch beetle (ELB) (*Dendroctonus simplex* LeConte), a native insect of tamarack (*Larix laricina*), which is causing increased tree mortality due to impacts from climate change. This increase in mortality creates a need to evaluate methods used to detect and quantify areas impacted. In northern Wisconsin, 50 stands or aerial detection polygons were surveyed in the field during the 2023 growing season to explore different detection tools for ELB. Twenty polygons identified by aerial sketch-map surveys as having ELB mortality and twenty tamarack stands identified as disturbed by a satellite imagery remote sensing algorithm (Astrape) were visited, along with ten randomly selected stands identified as tamarack by the Wisconsin forest inventory database (WisFIRs) for landscape level context. For each of the detection methods and the random stands, summary statistics on species composition, mortality, signs of ELB, invasive species, and water presence were calculated. Tamarack stands in northern Wisconsin were highly heterogeneous in species, which is likely contributing to the difficulties identifying both tamarack mortality and eastern larch mortality specifically caused by ELB across the two detection methods. ELB was common across the landscape but was not always associated with high levels of mortality. While overstory tree mortality was often identified in the aerial sketch map surveys and Astrape, this was not always tamarack mortality; some areas were even absent

of tamarack. Current methods of detection may be underperforming in this environment. With evolving changes in climate and dynamics between forests and insects, careful evaluation and innovation of detection methods are required to manage these environments effectively.

Introduction

Climate changes are leading to novel interactions between forests and insects. This poses a need to explore methods to detect and monitor insect disturbances, especially with an expected increase in disturbances with a changing climate (Jactel et al., 2019; Seidl et al., 2017). The genus *Dendroctonus* includes multiple native insects in North America shifting life history dynamics due to climate change. This shift is evident for one of the most well-known *Dendroctonus* species, mountain pine beetle (*Dendroctonus ponderosae*) (MPB). Mountain pine beetle has affected ecosystems not conventionally considered at risk due to range expansions (Amalesh Dhar et al., 2016; Cudmore et al., 2010; Howe et al., 2021; Mitton and Ferrenberg. 2012,). Similar range expansions have also been recorded for the southern pine beetle (*Dendroctonus frontalis* Zimmermann) (Dodds et al., 2018). Beyond range expansions, outbreaks of spruce beetle (*Dendroctonus rufipennis*) have also been linked to increased overwintering survival, faster maturation rates, and droughts (Berg et al., 2006).

Detection can be an important first step in identifying the location and impact of insect outbreaks, understanding insect-forest dynamics, and developing appropriate management strategies. Previous and current methods of detection have commonly included the use of aerial imagery, stereoscopes, field surveys, aerial sketch-map surveys, and increasingly, remote sensing technologies involving satellite, aerial, or drone imagery (Ciesla, 2000; Torres et al., 2021). Early detection is considered a critical time when management actions could focus on eradication and control, before the insect becomes established and long-term management strategies are needed (Waring et al., 2005).

An example of an emerging issue concerning mortality of tamarack caused by a native insect in Wisconsin and more broadly, the Great Lakes Region, is eastern larch beetle (ELB) (*Dendroctonus simplex* LeConte). Historically, ELB attacks were scattered and isolated; however recent studies show that this may be changing due to possible factors like climate change, and the most recent example is the epidemic level outbreak occurring in northern Minnesota (Hopkins 1909; Langor and Raske 1987a; 1989, Mckee et al., 2022; Mckee and Aukema, 2015a; Venette and Walter, 2012). This outbreak in particular is largely attributed to increased reproduction by the beetle due to warmer temperatures and longer growing seasons (Mckee et al., 2022; Venette and Walter, 2012). In the state of Wisconsin, increasing ELB damage was noticed initially in 1999 and has been mapped annually since 2012 (excluding the year 2020) (WI DNR, 2019).

The detection of species-specific forest pests, like ELB, relies on a knowledge of signals and characteristics of the individual pests and host species. For many species impacted by *Dendroctonus* there are visual cues within the host tree species that could be observed from afar (e.g. the red or grey stage in mountain pine beetle) (Ministry of Forests, 1995). Because these attacks have spatial correlations, aerial or remote surveys for obvious red attack stages can be used to estimate locations of adjacent *Dendroctonus* in the green attack stages and allow for targeted ground surveys (Wulder et al., 2009). However, detection for ELB in tamarack may be more difficult.]

For example, tamarack differs from other conifers in the region, because it is a deciduous conifer; the yearly loss of needles may impact the window for detection. This could make it more difficult to detect the signs of ELB which include premature yellowing and loss of needles (Mckee, 2015; Mckee et al., 2022; Seybold et al., 2002). Also, detection of infestations in the early stages can be tricky because not all infested trees will yellow early and trees often start yellowing from the bottom of the tree (Seybold et al., 2002). This could potentially hinder aerial detection or other remote sensing efforts when the tops of the trees still appear green. It may not be until the next growing season when the tree fails to flush out that reliable detections can be made (Seybold et al., 2002). Early on the ground identification may also be difficult because its other signs like galleries, frass accumulations, and holes in particular may be hard to see in the early stages or in light infestations and require close examination because beetles often enter the tree under bark flaps or crevices. (Mckee, 2015). These difficulties and challenges necessitate an evaluation of current detection methods to ensure that these infestations are being accurately detected and quantified.

Aerial sketch-mapping, the method Wisconsin has been using to track ELB across the landscape, typically involves hand drawing polygons of affected areas on a map from an airplane or helicopter. It is known as a low-cost method of developing coarse resolution landscape assessments and can have relatively high accuracy (70%) (Coleman et al., 2018; McConnell et al., 2000). The surveys can be general, looking for any and all disturbances on the landscape, or specific to a particular host or disturbance agent. Initial on the ground reviews conducted in summer of 2021 and 2022 of aerial sketch-map detected ELB mortality in Wisconsin presented variability in sites, with some stands appearing mostly healthy or only containing a small proportion of tamarack (Francart *in preparation*). This unexpected observation in the field has raised concerns about what these aerial sketch map surveys are clueing in on and presents a need to evaluate current aerial detection methods being used to locate and quantify ELB infestations in Wisconsin, as well as consider other methods.

One method worth considering is using satellite imagery and remote sensing change detection algorithms. Satellite sensors capture the reflected electromagnetic radiation from earth and have the ability to pick up on spectral changes in vegetation caused by disturbances. This makes them a potentially useful data source in forest health management. Satellite imagery provides consistent data input at regular intervals, making it useful for monitoring changes to the landscape, and it can be a low-cost method depending on the source used (Ciesla, 2000). Several algorithms using imagery from Landsat and Sentinel 2 have achieved promising accuracy (~70-98%) at detecting beetle attack (Fernandez-Carrillo et al., 2020; Meddens et al., 2013; Migas-Mazur et al., 2021; Senf et al., 2015). Many studies have looked at the detection of *Dendroctonous* beetles such as mountain pine beetle and spruce bark beetle, but to this date, no studies have specifically considered the detection of ELB damage using satellite imagery remote sensing (Abdullah et al., 2019; Fernandez-Carrillo et al., 2020; Meddens et al., 2013; Senf et al., 2015).

ELB, in this context, provides an opportunity for a case study to better understand the utility of different detection tools and methods for species specific disturbances. This study provides baseline data on the extent and intensity of ELB infestations across the landscape and helps determine the feasibility of using aerial sketch map surveys and a specific satellite imagery remote sensing algorithm as a tool to detect ELB mortality in northern Wisconsin. Further insight and improvement on these detection methods could help guide management efforts across the state and have broader implications for invasive species management and other forest health concerns.

Methods

Site Description and Study Extent

The study occurred in central and northern Wisconsin; an upland peneplain region, characterized by its many forests, kettle lakes, and wetlands (Martin and Twaites, 1916; Pohlman et al., 2006). The majority of the land area is forested, with aspen/birch, spruce/fir, red/white/jack pine, and maple/beech/birch types being the most abundant. In the late 19th century, most of this area was logged for the attempted development of agricultural land (referred to as the cutover period), significantly altering the landscape (Roth, 1898).

Generally, Wisconsin has a continental climate with warm summers and cold winters

(Wisconsin State Climatology Office). Rhinelander, Wisconsin (a central city within the study region) has an average low temperature in January of 0.4℃, and an average high in July of 25.6℃ (National Oceanic and Atmospheric Association [NOAA]). The study extent is a circle with a 40.2 kilometer radius with its centroid located at -89.64765386, 45.78185522 (Figure 3.1). Only public lands within the study extent were considered.

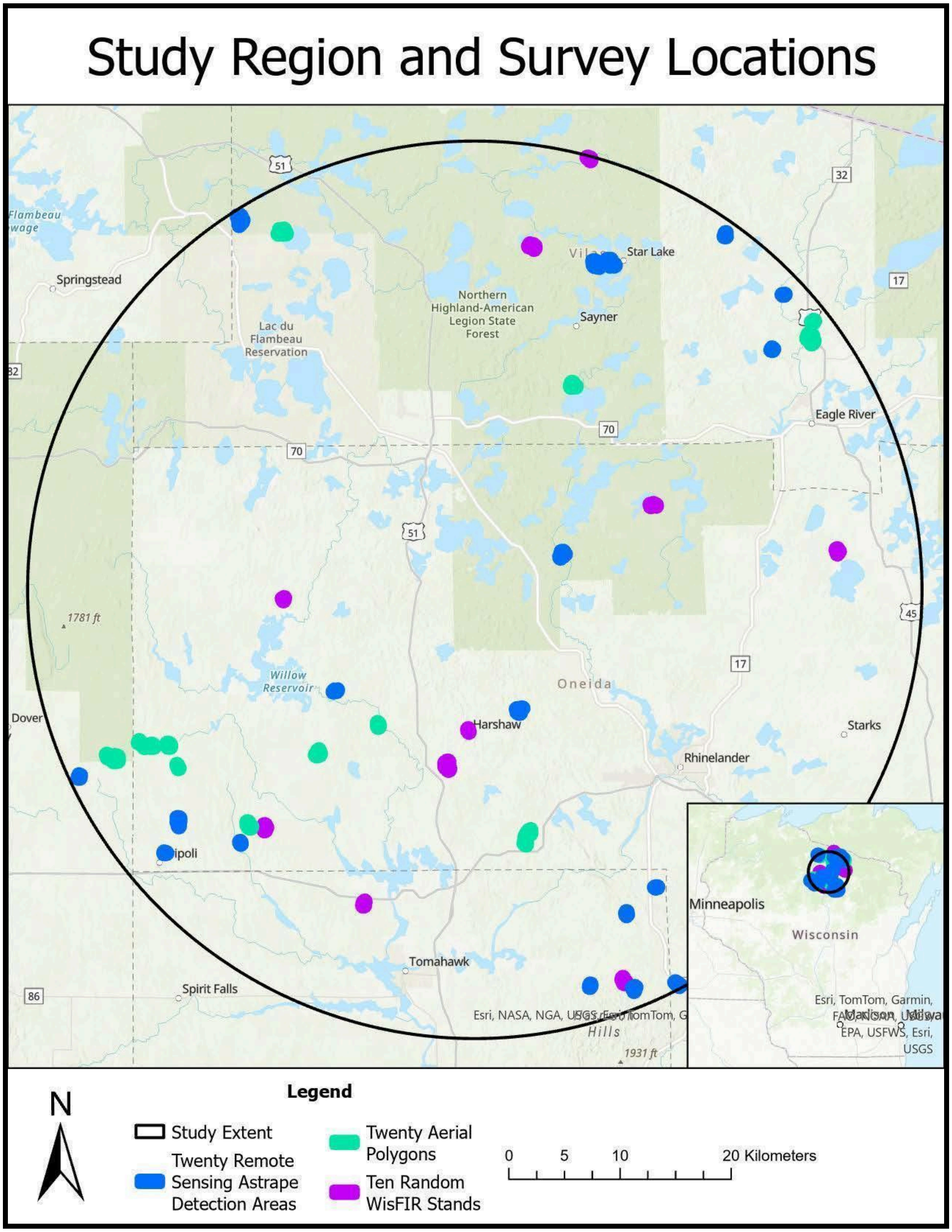

Figure 3.1 Map of study extent in north central Wisconsin that covers approximately 500,000 hectares.

Tamarack Silvics

Tamarack can be characterized as a shade intolerant, potentially fast growing, pioneer conifer species that is often the first tree species to establish on bog sites (Brown, 1982; Johnston, 1980; Johnston, 1990). It can be found growing on a wide variety of sites and climatic conditions across its range, which extends from eastern Canada and northeastern United States to Alaska (Johnston, 1990). In the southern extent of its range in the Great Lakes region, tamarack is commonly found on sites that are poorly drained, like forested swamps and bogs, where it can compete well due to its tolerance to cold, wet, and acidic soils (Heinselman, 1970; Johnston, 1990; Jeffers, 1975; Johnston, 1980; Johnston, 1990; Duncan, 1954). In Wisconsin, tamarack does not tend to grow in extensive homogenous stands and more commonly is found in small pockets, which differs from tamarack in north central Minnesota (Curtis, 1959; Johnston, 1990). In mixed stands, black spruce is one of the most common associates, but other species like quaking aspen (*Populus tremuloides*), balsam fir (*Abies balsamea* L.), northern white cedar (*Thuja occidentalis* L.), red maple (*Acer rubrum* L.), red pine (*Pinus resinosa* Aiton), white pine (*Pinus strobus* L.), black ash (*Fraxinus nigra* Marsh.), and white spruce (*Picea glauca* (Moench) Voss) can be found as associates depending on the site and region (Curtis, 1959; Johnston, 1990; Johnston, 1980).

Eastern Larch Beetle

ELB is a native bark beetle that closely follows the range of its host, tamarack, and historically attacks dead and dying tamarack (Hopkins, 1909; Seybold et al., 2002). In the spring, beetles synchronously emerge from their overwintering trees and use a mass attack strategy to overcome tree defenses such as resin production and proceed to reproduce and feed on the phloem of new host trees. They effectively cut off the transportation of sugars, proteins, and other molecules that the tree needs by tunneling underneath the bark (Hopkins, 1909; Seybold et al., 2002). The number of broods they produce each year largely depends on the climate and growing degree days and at least part of the population is capable of bivoltinism or semi voltinism (Hopkins, 1909; Langor and Raske, 1987a; Mckee and Aukema, 2015b; Seybold et al., 2002).

Site Selection

WisFIRS

Random stands of tamarack were selected using the Wisconsin Forest/Field Inventory and Reporting System Public Lands application (WisFIRS). This is a database for forest and habitat managers to track and store data collection, treatments, management actions etc. In order to better understand the base level condition of tamarack and dynamics with ELB across the landscape, ten WisFIRS stands categorized as tamarack stands were randomly selected from the database to be surveyed.

Aerial Sketch-Map Surveys

The dataset used for this research includes polygons detected as having ELB infestations via aerial surveying from 2012-2021 (excluding 2020). Surveys do not typically follow a predetermined pattern from year to year. Where flights are conducted depends on previous surveys and forester input. Occasionally more systematic surveys are used. Typically, a small crew will fly in a Cessna 172, 180, or 182 between 1,000 and 2,000 feet above ground level. The crew may have GPS points of areas they want to look at either from prior surveys, ground observations, or from foresters' reports. The crew utilizes a tablet with background aerial imagery on the Forest Service application Digital Aerial Sketch Mapper that tracks the plane as it flies. From the plane, crew members draw polygons around the areas on the ground that appear to have dead or dying trees. Ground survey protocols for accuracy vary for ELB and are meant to be adaptive for individually determined needs (Mike Hillstrom, *personal communication*).

Twenty aerial polygons drawn from these sketch-map surveys were randomly selected. Aerial polygons from the surveys were clipped to a public lands layer. Clipping this layer often created fragmented pieces from original polygons. Each fragmented polygon was treated as a potential site to be randomly selected for validation.

Satellite Imagery Remote Sensing Algorithm (Astrape)

The change detection algorithm based on satellite imagery used for detection of healthy and unhealthy tamarack stands is an algorithm called Astrape (Wegmueller and Townsend, 2021). The algorithm uses Sentinel-2 and Planet Dove imagery from July 2018 and July 2021 to segment and classify disturbance using Jenks Natural Breaks and Extreme Gradient Boosting (Wegmueller and Townsend, 2021). The output was a raster dataset with 10x10m cells numbered from 1-9 describing change detection severity, with 9 being the most severe.

In the field, pilot surveys from the summer of 2022 indicated that stands overlapping with the 9 category contained areas of tamarack infested with or killed by ELB. Categories 8-9 were chosen as a suitable severity level, and the raster dataset with those values was converted to a polygon layer and intersected with WisFIRS tamarack stands. A stand was considered unhealthy if it had more than 30% overlap with the highest category of change detection (8-9). Twenty stands detected as unhealthy from this remote sensing algorithm were randomly selected to be surveyed.

In total, 50 randomly selected stands/polygons (**Appendix 3A**) were selected, visited, surveyed, and analyzed to evaluate detection methods: 20 aerial sketch-mapped polygons, 20 Astrape, satellite imagery remotely sensed WisFIRS tamarack stands detected as unhealthy, and 10 randomly selected WisFIRS tamaracks stands. Only polygons and stands between 5-60 acres, within 0.5 miles of a road, and on public property were eligible for selection due to accessibility needs and feasibility for surveying. A preliminary desktop review of selected sites was conducted to confirm accessibility, and sites deemed unsuitable either due to water body crossings or needing to cross private land were removed and alternative random sites were used.

Stand Evaluation Procedure

To evaluate a stand or polygon, we used a systematic approach by first creating a 45-m^2 grid over the area. We then generated centroids for each grid cell to be used as a plot center. We scaled the number of plots sampled per site based on size of the stand or polygon (Table 3.1).

Plots were sampled in a snaking pattern starting from a suitable “corner” of the stand or polygon. A minimum of five plots were sampled per site.

Table 3.1 Plot selection guide

Size of Stand (hectares)	Plot Selection
2.0 - < 4.0	Every other cell
4.0 - < 12.1	Every 5th cell
12.1 - 24.3	Every 8th cell

Survey Protocols

At each stand or polygon a brief visual assessment of conditions at each of the predetermined sample plots was recorded by answering a series of multiple choice questions (**Appendix 3B**) about overstory species composition, overstory tree mortality, signs of ELB, hydrology, and invasive species. Plot extents were determined by the visual range of the observer from the plot center. The estimate of this average distance was approximately five to ten meters depending on site conditions like tree and understory density.

Analysis

In total, 544 plots were surveyed in the field, seven plots (0.01% of data) were excluded from analysis due to missing information. Data from aerial survey polygons and random WisFIR tamarack stands were summarized at the stand level. Data from aerial surveys were summarized at the stand/polygon level because the whole area represents the detection of a disturbance. Data from the remote sensing surveys were summarized at the plot level to evaluate what fell in the actual area of detection in a tamarack stand.

Data was summarized by flow chart based on whether or not tamarack was a major component of the stand or plot ($\geq$ 50% of plots had tamarack as a dominant species or tamarack was dominant in the plot), if there was dead tamarack in the majority of the stand or plot ($\geq$ 50% of plots or presence of dead tamarack in the plot), and if signs of ELB were present. Additional tables were created to give a breakdown of plots at each site for each survey type, providing what percent of plots at each site fell into categories related to tamarack presence, dominance, mortality, ELB signs, percent mortality, and severity of ELB (**Appendix 3C**).

To elucidate species composition in plots not dominated by tamarack, species were assigned a weighted value, depending on how many dominant species were identified. Plots with three dominant species would designate that each species be assigned a value of 0.33, species in plots with two dominant species were assigned a value of 0.50, and species in plots with only one dominant species were assigned a value of 1.

Results

WisFIRS Tamarack Stands

The randomly selected ten WisFIRS stands categorized as tamarack all had tamarack present in ≥ 50% of plots (**Appendix 3C**). However, tamarack was only *dominant* at ≥ 50% of plots in half of the stands (Figure 3.2). Many of the randomly selected WisFIRS stands included a component of black spruce, red maple, or balsam fir as an associate, with lesser occurrences of species like red pine, paper birch, yellow birch, oak, northern white cedar and aspen. Summaries of plots by stand show that eight out of 10 stands had signs of ELB present in at least one plot (**Appendix 3C**). Only three of these stands with signs of ELB had any plots with ≥ 75% of the overstory tamarack being dead. In general, in all of the plots with signs of ELB, 25% percent had tamarack mortality ≥ 50%.

Out of the 10 stands observed, only four of them were identified as having tamarack dominant in ≥50% the plots, with dead tamarack in ≥50% of the plots, and with signs of ELB (Figure 3.2). The one remaining stand dominated by tamarack had dead trees in the majority of the plots, but they were mostly other dead species, and not dead tamarack.

In the five stands where tamarack was not dominant in the majority of plots, black spruce was the most abundant dominant species in plots not dominated by tamarack according to weighted value in three of the stands. The remaining two stands had more mixed species compositions in non-tamarack dominated plots with red maple, northern white cedar, black ash, black spruce, red oak, balsam fir, and aspen species present. In the five non-tamarack dominant stands, only two had any dead tree species in at least half the plots. Dead tamarack and signs of ELB were found in two and three of these stands, respectively.

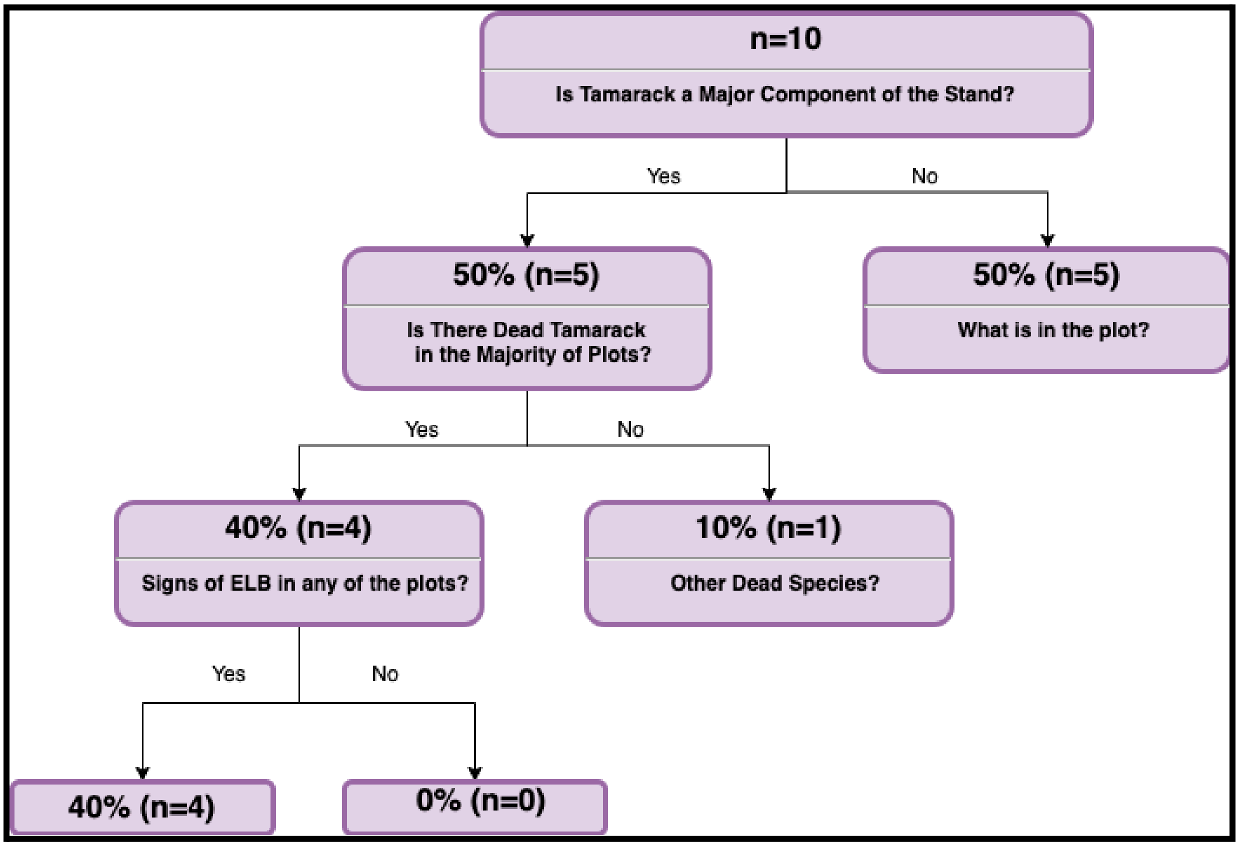

Figure 3.2 Demonstrates the proportion and percent (4/10 and 40%) of ten random WisFIRS tamarack stands surveyed that can be classified as heavily tamarack dominant (≥ to 50% of plots with tamarack dominant overstory) stands with high mortality (≥ 50% of plots with dead tamarack in overstory) and ELB present.

Aerial Sketch Map Polygons

Of the 20 aerial sketch map polygons, the majority had tamarack present in ≥ 50% of plots (**Appendix 3C**). However, tamarack was only *dominant* at ≥ 50% of plots in nine of the stands (Figure 3.3). Many of the aerial stands included a component of black spruce with lesser occurrences of species like red maple, balsam fir, aspen species, paper birch, oak, northern white cedar, red pine, and white pine. Summaries of plots by stand show that 12 (60%) out of 20 stands had signs of ELB present in at least one plot (**Appendix 3C**).

Out of the 20 stands observed, only four of them were identified as having tamarack dominant in ≥ 50% of the plots, with dead tamarack in ≥ 50% the plots, and with signs of ELB (Figure 3.3). In the other five tamarack dominated stands, four of them had tamarack mortality occurring in 30 - < 50% of plots.

Eleven polygons had tamarack as a *dominant* species (≥ 50% of the overstory trees) in < 50% of the plots sampled (Figure 3.3). Of these 11 polygons, > 50% had some tamarack but it was not the dominant species; these plots were dominated by other conifer species like black spruce and balsam fir. While tamarack was not generally the dominant overstory species within these 11 polygons, eight of them did have either a dead tamarack or other dead tree species in the majority of the plots.

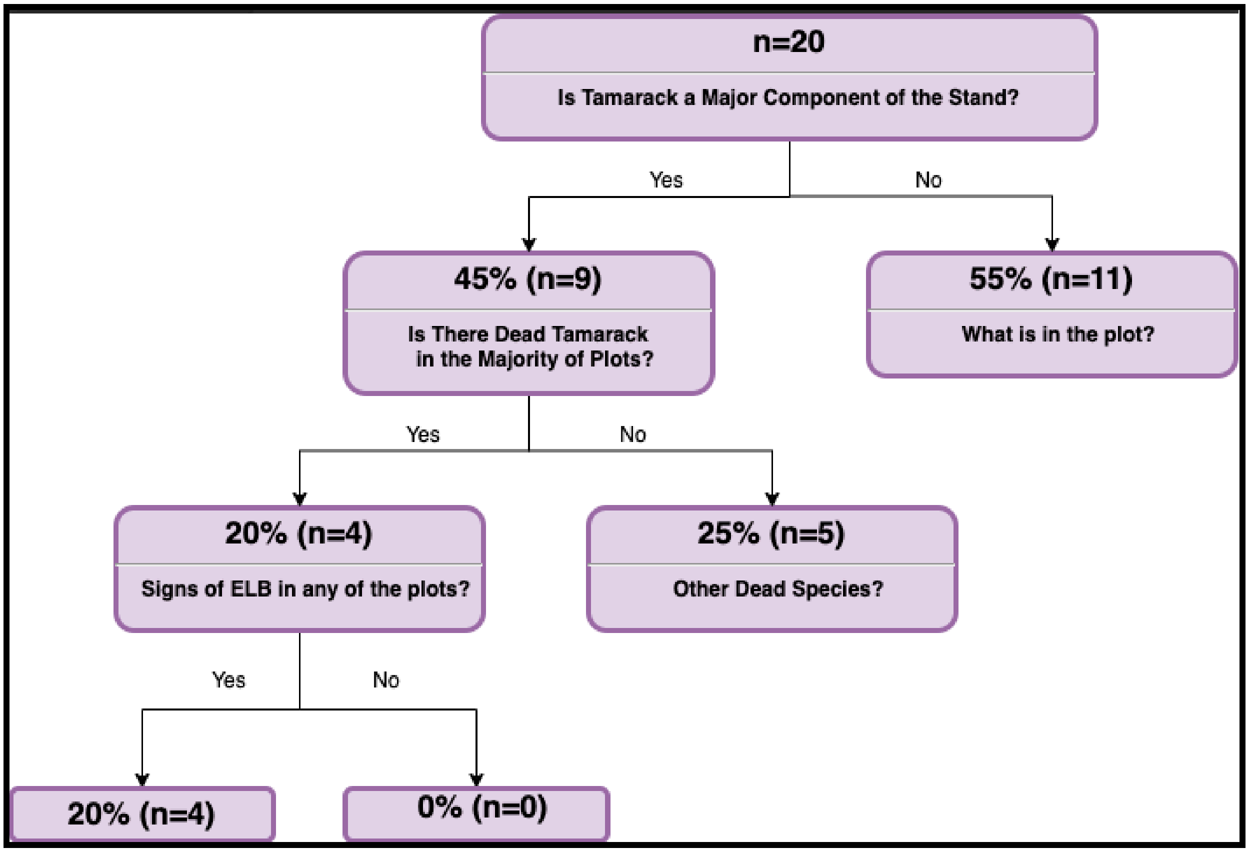

Figure 3.3 Demonstrates the proportion and percent (4/20 and 20%) of aerial sketch map polygons surveyed that can be classified as heavily tamarack dominant (≥ to 50% of plots with tamarack dominant overstory) stands with high mortality (≥ 50% of plots with dead tamarack in overstory) and eastern larch beetle present.

Remotely Sensed Stands

Data was summarized by plots based on their overlap with detected disturbance. Of the 148 plots, 41 plots contained dead tamarack and 36 plots contained signs of ELB. Eighty-eight plots were located in areas of detection by Astrape. Areas of detection overlapped with 68% of the plots with dead tamarack and 76% of the plots with signs of ELB.

Within the 88 detection plots, tamarack was present in 81% (n = 71) and dominant in 53% (n = 47) of these plots, respectively (Figure 4 and Appendix C). In the tamarack dominant plots, 21 plots had dead tamarack in them with 18 plots containing signs of ELB. In the remaining 26 plots, 12 had other dead species. Out of the 88 plots observed, only 18 (20 %) of them were identified as being tamarack dominant, having dead tamarack, and with signs of ELB (Figure 3.4).

In plots where tamarack was not dominant in at least half the overstory, black spruce was the most abundant dominant species according to weighted value (42%), followed by eastern white pine (19.1%), and northern white cedar (7.3%). Hardwoods and other conifer species were occasionally dominant (21.8%) and the remainder had no overstory (2.4%) or was not recorded (2.4%). In the 41 other plots not dominated by tamarack, 33 of them had either a dead tamarack or other dead species in them (Figure 3.4).

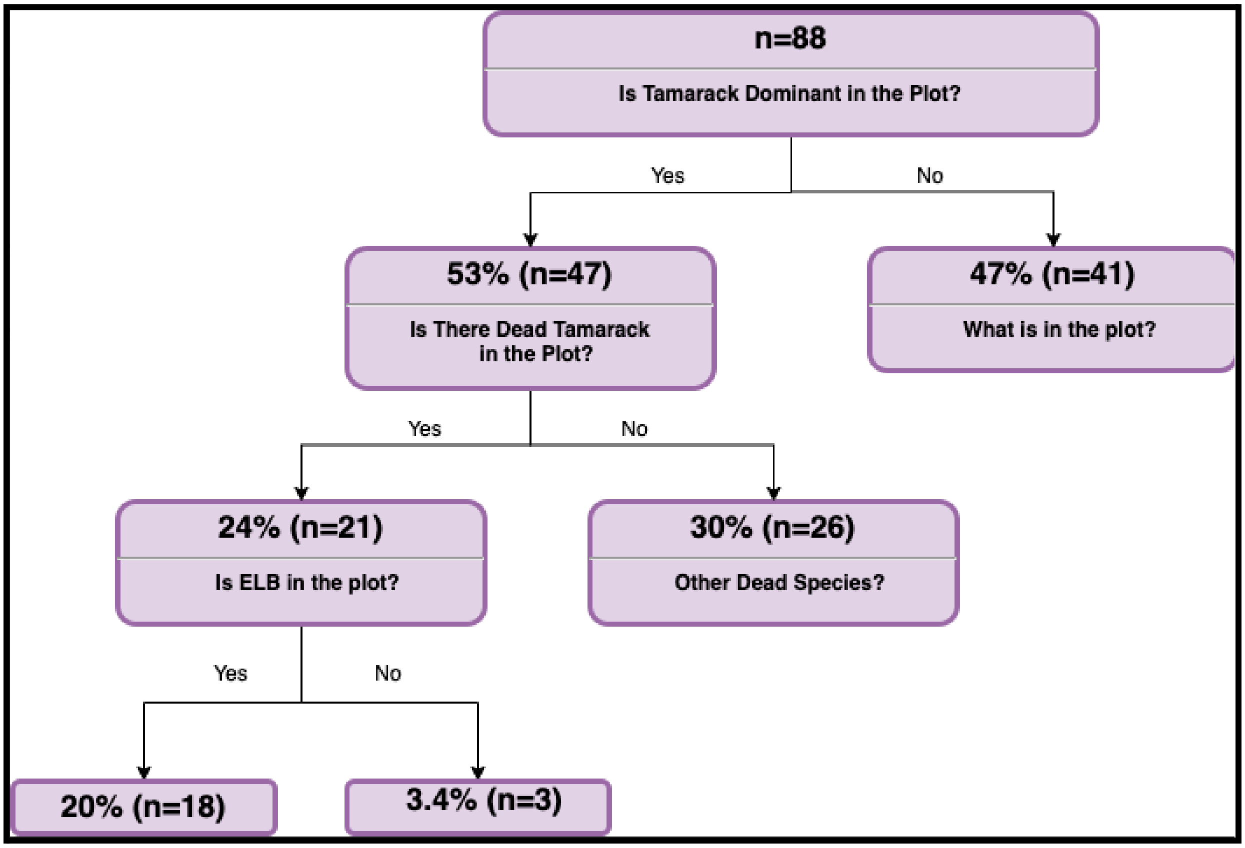

Figure 3.4. Demonstrates the proportion and percent (18/88 and 20%) of surveyed plots detected as disturbed by a remote sensing algorithm, Astrape, that can be classified tamarack dominant (≥ 50% of overstory with tamarack dominant), with tamarack mortality (at least one dead overstory tamarack), and eastern larch beetle signs present.

The remaining 60 plots were located in areas not detected as disturbance by Astrape. Tamarack was present in 71.67% (n=43) of these plots and was dominant in 51.67% (n=31) (Figure 3.5 and Appendix 3C). Of the 31 plots dominated by tamarack, 12 (39%) had dead tamarack in them with six of these plots containing signs of ELB. In the remaining 19 plots, 5.26% (n=1) had other dead species.Out of the 60 plots observed, only six plots (10%) were identified as being tamarack dominant, having dead tamarack, and with signs of ELB (Figure 3.5).

In the 29 plots not dominated by tamarack in half the overstory, black spruce was the most abundant dominant species (29.5%), followed by northern white cedar (19.3%) and red maple (10.3%) according to the weighted value of species. Other hardwood and conifer species were occasionally dominant (25.6%), and the remainder had no overstory (11.6%) or was not recorded (3.9%). Only one plot not dominated by tamarack had a dead tamarack and 13 of the non-tamarack dominated plots had other dead species in them (Figure 3.5).

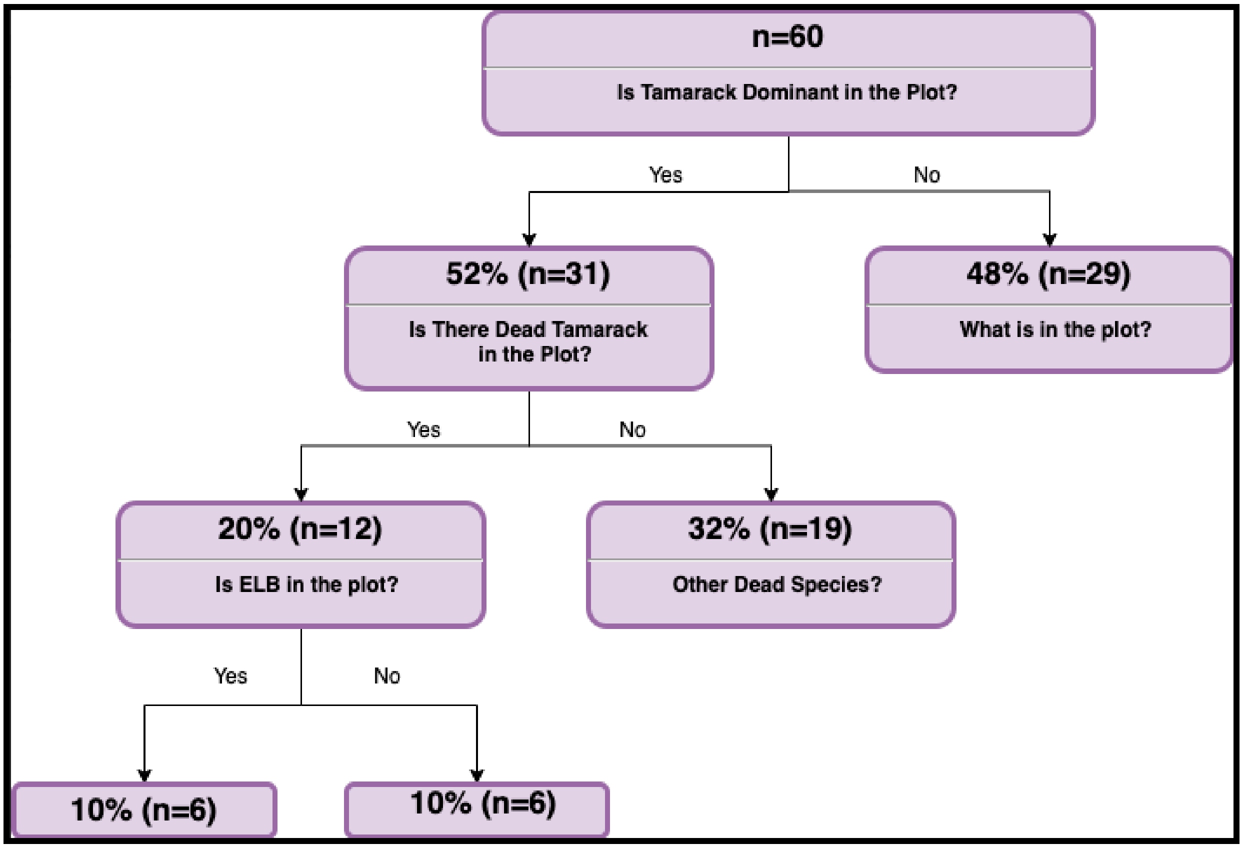

Figure 3.5. Demonstrates the proportion and percent (6/60 and 10%) of surveyed plots not detected as disturbed by a remote sensing algorithm, Astrape, that can be classified tamarack dominant (≥ 50% of overstory with tamarack dominant), with tamarack mortality (at least one dead overstory tamarack), and eastern larch beetle signs present.

Discussion

When developing a detection model for a forest insect, it is critical to consider the processes and interactions among the life history of the host, the pest, and the surrounding ecosystem, as well as the tools and resources available to detect these processes. Our results highlight the difficulties in detecting ELB in tamarack stands among multiple detection methods in northern Wisconsin. The tamarack stands were variable in nature, sometimes being pure or mixed stands, intermixed within upland forest types, and having ranging levels of mortality from ELB. Additional variables including levels of standing water, water bodies, and invasive species that can aid in detection in other systems (e.g. increased water levels post emerald ash borer invasion), were detected with low frequency (Ahmed et al., 2011; Robertson et al., 2018; Slesak et al., 2014; Townsend, 2002) (**Appendix 3C**). Both aerial sketch map surveys and the remote sensing algorithm Astrape have the potential for identifying ELB mortality, but with significant limitations.

While tamarack is its own forest type within northern Wisconsin, only 5 of the 10 WisFIRS stands were typed as tamarack dominant in over half the plots. This discrepancy could be due to differences in interpretation; WisFIRS's classification is based on inventory data (e.g. fixed radius plots with calculated TPH and basal area) and our study used visual interpretation of the overstory. Despite this difference between classification, it is evident that areas of tamarack stands visited often contained or were dominated by species other than tamarack.

Mixed species are common for this forest type (Roth, 1898), and this characteristic presents several challenges for detection. In tamarack stands with more mixed composition, it is possible to mistake other tree species for tamarack when being viewed from afar in aerial surveys or satellite imagery. Accurate identification of tamarack may be especially difficult if the other species present is black spruce, one of its most common associates in the region, which has a similar form to tamarack (Curtis, 1959; Johnston, 1990; Johnston, 1980). Differentiating between dead tamarack and black spruce would be even more difficult due to the lack of leaves (McConnell, 2000). The difficulty of identifying disturbances in mixed stands has been recognized in a previous detection study that found a higher misattribution of disturbance agents in mixed stands when compared to pure stands when using remote sensing (Senf et al., 2015).

This mixed species composition was observed in both aerial sketch map survey polygons and the areas detected as disturbed by Astrape. In both instances, approximately half of the stands and/or plots were dominated by other species, often black spruce, reaffirming the suspected challenges associated with accurately identifying species using aerial or remote techniques.

It is important to note that Astrape is not meant to be species specific and must be used in conjunction with ancillary data of known areas of species to be functional in that capacity. Given this, the results of Astrape were expected to reflect the species composition of the input WisFIR data. Currently, remote sensing techniques using satellite imagery struggle to identify trees to species on a practical level (Fassnacht et al., 2024). This suggests that this method alone, cannot be used to accurately identify and detect species specific disturbances like ELB without having additional data with target species location.

In contrast, aerial sketch map surveys for ELB are meant to be species specific, but previous assessments have shown that with aerial sketch map surveys in general, accuracy in detection was found to decline as classification increases in specificity from genera to species for both host and pest species, suggesting that aerial sketch map surveys may need additional data or background knowledge to accurately classify disturbances limited to a specific species (Coleman et al., 2018). In aerial sketch map surveys, these trees are being viewed from 1000-2000 feet, meaning there are fewer characteristics available for identification than on the ground surveys, and because tamarack here can often be found in small pockets or confined to the center or edges of wet areas, it may be hard to visually delineate those areas from other adjacent forest types, leading to inaccurate disturbance boundaries (Curtis, 1959; Johnston, 1990; Roth, 1898). While most areas of detection not dominated by tamarack were dominated by lowland conifer types like black spruce, there were a few polygons surveyed that contained unexpected species mixtures. There were two polygons where there were no tamarack and the polygons were actually a hardwood species mixture. Additionally, there was one aerial polygon where just under half the polygon was tamarack and the other half was an alder shrub swamp with no overstory trees.

These two examples highlight the potential, when quickly sketching a polygon in the air, to inflate the area impacted by a disturbance agent. Aerial sketch map surveys are prone to human error and judgment because it is difficult to capture details moving at such a high speed (Ciesla, 2000; McConnell et al., 2000). The polygons with hardwood mixtures and alder shrub swamp were not the full original polygons drawn from the aerial surveys, they were just portions that fell on public land. These polygon areas, however, were 46.8, 20.9 and 13.6 acres in size. This demonstrates that individual polygon areas of ELB disturbance could be overestimated using this method which has been found previously to occur with aerial sketch map surveys (Bright et al., 2020; Egan et al., 2019; Harris and Dawson, 1979). These factors, combined with the knowledge that many of the polygons were more heavily dominated by black spruce, suggests these surveys may not accurately be capturing the true level of disturbance caused by ELB due to the mixed nature of species composition and high heterogeneity in and between stands in this region.

The range in mortality levels occurring in these tamarack stands further complicates the detection of ELB. While some of our WisFIRS stands with ELB had plots with tamarack mortality over 75%, the majority fell in the under 50% mortality categories. These areas of lower mortality may be difficult to identify because dead trees or pockets of trees may be surrounded by live ones, making them more difficult to see from an aerial survey or to distinguish in a satellite imagery pixel containing a mixture of reflectance values (Hall et al., 2016). This was found to be true for aerial sketch map surveys conducted in Minnesota, where areas determined to have no ELB from the air, were revealed to have low level infestations on the ground (Shaunette, 2022). Remote sensing studies using satellite imagery have also reported more challenges detecting low level or early attacks (Coops et al., 2006; Senf et al., 2015; Wulder et al., 2006).

At a baseline level, 40% of the WisFIR tamarack stands were identified as being dominated by tamarack, with tamarack mortality in the majority of the stand, and containing ELB. This value was 20% for both the aerial sketch map survey polygons and areas of detection by Astrape. While these values were not statistically compared, this value is lower than was expected given the goals and application of each detection method. With aerial sketch map surveys and Astrape, given they are purposefully looking for disturbance and not derived from a random sample of stands, we expected to see a higher detection rate. These values suggest that both methods may be missing areas that have high tamarack dominance, wide tamarack mortality, and ELB. For aerial surveys, this could partially stem from survey methods that do not systematically cover the same areas year to year (Hillstrom, *personal communication*). But even systematic surveys, like the ones conducted in Minnesota, can miss low level infestations (Shaunette, 2022).

An important factor to keep in mind in the discussion of the results for the remotely sensed stands and plots is that the algorithm used in this study, Astrape, was not originally designed for the purpose we are using it for. Astrape was originally designed for detecting larger scale abiotic disturbances with known occurrence times and has proven to be accurate when used in those circumstances (Wegmuller and Townsend, 2021). ELB is not an abiotic disturbance, and its impact does not have a defined start and end time, which makes using Astrape for its detection difficult because it uses imagery from two set dates. This means that disturbance that occurs outside our set times could be missed in our search.

Another consideration is the algorithm's use of Sentinel-2 data, which has been found in a previous study to have high accuracy in areas with large continuous areas of bark beetle infestation, but not so well in small infestation areas (Zimmerman et al., 2020). As shown in our baseline data, many of our stands may have small areas of infestation due to the mixed nature of stands, and many of our stands were small in size with over half being under 10 acres in area

(Appendix 3A). Some further discretion is needed in considering the Astrape results with the context of the baseline WisFIRS data because the WisFIRS's value is based on stand level summarization, and the remote sensing value is based on plot level summarization.

Another caveat to the use of this algorithm in tamarack stands is that the algorithm may actually be picking up on areas with changes in water content and identifying that as disturbance (Wegmuller, *personal communications*). Tamarack stands are often in wetter environments, and this confusion may be more prominent in stands with a lower density of trees where more of the ground is exposed and thus being reflected back to the satellite (Roth, 1898; Wegmuller, *personal communications*).

While these methods may be missing some of the areas of high tamarack dominance, wide tamarack mortality, and ELB, there does appear to be mortality occurring in these areas of detection that both methods noted. In the aerial polygons, the majority of stands not dominated by tamarack had tree mortality occurring in most of the stand area. And in the stands dominated by tamarack but without wide tamarack mortality, at least 25% of the plots in each stand had some tree mortality occurring. In the Astrape detection areas, 63% of plots not dominated by tamarack had other dead species in them and in plots dominated by tamarack but with no dead tamarack, 46.2% had other dead species. As we did not statistically compare these groups of data it cannot be said if this mortality level is significantly higher from baseline WisFIRS data where only two out of five stands not dominated by tamarack had tree mortality in over half the plots and the one stand that was dominated by tamarack, but did not have wide tamarack mortality, had overall tree mortality in over half the stand.

For both methods, the mixed species composition and heterogeneity of these stands likely resulted in the inclusion of species other than tamarack in these detection areas, which could be leading to overestimation of ELB disturbance area, particularly at the polygon level. Simultaneously, the number of detection areas representing tamarack dominant stands with widespread tamarack mortality and signs of ELB was less than expected given the context of the WisFIRS data, hinting at possible underestimation of the total areas of disturbance across a landscape. Many of these detection areas did have some level of mortality occurring in them, meaning they may be useful tools to aid in the identification of potential areas of ELB disturbance, but cannot be relied on alone as the sole measure of ELB disturbance across a region.

Conclusion

Climate change is going to continue to have an impact on forest disturbances, and often may result in new management issues and concerns, as evidenced by the unprecedented ELB morality levels across northern Minnesota. Given the remote nature of these forest ecosystems and the deciduous nature of tamarack, conventional on-the-ground surveys when sites are most accessible (frozen ground) are difficult and limited in their ability to detect dead and dying trees quickly. Hence, aerial or remotely sensed tools would be ideal to aid in detection.

This study highlights the challenges and opportunities in using two detection methods, aerial surveys and a remote sensing algorithm, for detecting insect specific disturbances. In cases where forests are of a mixed species composition, these methods may struggle to distinguish between tree and insect species, but they show promise as tools to identify areas of potential mortality. Once areas of potential morality are identified, then further investigations could be implemented to confirm the disturbance agent. It may be useful to combine methods of detection to either confirm locations or investigate locations further where there are method disagreements. For example, aerial photography or stand maps could be overlaid with aerial sketch map polygons to identify potential areas of confusion and then visited in the field to verify. Even with ever evolving technology in remote sensing techniques, it will likely still be necessary to verify these detected disturbances through on the ground surveys. This can also lead to an iterative process in model refinement. For best results, managers should consider multiple methods of detection and during forest inventory and be vigilant during other field visits since early detection can allow for control of a forest health threat.

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Chapter 4: Tamarack and eastern larch beetle dynamics; Exploring infestation, structure, and composition of tamarack in Northern Wisconsin

Abstract

The eastern larch beetle (ELB) (Dendroctonus simplex LeConte), which is native to North America, has historically been considered unaggressive. However, recent developments in the beetle's life history with climate changes and outbreaks in tamarack (Larix laricina (Du Roi) K. Koch) forests have provoked the reconsideration of this classification. The beetle has caused considerable damage to tamarack in the southern extent of its range in northern Minnesota. However, limited work has occurred in the neighboring state of Wisconsin to quantify impact and ecosystem response from ELB in tamarack forests. Across north central Wisconsin 29 areas were inventoried using fixed radius plots to quantify overstory trees, the regeneration layer, ground cover, ELB presence, invasive species, and canopy coverage. Data were analyzed using ANOVA, t-tests, and linear regression. Plots with ELB had a higher proportion of host species ($p=0.006$), higher total large tree density and large tamarack tree density ($p=0.033$, $p=0.004$ respectively), and a lower amount of small tamarack trees ($p=0.005$) compared to plots without ELB. Large tree density was also found to be negatively correlated to live tamarack regeneration ($p = 3.459e-05$, $R2=0.079$). This information can be used to guide management efforts across the Great Lakes region

Introduction

Insects are an important disturbance agent in forest ecosystems, shifting and influencing stand dynamics and subsequent structure and composition (Byler and Hagle, 2000; Schowalter, 2000). According to the FAO Global Resource Assessment, insect pests and disease are the leading cause of disturbance in North America, with approximately 85 million hectares of forests impacted by insect pests between 2003 and 2012 globally (van Lierop, et al. 2015). Insects are also a fundamental component of the functioning of a forest ecosystem, serving as pollinators, acting as a food source to other wildlife, predating other plants and wildlife, and assisting in other ecosystems functions (Greenwood, 1987; Mattson & Addy, 1975).

Beetles within the *Dendroctonus* genus have been previously described as a natural thinning agent during endemic population levels, focusing their attacks on suppressed, dying, or recently dead trees (Oliver, 1995). However, when conditions align this genus can cause extensive tree mortality, which has been especially common during the first 25 years of the 21st century in North American forests (Bentz, 2008). *Dendroctonus* beetles burrow through the bark, reproduce, and feed on the phloem of conifer species like *Pinus*, *Picea*, or *Larix*. Under endemic levels, healthy trees can use resin to defend themselves. However, under epidemic conditions, beetle populations can be so high that mass attacks through the use of pheromones can overcome tree defenses, resulting in mortality of healthy trees (Bentz, 2008).

Climate change is impacting the dynamics between insects and forests through direct and indirect pathways such as changes in voltinism, survival, predation by parasitoids, host availability, disturbance frequency and intensity, and range shifts (Jactel et al., 2019; Pureswaran et al. 2018). This impact has become evident in the *Dendroctonus* genus in recent decades. For example, mountain pine beetle (*Dendroctonus ponderosae* Hopkins) and southern pine beetle (*Dendroctonus frontalis* Zimmermann) have experienced range expansions, and spruce beetle (*Dendroctonus rufipennis* Kirby) outbreaks in the 1990s were attributed to faster maturation rates, increased overwinter survival, and drought (Carroll et al., 2004; Berg et al. 2006; Dhar et al., 2016; Dodds et al., 2018; Howe et al., 2021). These outbreaks are resulting in changes in forest composition and structure (Audley et al., 2020).

Similar changes are being observed in another *Dendroctonus* species which was previously considered benign. With warmer temperatures and longer growing seasons, eastern larch beetle (*Dendroctonus simplex*[ELB]), has had an increased rate of reproduction (Mckee, et al., 2022; Venette and Walter, 2012). ELB is a native, historically unaggressive, bark beetle that closely follows the range of its primary host, tamarack (*Larix laricina* (Du Roi) K. Koch) (Hopkins, 1909; Seybold et al., 2002). ELB feeds on the phloem of tamarack and cuts off the transportation of sugars, proteins, and other necessary molecules by tunneling underneath the bark, leading to tree mortality (Hopkins 1909; Seybold et al., 2002). ELB tend to emerge synchronously in the spring from their overwintering trees and mass attack, allowing them to overcome tree defenses. While attacks from ELB were previously scattered and isolated, recent developments in the Great Lakes region show that attacks may be evolving due to the changing climate, as evidenced by the unprecedentedly long and extensive outbreak occurring in northern Minnesota (Langor and Raske, 1989; Mckee et al., 2022).

Disturbances like defoliation and other stress causing events like droughts, flooding, and windthrow are known risk factors for ELB infestations (Crocker et al., 2016; Ward and Aukema, 2019). Climate changes could also result in the increased frequency and intensity of some of these disturbances that serve as predisposing factors, further increasing the disturbance impact on tamarack (Dale et al., 2001). However, there was no apparent predisposing factor associated with the ELB outbreak occurring in Minnesota thus creating concern about increased mortality of tamarack since this species has an important ecological, economical, and cultural role (Boyd et al., 2013).

Tamarack is generally known as a shade intolerant, potentially fast growing, pioneer conifer species on bog sites (Brown, 1982; Johnston, 1990). However, a recent study shows that it may be more shade tolerant than previously thought (Shaunette, 2022). Across its range from eastern Canada and northeastern United States to Alaska, it can be found growing on a wide variety of sites and climatic conditions (Johnston, 1990). While it is more common to find tamarack on drier sites in the northern part of its range, the Great Lakes region represents the southern extent of its range, where tamarack is often found in poorly drained, forested swamps and bogs (Fowells, 1965). In these sites, tamarack can compete well due to its tolerance to cold, wet, and acidic soils (Duncan, 1954; Jeffers, 1975; Johnston, 1980; Johnston, 1990; Heinselman, 1970).

Successful regeneration and persistence of tamarack across the landscape may face some barriers. A tamarck's seed rain does not fall far from the parent trees and germination can be limited due to lack of preferred seedbed and/or impacts from predation, bateria, or fungi (Brown, 1982; Brown, 1988; Davis et al., 1996; Duncan, 1954; Johnston, 1990). Preferred seedbeds may vary by site, for example, in Alaska, mineral seedbeds seemed to suit regeneration best in riverbottom sites, but feather moss or sphagnum moss was better suited on bog sites (Brown 1982, 1988). If a seedling can be established on a site, some level of disturbance may be required to maintain tamarack presence; tamarack regeneration is rare in dense stands (Brown, 1982; Shaunette, 2022). Further exacerbating the challenges to tamarack persistence and regeneration is the declining amount of suitable habitat in its southern range due to climate change (Janowiak et al., 2014).

Studies conducted on ELB and tamarack in northern Minnesota have been able to identify some potential relationships between infestation and stand structure and composition. There is strong evidence to suggest an initial host selection preference of larger diameter tamarack, but it is noted that as populations approach epidemic levels or stands increase in density this selectivity may decrease (Crocker et al., 2016; Langor and Raske, 1987; Mckee et al., 2022; Shaunette, 2022; Werner, 1986) .

Given the limited research on ELB dynamics in Wisconsin, the overarching goal is to explore if ELB dynamics are similar to neighboring Minnesota. Specifically, forest inventory plots were sampled to analyze relationships between ELB presence and abundance and elements of forest structure and composition like species diversity, density, regeneration, and diameter sizes. Additionally, forest inventory plots were selected using a range of different detection methods to explore similarities or differences among detection methods related to structure, composition, and ELB presence. Increasing this understanding could help guide management efforts across the state and the broader Great Lakes region in the face of a changing climate and disturbance dynamics.

Methods

Eastern larch beetle history in the Lake States

While ELB has been considered a benign bark beetle, outbreaks have been recorded during the late 20th century across eastern North America and Alaska, however, they typically followed defoliation events or other predisposing factors, resulting in researchers and managers placing more importance on the initial factor (e.g. environmental conditions compared to ELB) (Langor and Raske, 1989). Several infestations of ELB have been recorded in the states of Minnesota and Michigan in the 20th century (Langor and Raske, 1989). More recently, an outbreak has been occurring in northern Minnesota with no predisposing factors since 2001, and as of 2023 it has impacted over a million acres (MN DNR, 2024). In the neighboring state of Wisconsin, increasing ELB damage was noticed initially in 1999, and aerial sketch mapping for the insect has taken place since 2012 (excluding the year 2020). Recent ELB damage in Wisconsin has often been found occurring in association with flood damage and larch casebearer infestations (WI DNR 2021, WI DNR 2020). However, no formal research has been conducted on the interactions of ELB and tamarack in this region of Wisconsin.

Tamarack Forest Ecosystems in Wisconsin

Currently, most tamarack is found in the northern part of the state, with lesser amounts found in the central region (WI DNR, 2019b). In Wisconsin, mixed and homogenous stands of tamarack can be found, but tamarack is more often found in smaller pockets and less often in extensive homogenous stands than tamarack forests in northern Minnesota (Johnston 1990, Curtis 1959). When tamarack occurs in mixed stands, one of its most common associates is black spruce (*Picea mariana* (Mill)) but it can also be found with other species like red pine (*Pinus resinosa* Aiton), white pine (*Pinus strobus* L.), balsam fir (*Abies balsamea* L.), northern white cedar (*Thuja occidentalis* L.), black ash (*Fraxinus nigra* Marsh.), red maple (*Acer rubrum* L.), quaking aspen (*Populus tremuloides*), and white spruce (*Picea glauca* (Moench) Voss) (Curtis, 1959; Johnston, 1990; Johnston, 1980). In Wisconsin, the volume of tamarack is the highest in the 5-8.9 inch diameter class when considering growing stock (trees over 5 inches diameter) volume (WI DNR, 2019b). Tamarack in Wisconsin are aging; while volume and number of trees in all size classes are increasing, the largest increase appears to be in saw-timber sized (13+ inch diameter) trees (WI DNR, 2019b).

Site Description and Study Extent

The circular study region includes public lands within a 40.2 kilometer radius. It is located in north central Wisconsin and excludes the small area of Iron County (Figure

4.1). This region is notable for its many wetlands, kettle lakes, and forests and is an upland peneplain (Martin and Twaites, 1916; Pohlman et al., 2006). The most abundant forest types are aspen/birch, spruce/fir, red/white/jackpine, and maple/beech/birch. It is important to note that the forested landscape here has been significantly altered by intensive logging that occurred in the late 19th century (Roth, 1898). The climate can be described as having cold winters, with an average low of 0.4℃ in January, and warm summers with an average high of 25.6℃ in July (National Oceanic and Atmospheric Association [NOAA]).

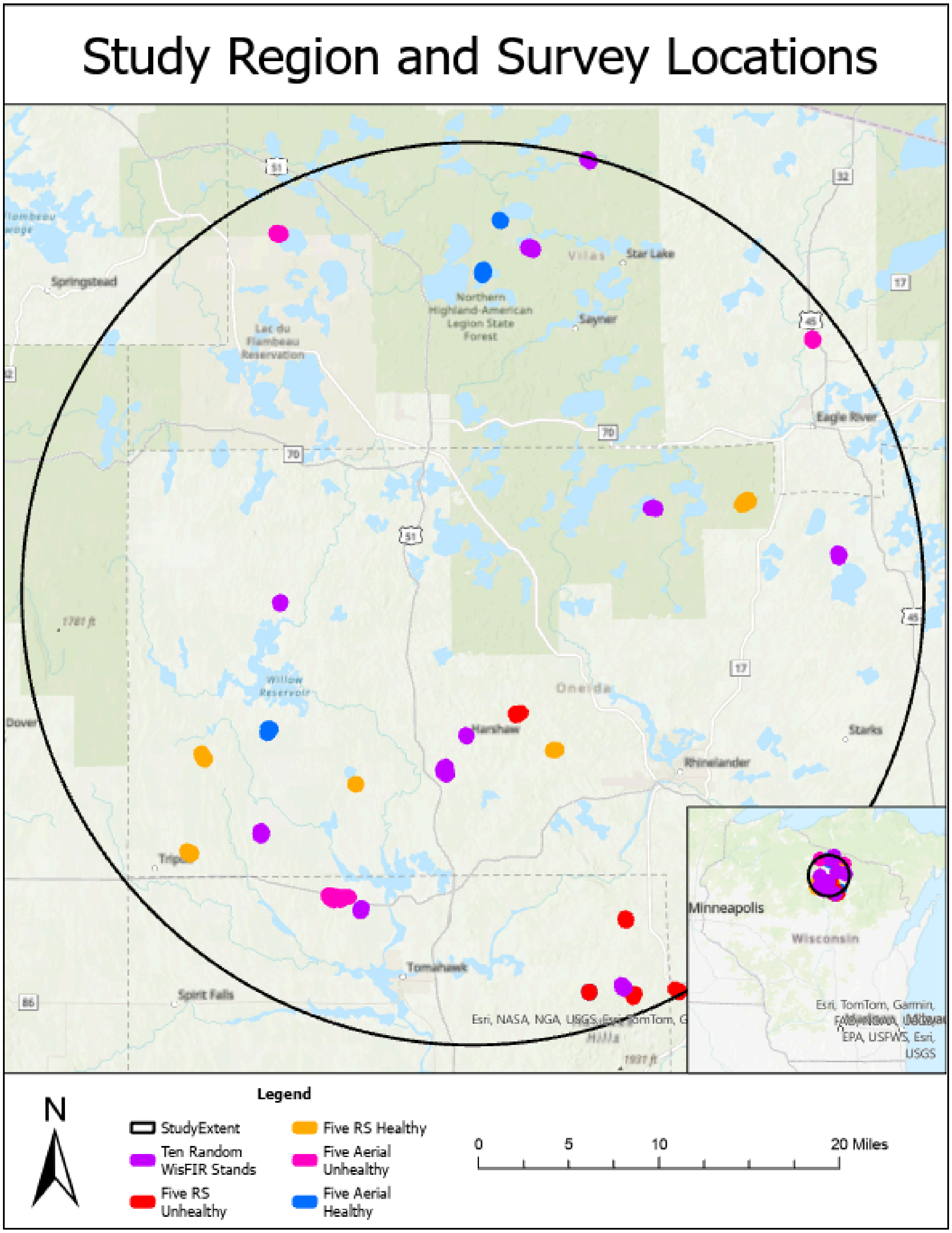


Figure 4.1 Map of study extent in north central Wisconsin with a centroid at -89.64765386, 45.78185522 that covers approximately 500,000 hectares.

Site Selection

Using a combination of detection methods, 30 stands or polygons with multiple plots were sampled (**Appendix 4A**). Specifically, samplings occurred in 10 WisFIR tamarack stands (no detection method used in selection), 10 Aerial sketch-mapped detected stands (Five Healthy and Five Unhealthy), 10 Astrape detected stands (Five Healthy and Five Unhealthy). One site/six plots were randomly selected for both the Five Aerial Healthy and Five RS Unhealthy survey types). In order to be eligible for survey, polygons and stands had to be 5-60 acres in size, within 0.5 miles of a road, and located on public property. Additional details are described below and in Francart (2024).

The Wisconsin Forest/Field Inventory and Reporting System Public Lands application (WisFIRS) was used to identify stands of tamarack. Ten stands from WisFIRS were randomly selected to be surveyed. WisFIRS stands were also used to confirm tamarack presence in aerial sketch-map polygons and a change detection algorithm based on satellite imagery (Astrape).

Aerial Sketch-Map Surveys

Aerial polygons were intersected with WisFIRS stands to find stands confirmed as tamarack and as unhealthy. A WisFIRS stand was classified as unhealthy if it had more than 30% overlap with an aerial detection polygon which is in line with other bark beetle species (Meddens et al., 2012; Langor and Raske, 1989). Five aerially detected, as unhealthy, WisFIRS stands were randomly selected. Additionally, five aerially detected, as healthy, WisFIRS stands were randomly selected. These healthy stands were tamarack stands that did not overlap with aerial polygons significantly.

The aerial sketch map survey includes polygons detected as having ELB disturbance from 2012-2021 (excluding 2020). The state of Wisconsin conducts these surveys with small crews in a plane flying between 1,000 to 2,000 feet with the assistance of the Forest Service app Digital Aerial Sketch Mapper and background maps. When a crew member identifies an area of disturbance on the ground, they draw a correlating polygon on the application where the disturbance would be located. While flights do not typically follow a predetermined pattern from year to year, they often depend on previous survey locations and forester input.

Satellite Imagery Remote Sensing Algorithm (Astrape)

WisFIRS stands were also used in combination with a satellite imagery change detection algorithm (Astrape) to locate unhealthy and healthy tamarack stands across the landscape. Astrape leverages imagery from Sentinel-2 and Planet Dove imagery to segment and classify disturbance using Jenks Natural Breaks and Extreme Gradient Boosting (see Wegmueller and Townsend, 2021 for additional detail). In this case, images from July 2018 and July 2021 were used to develop the output needed for this study. The output was a 10x10m raster dataset with cells ranked from 1-9 on change detection severity, with 9 being the most severe.

It was noticed in initial field surveys conducted in the summer of 2022 that stands overlapping with the 9 category of disturbance frequently contained areas of tamarack infested with or killed by ELB. Categories 8-9 were chosen as a suitable severity level level, and those areas were intersected with WisFIRS tamarack stands. A stand was considered unhealthy if it had more than 30% overlap with the highest category of change detection (8-9). Five stands detected as unhealthy and five detected as healthy with this method were randomly selected to be surveyed and inventoried to assess stand dynamics.

Survey Protocols

The number and location of plots sampled was dependent on access points and stand and polygon size. A grid cell map (45x45m) was overlaid on entire WisFIRS stands to locate sampling plots. Plots were located by navigating to the centroid of grid cells. Plots were taken every xth grid cell (see **Table 4.1** for grid cell selection guide) in a snaking pattern starting from a suitable “corner”. A minimum of five plots were sampled per site. Inventory plot locations occasionally had to be adjusted when it landed in inaccessible locations (e.g. open water) or was along a hard edge like a different stand type or road, if physically accessible.

Table 4.1 Inventory plot selection guide showing stand size and plot sampling frequency.

Size of Stand (hectares)	Plot Selection
2.0 - < 4.0	Every other cell
4.0 - < 12.1	Every 5th cell
12.1 - 24.3	Every 8th cell

Inventory Protocol

Trees and shrubs

Trees and shrubs (woody species that can compete with trees) were counted and measured in circular plots. In the large (6.55m radius) circular plot (purple in Figure 4.2) small trees (7.6cm < DBH ≤ 12.7cm) were tallied by species and status (live/dead) and large trees (DBH >12.7cm) were recorded on an individual basis, taking note of species, DBH, status (live/dead), and presence of beetle holes. This study only looked for signs of eastern larch beetle activity in large trees over 12.7 cm because previous studies have suggested ELB does not prefer trees smaller than that (Mckee 2015). In the four small (1.78m) circular plots (gray in Figure 4.2), trees and shrubs were tallied by species and status (live/dead) in two size classes (Seedlings: height > 0.3m and DBH ≤ 2.5cm and Saplings: 2.5cm < DBH ≤ 7.6cm).

Ground cover functional groups, Invasive Species, & Densiometer Reading

Each plot contained four 1m^2 square ground cover quadrats (green squares in Figure 4.2). Quadrats were established by walking five meters in each cardinal direction, beginning to the north. The left edge of each quadrat aligned with the transect line and started at the end of the line; an offset occurred if a tree with a diameter greater than 12.7 cm fell within the frame. Cover was recorded using the Braun-Blanquet cover classes for each functional group; cover classes are <1%, 1-5%, 6-25%, 26-50%, 51-75%, and >75% (Table 4.2). For functional groups and invasive species, those that were rooted within the quadrat were counted; function groups were trees/shrubs ≤ 12.7cm dbh, ericaceous/small shrubs, woody vines, ferns/fern allies, graminoids, other herbaceous, sphagnum moss, other moss, standing water, bare ground, rocks, leaf/needle litter, fine woody debris, coarse woody debris, dead vegetation, and other . The percentage is based on foliage/canopy cover, so the total percentage could be greater than 100% within the plot.

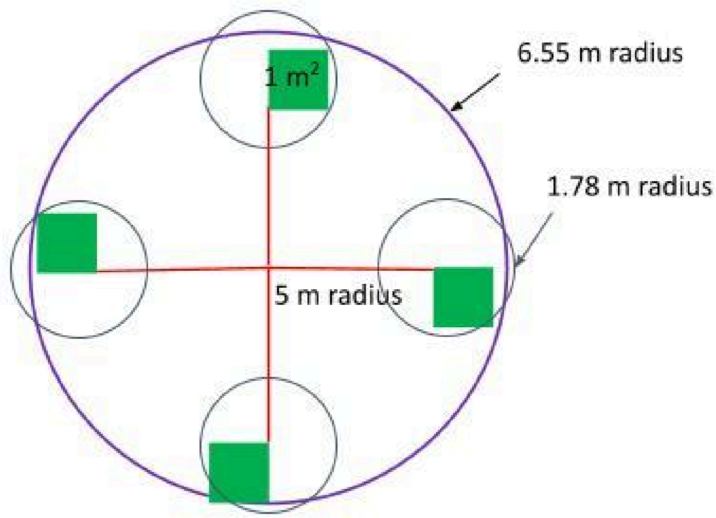


Figure 4.2 Fixed-radius plot layout for ELB and tamarack inventory

A densiometer reading was taken at each quadrat subplot facing the direction of the transect line (out away from plot center) using the strickler method (Strickler 1959, **Appendix 4B**). Using 17 estimation points (upper wedge shown below), points covered by canopy were counted and recorded.

Tree cores

At each plot one core was taken from a live tamarack tree in a dominant or co-dominant position (**Appendix 4C**) with a DBH > 12.7 cm. If there was no overstory tamarack tree, the most dominant species was cored instead. Core ID, tree species, DBH, height, and live crown ratio (**Figure 12.4**) were recorded. Trees were located for coring by choosing the first suitable tree encountered to the north. If there were no suitable trees to the north, the next suitable tree in the clockwise direction was chosen. If no suitable trees were available in the plot, the first suitable tree north of the plot but within the cell was chosen.

Analysis

Data was collected and analyzed using Google Sheets, Microsoft Excel, and R programming language. Some entries of data were removed for quality purposes. Seven seedling entries (0.5% of data) and two large tree entries (0.14% of data) were excluded from analysis due to no recorded live or dead status.

In the large tree size, trees were identified to species, and classified by mortality status, and presence of ELB, and these groups were largely determined by frequency of occurrence in the surveys. Large tree groups included live tamarack without ELB, live tamarack with signs of ELB, dead tamarack without ELB, dead tamarack with signs of ELB, live black spruce, live balsam fir, live red maple, live northern white cedar, other live species [sugar maple (*Acer saccharum* Marshall), yellow birch (*Betula alleghaniensis* Britton), paper birch (*Betula papyrifera* Marsh.), black ash, jack pine (*Pinus banksiana*), red pine, white pine, quaking aspen, red oak (*Quercus rubra* L.), and hemlock (*Tsuga canadensis (L.) Carr*)], and non-tamarack dead species (which also included unidentifiable dead trees).

In small trees, saplings, and seedlings, stems were identified to species, and mortality status. ELB presence was not noted for these size categories. These sizes were grouped into live tamarack, dead tamarack, live black spruce, live balsam fir, live red maple, other live, and non-tamarack dead species. The other live in the small trees category included sugar maple, serviceberry (*Amelanchier* spp.), yellow birch, paper birch, black ash, red pine, white pine, quaking aspen, red oak, northern white cedar, and American elm (*Ulmus americana* L.). For saplings and seedlings, the following species were grouped into a live woody shrubs category: mountain maple (*Acer spicatum*), alder (*Alnus* spp.), serviceberry, black chokeberry (*Aronia melanocarpa*), bog birch (*Betula pumila*), dogwood (*Cornus* spp.), hazelnut (*Corylus* spp.), holly (*Ilex* spp.), honeysuckle (*Lonicera* spp.), chokecherry (*Prunus virginiana* L.), willow (*Salix* spp.), alder leaved buckthorn (*Rhamnus alternifolia*), and mountain ash (*Sorbus* spp.). Species grouped into a separate other live species category for seedlings and saplings include sugar maple, paper birch, yellow birch, black ash, green ash, glossy buckthorn (*Frangula alnus*), jack pine, white pine, quaking aspen, black cherry (*Prunus serotina* Ehrh*.)*, red oak, northern white cedar, and American elm. Paper birch was put in the other live category for seedlings and sapling due to low frequency of occurrence, but its occurrence was more frequent in the small tree category.

Ground cover Braun Blanquet cover classes were converted to values representing their midpoint. Plot values were calculated by averaging these percentages across the four subplots in the plot. Densiometer point counts were kept as the same values and averaged in the same manner to be compared as relative values.

Data was initially summarized on a plot level, then stand level, and subsequently by stand/survey type (10 Random WisFIRs, Five Aerial Healthy, Five Aerial Unhealthy, Five Remote Sensing Healthy, and Five Remote Sensing Unhealthy). Summary statistics included mean and standard errors on ground cover, closed canopy coverage, small tph, saplings per hectare, and seedlings per hectare, large tree tph, large tree basal area per hectare, and large tree average diameters were calculated, and one way analysis of variance along with Tukey HSD was used to find significant differences ($p < 0.1$) between the different survey types and the calculated summary statistics.

Results of the initial Anova and Tukey HSD comparing survey types led to the usage of scatter plots, ANOVAs (followed by Tukey HSD $p<0.1$), Welch's t-tests ($p<0.1$), and simple linear regression ($p<0.1$) to explore relationships between stand characteristics at a plot level. ANOVAs were used to assess differences in means for seedling and sapling regeneration, small tree recruitment, species diversity (Shannon diversity index), and tree density (tph) using the presence of signs of ELB in a plot as a grouping factor. The presence of signs of ELB consisted of three categories: plots with large size tamarack and signs of ELB, plots with large size tamarack and no signs of ELB, and plots that did not have any large size tamarack. Average diameter means were compared using Welch's t-test. Linear regressions were used to examine relationships between the proportion of large tamarack trees with ELB in a stand and diversity, density, and regeneration, and these tests excluded plots without overstory tamarack. Linear regressions were also used to examine relationships between large trees and regeneration and plots without large tamarack were not excluded from these.

To explore potential community dynamics at both the stand and the plot level, a non-metric multidimensional scaling (NMS) ordination was run using TPH across the vegetation layers (large trees, small trees, saplings, and seedlings) by status (live and dead) (PC-Ord Version 6.255) (McCune and Mefford, 2011). The plot level species matrix included 137 species recorded in the large tree, small tree, sapling, and seedling layer by live or dead (e.g. black spruce was live and dead in all layers of the vegetation so a total of 6 columns of data for this species) included all species recorded (n = 137). Due to high stress during initial runs of the plot level ordinations, rare species were removed. Species which occurred in ≤ 7 (3.5%)of the plots were removed; a total of 83 species were removed resulting in retention of 53 species across all three vegetation layers. The environmental matrix (secondary matrix) included detection method, presence of ELB in live and dead standing trees, and percent cover of the understory vegetation by functional class to explore relationship to environmental variables. NMS was used due to relaxed assumptions of normality (McCune et al. 2002). Autopilot mode (slow and thorough) was selected using Sørensen (Bray–Curtis) distance measurement and a random starting configuration. Two hundred and fifty runs were completed for both the real and randomized data to determine dimensionality. Species and environmental variables with an r^2 (squared correlation coefficient) greater than 0.2 were considered meaningful (Wilson et al. 2013). A one-factorial permutation-based Multi-Response Permutation Procedures (MRPP) was run using Sørensen (Bray–Curtis) distance to explore differences among the different detection methods at both the plot at stand level.

Results

Overview

Across all detection methods and stands, a total of 198 plots and 29 unique sites were inventoried. A total of 835 tamarack trees greater than 12.7 cm were measured with the vast majority being alive (n = 693, 83%). Fifteen unique tree species were identified in the large and small tree categories, and 31 unique woody species were identified in the seedling and sapling layer. Of all the plots surveyed, invasive species only showed up in one plot in the ground cover quadrats (cattail (*Typha* spp.)) and in four plots in the seedling and sapling layer (glossy buckthorn (*Frangula alnus* Miller)).

Of the large tamarack trees, 2.9% of live trees had signs of ELB and 90.8% of dead trees had signs of ELB. At the site level, there was variation with levels of ELB infestation observed within the plots. Across all detection types, approximately a third of plots with tamarack contained ELB, and the average proportion of trees with ELB in a plot was 58% (Table 4.3). Within plots, the proportion of trees with ELB showed a significant positive linear relationship with the amount of dead tamarack trees (p=2.2e-16; $R^2 = 0.599$).

Table 4.3 Proportions of plots with large tamarack (LALA), ELB, and average proportion of large trees with ELB (excluding plots without ELB) and standard error across the study.

Stand Type	Plots w/ LALA	Proportion w/ ELB	Avg Proportion of Trees w/ ELB
All Stands	142	.27	.58 (± .05)
Ten Random WisFIRS	46	.28	.57 (± .10)
Five Aerial Healthy	31	.10	.43 (± .04)
Five Aerial Unhealthy	19	.37	.69 (± .08)
Five RS Healthy	28	.32	.63 (± .12)
Five RS Unhealthy	24	.33	.49 (± .14)

Stand Composition and Structure by Detection Type

There were no significant differences in large tree tph or basal area per hectare of total live trees, live tamarack without ELB, live tamarack with ELB, dead tamarack without ELB, dead tamarack with ELB, or total dead trees between detection types (Table 4.4). Among detection types, the average total live tph across species ranged from 214.5 ± 93.8 standard error in Aerial Unhealthy to 565.6 ± 150.9 in Aerial Healthy. Average live tamarack tph ranged from 173.1 ± 83.5 in Aerial Unhealthy to 290.0 ± 87.9 in Remote Sensing Unhealthy and average dead tamarack with signs of ELB ranged from 10.7 ± 7.3 in Aerial Healthy to 90.5 ± 67.4 in Aerial Unhealthy.

While many aspects of stand structure were not significantly different among detection types, average diameter and other stand features were significantly different. Average diameter of dead large tamarack without ELB in Remote Sensing Unhealthy and Aerial Healthy was significantly smaller than Aerial Unhealthy ($p = 0.045$, $p = 0.047$ respectively). Average diameter of live large trees was significantly smaller in Aerial Unhealthy than in Random WisFIRS ($p = 0.070$) and Remote Sensing Healthy ($p = 0.046$). Dead sapling tph was greater in Remote Sensing Healthy than Aerial Unhealthy ($p = 0.022$). Dead small tph was greater in Aerial Healthy than in Remote Sensing healthy ($p = 0.031$). Total live seedling tph was greater in Aerial Unhealthy than in Aerial Healthy ($p = 0.077$). Dead tamarack seedling tph was greater in Aerial Unhealthy than in Aerial Healthy ($p = 0.008$), Remote Sensing Healthy ($p = 0.025$), and Remote Sensing Unhealthy ($p = 0.017$). Live tamarack seedling tph was greater in Aerial Healthy than in any other survey type ($p = 0.014$). There were significant differences in the canopy cover among detection methods. The Aerial Unhealthy stands had a significantly lower amount of closed canopy points (9.31 ± 2.53) than Random WisFIRS (14.3 ± 0.59, $p = 0.016$), Aerial Healthy (14.6 ± 0.267, $p = 0.034$), and Remote Sensing Healthy (14.4 ± 0.533, $p = 0.042$) stands.

Table 4.4 Mean, standard error, and significance (Anova and Tukey HSD) of tree species mortality status, and ELB presence across the five different types of detection for ELB and tamarack. Values were averaged at plot level, site level, and finally at detection type.

Survey Type and Tree Category	Large Tree Basal Area (Sqm/Hectare)			Large Tree TPH			Large Tree Average Diameter (cm)			Small Tree TPH			Saplings TPH			Seedlings TPH		
	Mean	Standard Error	(p<0.1)	Mean	Standard Error	(p<0.1)	Mean	Standard Error	(p<0.1)	Mean	Standard Error	(p<0.1)	Mean	Standard Error	(p<0.1)	Mean	Standard Error	(p<0.1)
Total Live Trees																		
10 Random WisFIR	16.6	2.6	a	512.4	57.4	a	19.6	0.8	a	451.9	58.5	a	1899.3	361.1	a	17283.1	4138.3	ab
5 Aerial Healthy	19.2	7.4	a	565.6	150.9	a	18.7	1.2	ab	499.8	79.1	a	930.9	173.9	a	9744.7	2030.6	a
5 Aerial Unhealthy	4.2	1.9	a	214.5	93.8	a	15.3	0.5	b	380.2	153.3	a	1261.7	88.2	a	37220.2	11874.7	b
5 RS Healthy	12.0	2.1	a	421.0	79.9	a	20.5	1.6	a	353.2	32.7	a	2466.9	775.2	a	32000.3	4852.7	ab
5 RS Unhealthy	18.3	6.9	a	469.3	129.7	a	19.7	1.2	ab	359.2	38.5	a	1335.5	391.5	a	20771.9	8336.8	ab
Live Tamarack without ELB																		
10 Random WisFIR	8.2	2.0	a	245.2	50.9	a	22.1	2.5	a	138.4	51.6	a	144.1	75.1	a	228.3	93.0	a
5 Aerial Healthy	8.8	3.6	a	252.7	50.5	a	19.8	2.7	a	173.9	49.1	a	107.4	41.3	a	199.6	120.2	a
5 Aerial Unhealthy	3.5	1.8	a	173.1	83.5	a	16.0	0.8	a	180.8	47.0	a	311.4	122.5	a	1844.4	927.3	b
5 RS Healthy	8.5	2.3	a	290.0	87.9	a	21.6	2.1	a	148.3	48.7	a	97.1	36.3	a	95.7	47.4	a
5 RS Unhealthy	10.3	3.3	a	272.0	42.0	a	20.7	2.3	a	166.5	58.9	a	78.7	56.6	a	179.2	91.3	a
Live Tamarack with ELB																		
10 Random WisFIR	0.4	0.2	a	10.4	4.8	a	22.5	4.8	a	-	-	-	-	-	-	-	-	-
5 Aerial Healthy	0.0	0.0	a	1.6	1.6	a	15.7	-	a	-	-	-	-	-	-	-	-	-
5 Aerial Unhealthy	0.2	0.2	a	8.9	8.9	a	14.7	-	a	-	-	-	-	-	-	-	-	-
5 RS Healthy	0.8	0.5	a	16.1	9.1	a	25.7	7.5	a	-	-	-	-	-	-	-	-	-
5 RS Unhealthy	0.1	0.1	a	2.5	2.5	a	26.4	-	a	-	-	-	-	-	-	-	-	-
Dead Tamarack without ELB																		
10 Random WisFIR	0.2	0.1	a	8.2	3.7	a	18.7	1.2	ab	15.0	4.6	a	19.8	11.7	a	78.6	24.8	ab
5 Aerial Healthy	0.0	0.0	a	1.6	1.6	a	13.0	-	b	17.0	9.1	a	14.6	9.9	a	0.0	0.0	b
5 Aerial Unhealthy	0.2	0.2	a	3.0	3.0	a	27.0	-	a	53.4	29.2	a	64.9	35.4	a	190.0	68.9	a
5 RS Healthy	0.2	0.2	a	4.2	4.2	a	20.7	-	ab	14.4	8.5	a	39.9	19.9	a	25.1	16.7	b
5 RS Unhealthy	0.1	0.1	a	5.1	3.2	a	14.7	1.1	b	32.4	12.1	a	37.3	29.2	a	16.7	16.7	b
Dead Tamarack with ELB																		
10 Random WisFIR	3.1	1.3	a	64.0	29.3	a	25.2	3.7	a	-	-	-	-	-	-	-	-	-
5 Aerial Healthy	0.4	0.3	a	10.7	7.3	a	20.3	3.6	a	-	-	-	-	-	-	-	-	-
5 Aerial Unhealthy	3.0	2.5	a	90.5	67.4	a	18.4	2.5	a	-	-	-	-	-	-	-	-	-
5 RS Healthy	2.2	1.1	a	50.2	26.9	a	23.8	3.4	a	-	-	-	-	-	-	-	-	-
5 RS Unhealthy	2.1	1.3	a	62.3	34.1	a	17.8	1.9	a	-	-	-	-	-	-	-	-	-
Total Dead Tres																		
10 Random WisFIR	4.5	1.5	a	111.1	34.4	a	21.7	2.0	a	78.6	13.4	ab	359.6	74.9	ab	1211.0	305.7	a
5 Aerial Healthy	2.5	0.9	a	104.5	31.0	a	15.8	0.8	a	147.2	31.7	a	321.6	40.8	ab	965.6	266.1	a
5 Aerial Unhealthy	3.3	2.7	a	96.5	73.2	a	18.2	2.3	a	78.6	29.9	ab	179.6	55.1	a	1081.7	289.3	a
5 RS Healthy	2.6	1.3	a	64.0	34.7	a	21.7	3.2	a	45.8	12.5	b	638.6	165.7	b	1767.0	300.5	a
5 RS Unhealthy	3.6	1.2	a	130.9	28.7	a	16.7	0.8	a	101.8	19.2	ab	473.4	51.7	ab	1312.0	373.4	a

Plot Level Differences in Overstory Structure

Given limited differences between detection types at the stand level in the large tree size category, analysis was conducted to explore dynamics at the plot level without regard to detection type. The density of large trees across species ranged from zero tph to 1,929 tph. Density of large tamarack trees ranged from zero tph to 1,855 tph. Plots with large tamarack and signs of ELB had a significantly greater density of large tph (719 ± 73.9) compared to plots with large tamarack without signs of ELB, (540 ± 32.1, $p=0.033$) and plots without large tamarack trees (273 ± 53.0, $p = 1.0E\text{-}07$) (Figure 4.3). Plots with large tamarack without signs of ELB also had significantly more large tph than plots without large tamarack trees ($p=8.690E\text{-}05$). Plots with large tamarack did not show a significant difference in canopy cover between those with ELB and those without ELB, 13.2 ± 0.5 and 14.5 ± 0.2, respectively ($p= 0.164$). Plots without large tamarack trees had a significantly lower amount of closed canopy points (9.92 ± 0.8) than plots with large tamarack with ELB ($p = 8.6E\text{-}05$) and plots with large tamarack without ELB ($p < 0.001$). In plots with large tamarack, those with signs of ELB had a significantly higher large tamarack tph (573 ± 64.7) than those without signs of ELB (361 ± 27.7) ($p=0.004$) (Figure 4.4). When live and dead large trees were combined, there was no significant linear relationship between the density of large tamarack tph or large tree tph and the proportion of large tamarack trees with signs of ELB. The proportion of ELB and tph of dead large tamarack trees had a significant positive relationship ($p = 2.2e\text{-}16$, $R^2 = 0.599$). Inversely, the proportion of ELB and density of live large tamarack trees had a significant negative relationship ($p = 2.160E\text{-}04$, $R^2 = 0.087$).

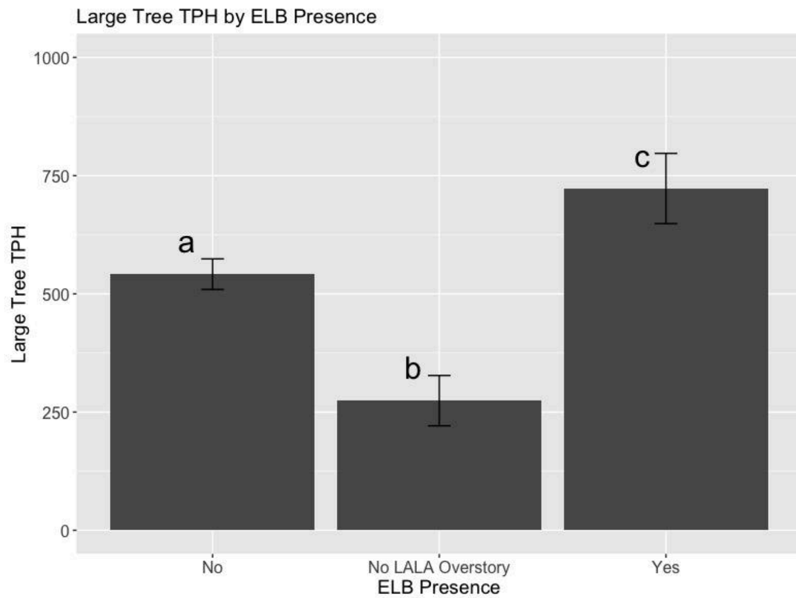


Figure 4.3 Average number of large tph of plots with large tamarack trees without ELB (left), plots with no large tamarack trees (middle), and plots with large tamarack trees with ELB (right), along with significance of differences (a, b, c).

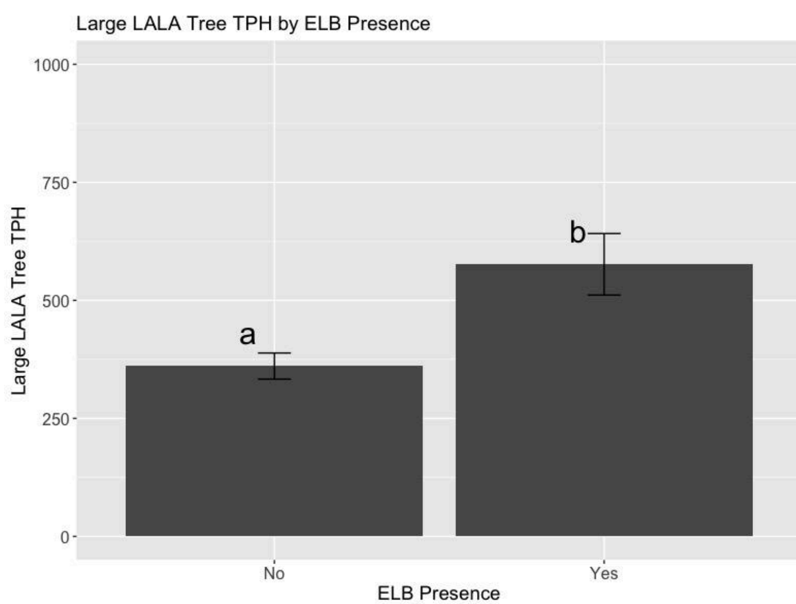


Figure 4.4 Average number of large tamarack tph of plots with large tamarack trees without ELB (left) and plots with large tamarack trees with ELB (right) along with significance of differences

(a, b).

Among plots, average diameters of large tamarack ranged from 12.9 cm to 67.2 cm. The average diameter of dead tamarack in plots without ELB was significantly smaller (15.5 ± 1.1 cm) than plots with ELB (21.0 ± 1.1 cm) ($p = 0.003$). There was no significant difference in live tamarack diameter in plots with or without ELB, 19.4 ± 0.9 and 20.0 ± 0.8 cm, respectfully, ($p = 0.003$).

In plots with tamarack, the proportion of large tamarack compared to total large trees in plots ranged from .07 to 1. Plots with tamarack with signs of ELB had a significantly higher average proportion of tamarack .84 ± .04 compared to plots without signs of ELB 0.70 ± 0.03 ($p = 0.006$). There was a significant, positive linear relationship found between the proportion of tamarack with ELB in a plot and the proportion of overstory trees being tamarack ($p = 0.089$, $R^2 = 0.0135$).

Regeneration and Recruitment

Live regeneration and recruitment of tamarack in plots ranged from 0 to 10,298 seedlings per hectare with a mean of 612.68 ± 108.14, 0 to 2,260 saplings per hectare with a mean of 176.32 ± 26.23, and 0 to 1,261 small tph with a mean of 179.86 ± 18.11. There were no significant differences between the mean amount of live tamarack seedlings or saplings in plots with large tamarack with and without ELB. (Figure 4.5). Plots without large tamarack had significantly more live tamarack seedlings (1,601 ± 327) than plots with large tamarack with ELB ($p = 1.300E-05$) and plots with large tamarack without ELB ($p < 0.001$). Plots without large tamarack also had more live tamarack saplings (274 ± 65, $p = 0.036$) than plots with large tamarack and ELB, but not plots with tamarack without ELB ($p = 0.141$). There was significantly more small tamarack trees in plots without ELB (264 ± 29.2) compared to plots with ELB (122 ± 26.8, $p = 0.005$) and plots without large tamarack trees (64.9 ± 19.5, $p = 3.500E\text{-}06$) (Figure 4.6). A significant negative linear relationship had been found for both live tamarack seedlings ($p = 1.339e\text{-}08$, $R2 = 0.148$) and saplings ($p = 3.459e\text{-}05$, $R2=0.079$) when compared with large tph across species. Similarly a significant negative linear relationship was also found for live tamarack seedlings ($p = 3.746e\text{-}05$, $R2 = 0.079$) and saplings ($p = 0.014$, $R2 = 0.026$) when compared to large tamarack tph.

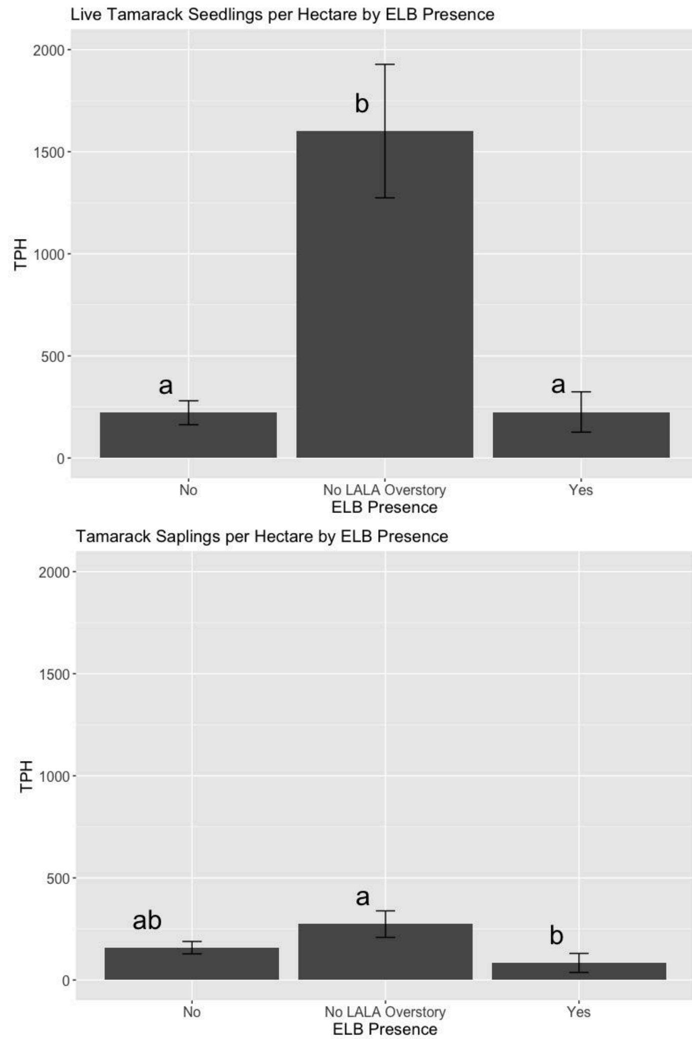


Figure 4.5 Average number of tamarack seedlings (top graph) and saplings (bottom graph) per hectare of plots with large tamarack trees without ELB (left column), plots with no large tamarack trees (middle column), and plots with large tamarack trees with ELB (right column), along with significance of differences (a, b, ab).

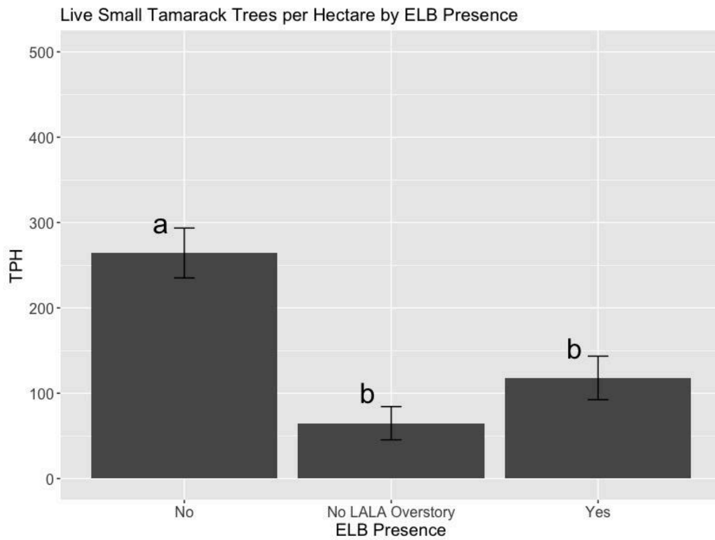


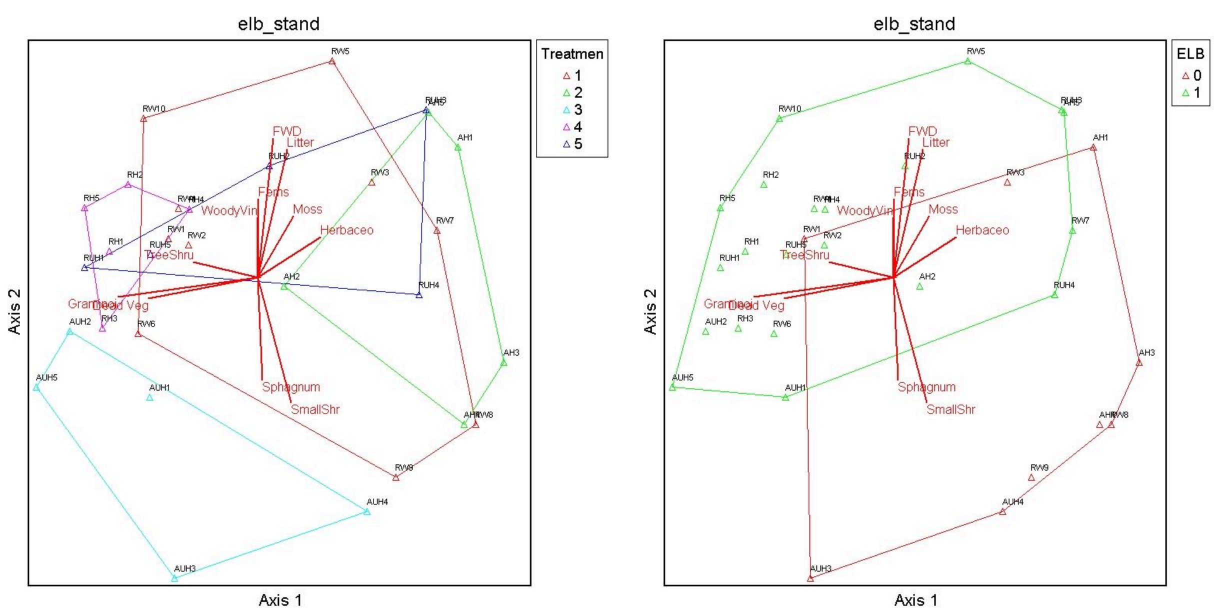
Figure 4.6 Average number of tamarack small tph of plots with large tamarack trees without ELB (left column), plots with no large tamarack trees (middle column), and plots with large tamarack trees with ELB (right column),, and with ELB along with significance of differences (a, b).

Diversity

Shannon index values in plots ranged from 0 to 1.6. There was no significant difference between plots with or without signs of ELB , 0.317 ± 0.07 and 0.460 ± 0.04, respectively. Plots without large tamarack had a significantly lower diversity index 0.272 ± 0.05 than plots with large tamarack but no signs of ELB (p=0.021).

Community Relationships at the Stand and Plot Level

In exploring community composition at the stand level, a two-dimensional NMS ordination had a final stress of 11.8 and final instability of 0.000. Eighty-six percent of the variation was explained in the ordination with axis 1 and axis 2 explaining 48.9% and 37.2%, respectively (Figure 4.7). Eleven environmental variables were strongly correlated with axis 1 or axis 2 (Table 4.5). While MRPP noted significant differences among all the treatments except for 1 versus 5 and 2 versus 5, there was significant overlap in ordination space for the majority of the different detection and forest health combinations outside of the aerial detection, healthy polygons which is observed in the lower left area of the plot (Table 4.5). Given this overlap, the grouping category was updated to ELB detected in a stand or absent. There was greater differentiation in ordination space between these two categories. MRPP detected significant differences between ELB presence and absence ($p = 0.0003$)



A **B**

Figure 4.7 Non-metric multidimensional scaling (NMS) ordination displaying the relationship among (A) different detection and status stands or polygons and (B) between the presence or absence of ELB. The ordination resulted in a two-dimensional solution with 48.9 % and 37.2%

of the variation explained in axis 1 and axis 2, respectfully. Environmental variables with r^2 values greater than 0.2 are displayed within ordination space. Detection and status codes (A) are 1 . ELB (B) presence is associated with 1 and absence is 0.

Using plot level data, the NMS ordination similarly resulted in a two-dimensional ordination with a final stress of 19.7. Axis 1 and axis 2 explained 48.1% and 29% of the variation, respectfully for a total of 77.2% of the variation explained. Six environmental variables were strongly correlated with axis 1 or axis 2 (Table 4.6). While there was significant overlap in ordination space for the detection and status types, MRPP noted significant differences among all the different detection and status types (Table 4.6). Additionally, significant differences were observed for plots with and without ELB (p = 0.014).

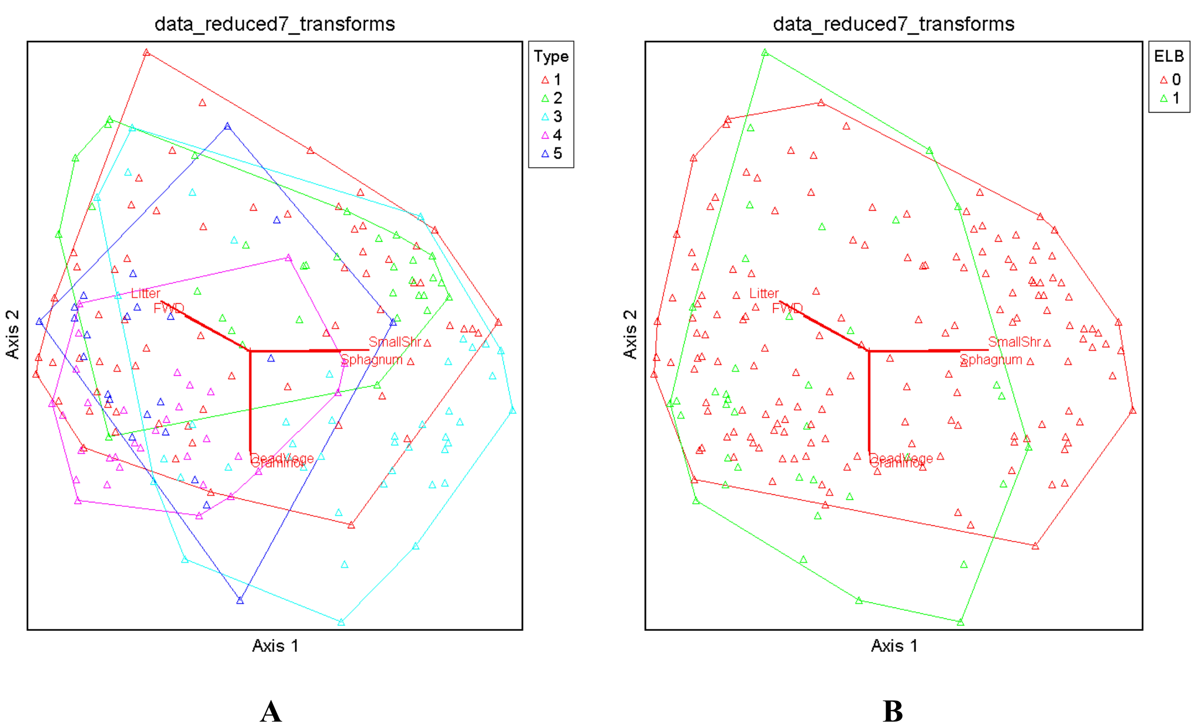


Figure 4.8 Non-metric multidimensional scaling (NMS) ordination displaying the relationship

among (A) different detection and status plots and (B) between the presence or absence of ELB at the plot level. The ordination resulted in a two-dimensional solution with 48.1% and 29% of the variation explained in axis 1 and axis 2, respectfully. Environmental variables with r^2 values greater than 0.2 are displayed within ordination space. Detection and status codes (A) are 1 . ELB (B) presence is associated with 1 and absence is 0.

Table 4.5 Environmental variables with strong correlations to axis 1 or 2 in ordination space. A correlation was considered strong if the R^2 was 0.2 or greater. All variables are percent cover variables.

Stand Level Ordination				Plot Level Ordination			
Axis 1		Axis 2		Axis 1		Axis 2	
Trees & shrubs	0.248	Small shrubs	0.483	Small shrubs	0.418	Graminoids	0.366
Graminoids	0.542	Woody Vines	0.230	Sphagnum	0.317	Dead vegetation	0.350
Herbaceous	0.243	Ferns	0.302	Litter	0.316		
Dead vegetation	0.425	Sphagnum	0.397	FWD	0.228		
		Moss	0.238				
		Litter	0.494				
		CWD	0.182				

Table 4.6 Mutli-response permutation procedures (MRPP) multi-comparison p-values for differences among detection types and forest health status at the stand and plot level.

Comparison	Stand Level p-value	Plot Level p-value
Random WisFir vs Aerial Healthy	0.096	>0.001
Random WisFir vs Aerial Unhealthy	0.011	>0.001
Random WisFir vs Remote Sensing Healthy	0.029	>0.001
Random WisFir vs Remote Sensing unhealthy	0.912	0.002
Aerial Healthy vs Aerial Unhealthy	0.003	>0.001
Aerial Healthy vs Remote Sensing Healthy	0.002	>0.001
Aerial Healthy vs Remote Sensing unhealthy	0.431	>0.001
Aerial Unhealthy vs Remote Sensing Healthy	0.006	>0.001
Aerial Unhealthy vs Remote Sensing unhealthy	0.011	>0.001
Remote Sensing Healthy vs Remote Sensing unhealthy	0.076	>0.001

Discussion

ELB is a common disturbance agent in northern Wisconsin tamarack forests. Approximately a third of all plots with large tamarack across all stands had signs of ELB. This is currently lower than the nearly 75% of tamarack stands in MN noted as impacted during the 20+ year outbreak as of 2023 (MN DNR, 2024). While ELB was the likely mortality agent of many of the dead tamarack with ELB signs, ELB signs and tamarack mortality were not consistent throughout the study region. Similarly, it was found in northern Minnesota that healthy stands of tamarack often had pockets of unhealthy tamarack and vice versa (Shaunette, 2022).

However, where ELB was found in this study region, on average, it was impacting over half of the large tamarack trees in a plot. In historical examples from the late 70s and early 80s, recorded mortality rates ranged from 25%-60% in outbreaks in Alaska, New Brunswick, and Nova Scotia (Werner, 1986). Most instances of ELB in this study were recorded in trees that were dead, with very few cases of live trees showing signs of ELB. This could partially be due to difficulty seeing signs, characteristic entrance, exit, and ventilation holes, in early stages of infestation or light infestation when the tree is still alive. It is likely that many trees identified with signs of ELB would have been infested at least a year prior to observations made in this study.

Detecting ELB has proven to be difficult. An original goal was to explore how stand structure and composition varied across different methods of detection and status of ELB (healthy versus healthy). While we initially expected there to be differences in the overstory (large trees) structure and composition, especially for variables related to ELB (e.g. dead tamarack), there were limited significant differences between detection methods when exploring at the stand level (Table. 4.4). However, canopy cover was different among the detection methods. The Aerial Unhealthy had significantly less canopy cover than Random WisFIRS. It is notable that two of the five stands belonging to the Aerial Unhealthy stand category had very few to no large trees.

Since there were limited differences among the detection types at the stand level, plot level analysis, not differentiating for survey type, allowed an opportunity to explore dynamics at a more local scale. This local level analysis allows for a more detailed and nuanced evaluation of the relationship of ELB to plot level mortality, density, diversity, composition, regeneration, and recruitment, especially because stand level averages do not always accurately reflect plot level measures of forest structure and composition, particularly in heterogeneous stands (Windmuller-Campione et al., 2022). Understanding these relationships at this scale is important, because many studies have identified links between these factors and impacts from forest insects. Density and tree diameter are two elements of forest structure and composition that have already been identified in previous studies to have an influence on ELB and tamarack dynamics (Crocker et al., 2016; Shaunette, 2022).

Generally, across the United States, denser forests are more prone to damage from insects, and stand density is a common metric for evaluating the susceptibility of stands to epidemic level *Dendroctonus* infestation (Asaro et al., 2023; Windmuller-Campione et al. 2021). In this study, a higher density of large trees (live and dead), when considering all species and just tamarack alone, was associated with the presence of ELB in a plot. This suggests a higher susceptibility to infestation with a higher density. However, there was no relationship between large tph across species or large tamarack tph and the proportion of large tamarack with ELB, meaning that this susceptibility may not increase proportionally with density.

Studies in Minnesota have identified differing relationships between density and ELB. In northern Minnesota, using the Forest Inventory and Analysis (FIA) database, it was initially found that density and mortality were actually negatively correlated in later stages of outbreak, possibly due to the higher abundance of larger diameter tamaracks in less dense stands (Crocker et al., 2016). However, an even later inventory study of the same region found that stands with recent and old high mortality had higher densities than stands with low mortality (Shaunette, 2022). That study also noted an overall decreased presence of larger diameter tamarack when compared to the earlier FIA study. While attacks on trees as small as a few centimeters have been recorded in previous studies, there appears to be an initial preference for larger diameter trees that may decrease as infestations continue and preferred hosts become less available (Crocker et al., 2016; Magasi, 1983; Shaunette, 2022; Werner, 1986). This emphasizes the impact that time may have on the relationship between density, diameter, and ELB as infestations progress. These relationships likely are not static throughout the progression of an epidemic and are influenced by a multitude of factors.

In this study, the average diameter of dead tamarack in plots without ELB was significantly smaller than dead tamarack in plots with ELB, suggesting an association of diameter size, tree infestation, and mortality. However, this significance was not found when comparing live tamarack in plots with and without ELB. Furthermore, the average diameter of tamarack trees in a plot and the proportion of trees with ELB were not strongly correlated. Since this study just offers a snapshot in time of infestation, it is hard to say whether these results are due to diameter being less important of a factor in a later stage of infestation as found in previous studies, or just a general lack of relationship between the factors in this region (Crocker et al., 2016; Shaunette, 2022).

Species diversity and composition can also play an important role in insect-forest dynamics and were considered in this study. The biodiversity-stability theory suggests that more diverse ecosystems have increased stability in response to the introduction of changes or disturbance. It is thought that more mixed systems experience less damage from herbivorous pests due to reduced access to host trees, increased predator impact, and diversion from other tree species, but that there are exceptions for polyphagous insects (Jactel et al., 2005). However, an evaluation of studies on this idea in forests of the boreal ecozone, have found that this proposed reduction of damage is not consistent across studies (Koricheva et al., 2006). One meta analysis study on tree and pest interactions proposes that many benefits of increased diversity might actually depend more on the proportion of host species composition (Koricheva et al., 2006; Jactel and Brockerhoff, 2007). Low availability of host species likely causes similar impacts to tree and insect interactions as higher diversity is proposed to do, such as by creating more of a barrier to locating host trees and reducing the amount of available substrate for insects.

In the *Dendroctonus* genus, many species locate their hosts using olfactory selection, and the dilution of this signal from presence of non-host trees could lead to more difficulty in locating desired host trees (Jactel et al., 2005; Zhang et al., 2004). It was also found that volatiles from some non-host species may actually act as a repellent for some *Dendroctonus* species (Dickens et al. 1992; Huber and Borden, 2001; Jactel et al., 2005; Zhang and Schlyter, 2004). Eastern larch beetles are a monophagous bark beetle with a chemical ecology that is not fully understood. Studies have identified several tamarack resin monoterpenes that the beetle is attracted to, and it is also known that female beetles, the host selecting species, emit pheromones to attract or repel other beetles; pheromones from beetles and host monoterpenes may work synergistically to attract other beetles (Mckee, 2015; Prendergast, 1991). These characteristics suggest that ELB populations could be influenced by species composition and diversity.

In this study region, there was a significant relationship found between large tree host species composition and ELB. Plots with ELB presence were found to have a higher proportion of large tamarack than plots without ELB, and this relationship appears to be linearly related as well, with higher proportions of large tamarack with ELB being positively related to the proportion of large trees that are tamarack. However, while plots with large tamarack and signs of ELB had a lower average Shannon diversity index than those tamarack plots without signs of ELB, this difference was not significant, suggesting a lack of relationship between diversity and ELB at this level. Furthermore, a linear regression further confirmed this lack of relationship when the proportion of tamarack trees with ELB were compared to the level of diversity.

A similar observation has been made in a study looking at spruce beetle, where increased Shannon diversity at the plot level did not result in increased spruce survival (Conner et al., 2014). One caveat to the result in this study of ELB is that the analysis of this observation did not incorporate other possible factors that may be influencing the presence or abundance of ELB at a plot. And while this observation suggests a lack of relationship at the plot level, this does not mean a relationship could not exist at a stand or landscape level. Previous meta-analyses have suggested that forest diversity at these levels, particularly at the landscape, may play an important role in insect and pest outbreaks (Jactel et al., 2005; Koricheva et al., 2006).

In an insect outbreak, it is also important to have an understanding of regeneration and recruitment, as this can provide insight to the future of that forest. There was a large range of regeneration and recruitment occurring across this study, with plots having anywhere from zero to thousands of live tamarack seedlings, saplings, or small tph. Variability in tamarack regeneration post ELB infestation has been recorded in other studies (Dubuque, 2019; Shaunette, 2022). In our analysis, no significant difference was found between the amount of live tamarack seedlings and sapling in plots with or without ELB, indicating that ELB infestation does not appear to be negatively or positively influencing seedling and sapling regeneration. Tamaracks are known to cast light shade, so the loss of needles from ELB may not be dramatically altering the light environment until the actual stem comes down (Johnston, 1990). Similarly, our study showed no significant difference between the amount of canopy coverage between plots with and without ELB. Some studies have suggested that stands with ELB, however, may be experiencing slower regeneration that, eventually, meets adequate stocking levels (Shaunette, 2022; Dubuque, 2019). Because our study did not evaluate timelines of infestation or age of regeneration, it cannot be said how long regeneration in this study has taken to establish or if it had established prior to infestation. Origin of regeneration, whether artificial or natural, was not considered either, however, other studies noted the ability of natural regeneration to achieve adequate stocking levels (Shaunette, 2022; Dubuque, 2019).

When looking at the number of small tamarack tph, the amount was significantly higher in plots without ELB compared to plots with ELB, suggesting that there may be a relationship between ELB presence and the recruitment of tamarack regeneration into the overstory. Similar relationships were observed in northern Minnesota, where stands with high, old mortality had significantly lower amounts of small tamarack when compared to stands with low mortality or high, recent mortality (Shaunette, 2022).

One surprising feature that was observed during sampling and analysis was the numbers of plots without a live or dead tamarack in the large size class. Some of these plots had other species in the large size class or did not have trees in the large size class at all. In general, these plots had greater tamarack seedling and sapling regeneration than plots with tamarack in the large size class.   This may be an example of the Janzen-Connell hypothesis where seedling mortality is positively related to proximity to conspecific trees. (Janzen, 1970). Adult trees have been found to influence seedling mortality by altering microenvironment, providing refuge, and interacting with natural enemies (Deniau et al., 2017; Janzen 1970).

This also could have been the result of the overall large tph being significantly lower in these plots than in plots with large tamarack, with or without ELB, resulting in better regeneration suitability for this shade intolerant, pioneer species. Canopy cover was less in plots without large tamarack than in plots with large tamarack, with or without ELB. This idea is further supported by the finding that there was a significant negative linear relationship between tamarack seedlings and saplings when compared with large tph across species and large tamarack tph. This could mean that ELB has the potential to stimulate regeneration in the long term, if it reduces large tree density. In contrast to seedling and saplings per hectare, small tamarack tph were significantly lower in plots without large tamarack when compared to plots without ELB, and were not significantly different from plots with ELB. This means that there could be other environmental factors that could be impacting the recruitment of tamarack into the large size class. For example, site quality may be influencing the ability of trees to achieve height and diameter growth sufficient to reach the overstory (Johnston, 1990). It could also be that these plots have experienced disturbance recently, and they may have simply not had time to achieve growth to small or large tree sizes.

Conclusion

Dynamics between ELB and tamarack likely vary by region, scale of area, and stage of infestation. Eastern larch beetle in north central Wisconsin appear to be common across the landscape, and associated with significant impacts to large tamarack trees where it is found, but impacts are not uniform across the study area. Several relationships between ELB and structural and compositional characteristics of tamarack systems were found to be significant. Factors like plot level density and level of host species composition were positively correlated with ELB presence. Levels of tamarack seedling and sapling regeneration were not related to ELB presence, but recruitment into the small tree size class may be negatively associated with ELB presence.

It is important to consider the broader landscape in the management of tamarack with the added challenges of a changing climate at the southern extent of its range and the increased risk of mortality from ELB. Because impacts may vary across the landscape, detection of the insect will be important to identify where management action may be needed. Furthermore, monitoring impacted stands may be required to ensure adequate regeneration and recruitment of tamarack into the overstory if that is the desired outcome for a stand affected by ELB.

There remains several areas to be further investigated with tamarack and ELB. Site quality may play a role in mortality and regeneration of tamarack with ELB. Other possible factors to consider are landscape level diversity and heterogeneity, topography, hydrology, and management history. While there is now some information on ELB and its relationship with tamarack stand structure and composition available, there is a lack of experimental study on this, which may be needed to further elucidate these relationships.

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Chapter 5: Conclusion

Novel interactions between forests, climate, and both invasive and native insects are occurring and are to be expected in the future. The goal of this research was to provide valuable insights for forest health management across the Great Lakes region in this context. Observations of detection, disturbance dynamics, and management interventions for some forest insects in lowland systems of this region were made.

Our first objective was to identify potential replacement species for black ash (*Fraxinus nigra* Marsh.), threatened by the invasive emerald ash borer, to plant in northern black ash wetlands that have adequate cold and flood tolerance for that system. Through a greenhouse and nursery outplanting experiment, I was able to quantify survival and growth for eighteen species over multiple years at sites of varying climates to provide insights about the interaction of flooding and climate. Some species that showed promise were American elm (*Ulmus americana* L.), river birch (*Betula nigra* L.), and swamp white oak (*Quercus bicolor* Willd), however, among these and all species, there were often tradeoffs between growth and survival, meaning there is no one singular best choice. Nursery management and experimental design could have also contributed to the variation we were seeing in survival and growth.

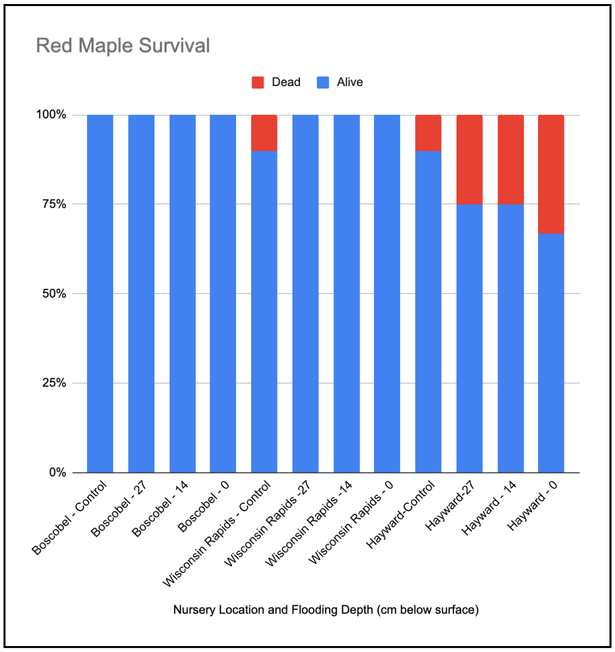
The second objective was to explore methods of detection for the native eastern larch beetle by ground truthing areas detected as disturbed. This was necessitated by an unprecedented outbreak occurring since 2001 in northern Minnesota. Accurate methods of detecting populations of this beetle are needed to monitor the current extent and locate future impacts. Observations of aerial sketch-map surveys and a remote sensing algorithm using satellite imagery show that these methods may assist in identifying potential areas of ELB disturbance, but they may not be sufficient on their own. Using a combination of detection methods like aerial surveys, satellite imagery, inventory data, or ground surveys could offer improvements.

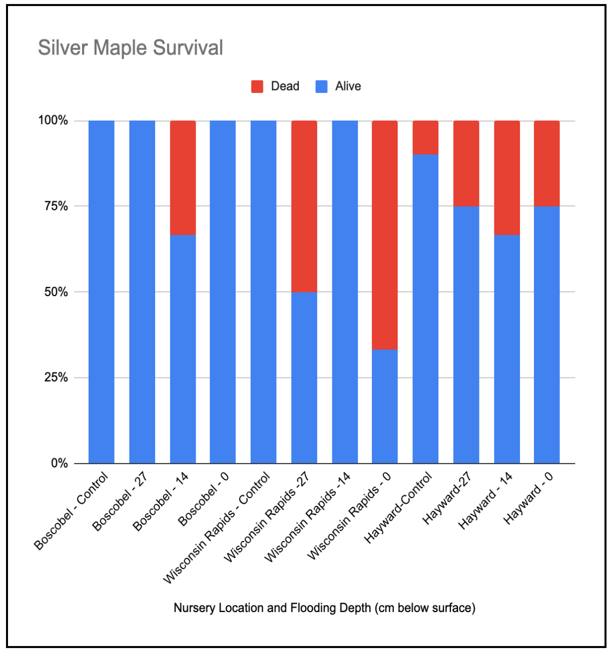
The last objective was to identify relationships between ELB and tamarack mortality, regeneration, structure, and composition in north central Wisconsin. Inventory data was collected on overstory trees, regeneration layer, ground cover, ELB presence, invasive species, and canopy coverage in fixed radius plots and analyzed at the plot level. ELB here was found to be relatively common across the landscape, and had a high level of infestation in the plots where it was found. Significant negative relationships were identified between ELB and large tree density and tamarack recruitment, and a positive relationship was found for percent host species composition. No relationships were found between ELB and tamarack regeneration or diversity. This information may help guide management efforts across the Great Lakes region.

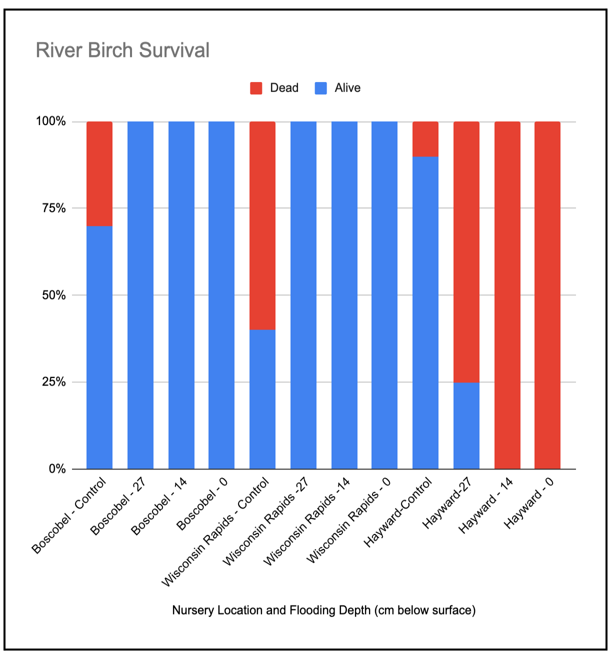
These observations represent a range of steps involved in the management of forest health. The first being detection of a species, understanding the dynamics or ecology of the species and system, and implementing action or treatments within a system to increase resilience. These steps are necessary for insects of both native and invasive origins, as changing dynamics present new impacts and challenges. While these insights provide specific information on a certain pest and forest ecosystem, they have wider implications for the management of forest insects.

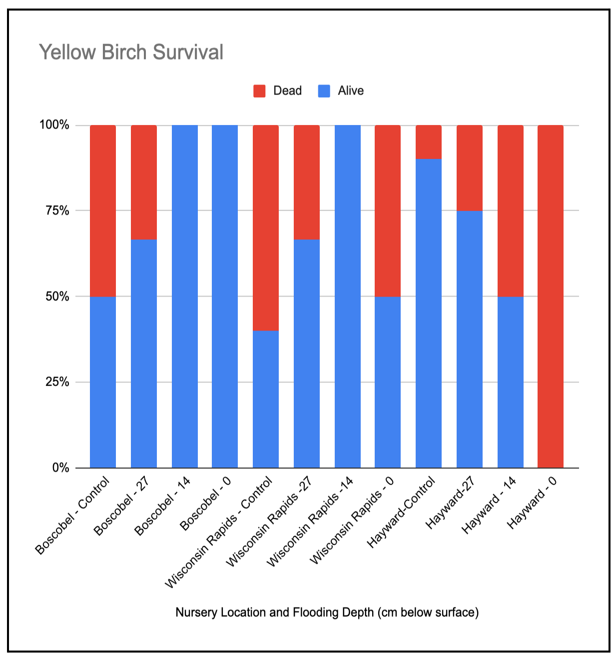
Appendix 2A

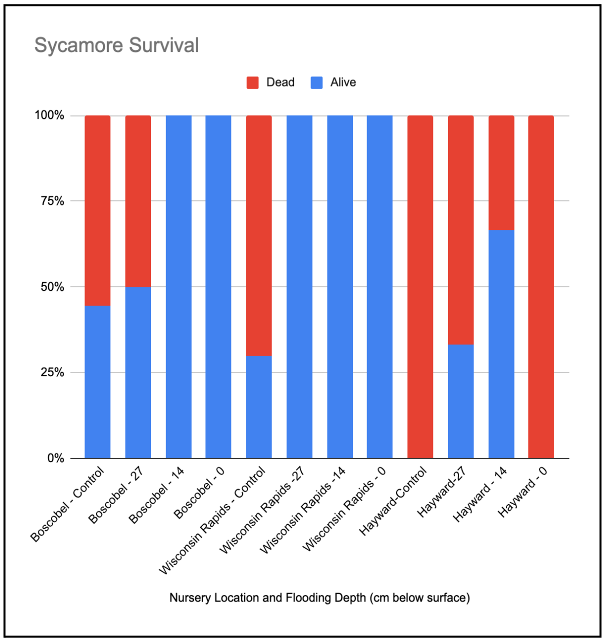
Percent survival of each species according to nursery and treatment.

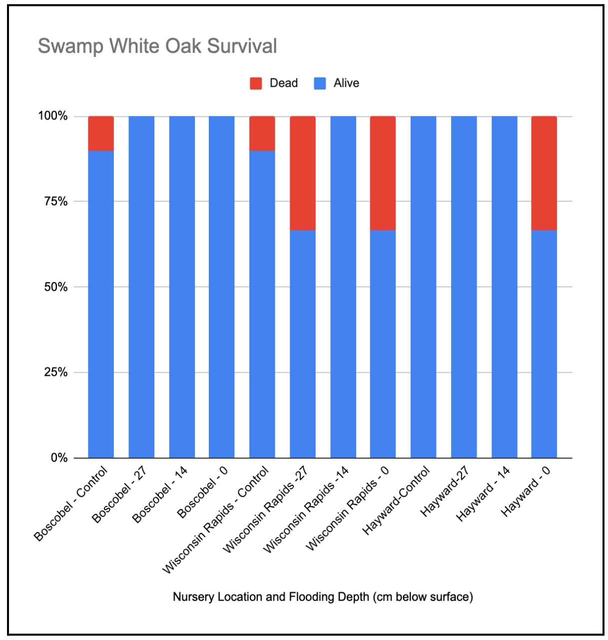


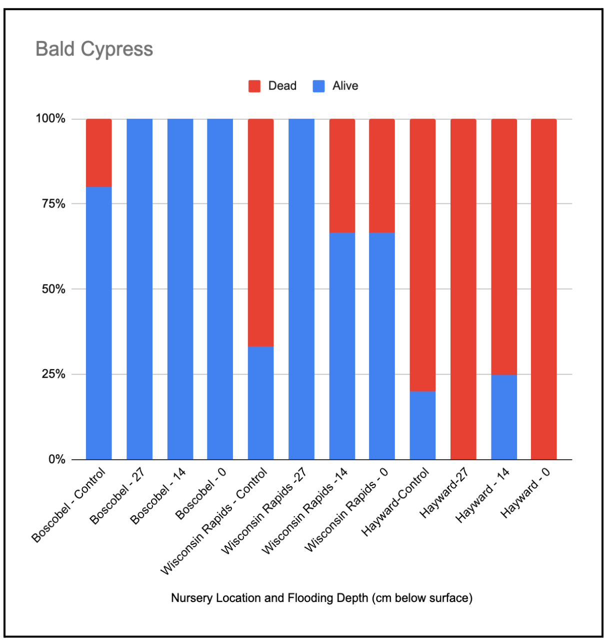


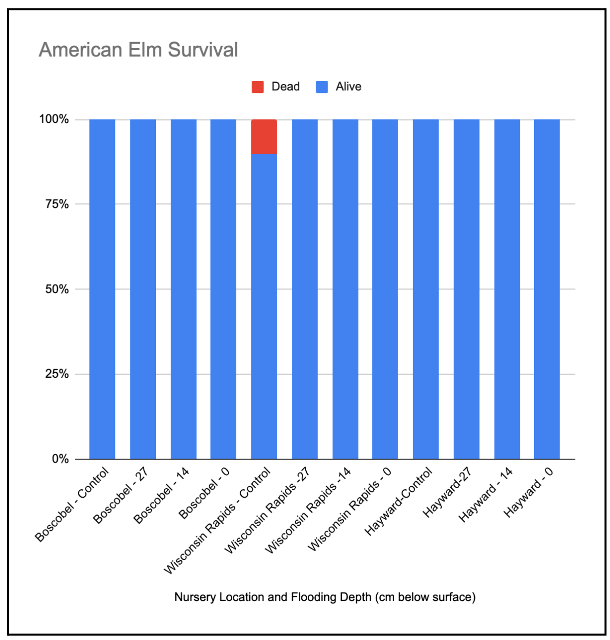





Sycamore Survival
Dead
Alive
100%
75%
50%
25%
0%
Boscobel - Control
Boscobel - 27
Boscobel - 14
Boscobel - 0
Wisconsin Rapids - Control
Wisconsin Rapids -27
Wisconsin Rapids -14
Wisconsin Rapids - 0
Hayward-Control
Hayward-27
Hayward - 14
Hayward - 0
Nursery Location and Flooding Depth (cm below surface)

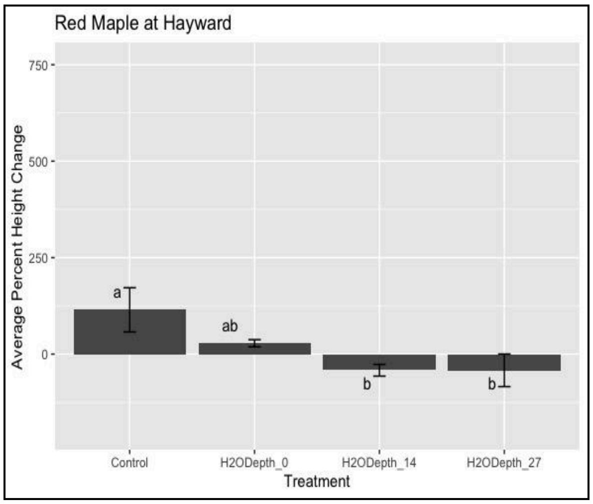

Swamp White Oak Survival
Dead
Alive
100%
75%
50%
25%
0%
Boscobel - Control
Boscobel - 27
Boscobel - 14
Boscobel - 0
Wisconsin Rapids - Control
Wisconsin Rapids -27
Wisconsin Rapids -14
Wisconsin Rapids - 0
Hayward-Control
Hayward-27
Hayward - 14
Hayward - 0
Nursery Location and Flooding Depth (cm below surface)

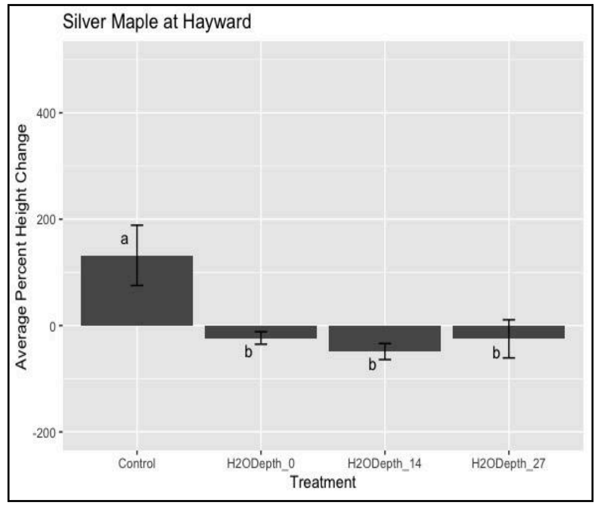

Bald Cypress
Dead
Alive
100%
75%
50%
25%
0%
Boscobel - Control
Boscobel - 27
Boscobel - 14
Boscobel - 0
Wisconsin Rapids - Control
Wisconsin Rapids -27
Wisconsin Rapids -14
Wisconsin Rapids - 0
Hayward-Control
Hayward-27
Hayward - 14
Hayward - 0
Nursery Location and Flooding Depth (cm below surface)

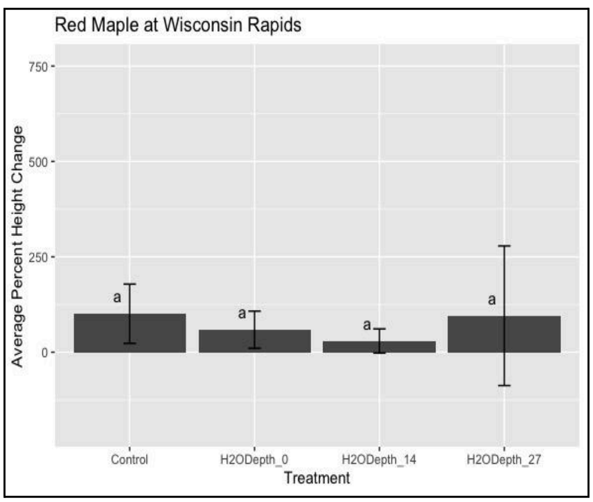

American Elm Survival
Dead
Alive
100%
75%
50%
25%
0%
Boscobel - Control
Boscobel - 27
Boscobel - 14
Boscobel - 0
Wisconsin Rapids - Control
Wisconsin Rapids -27
Wisconsin Rapids -14
Wisconsin Rapids - 0
Hayward-Control
Hayward-27
Hayward - 14
Hayward - 0
Nursery Location and Flooding Depth (cm below surface)

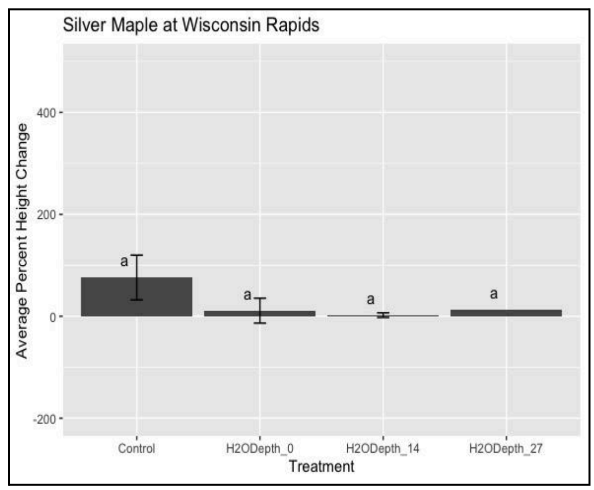
Appendix 2B

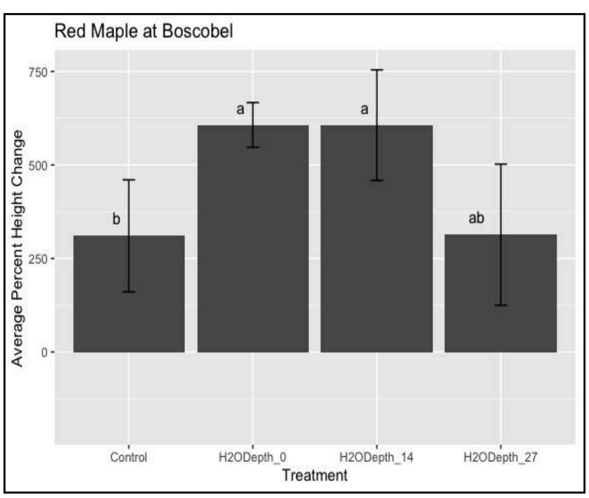
Average percent height change of each species according to nursery and treatment.

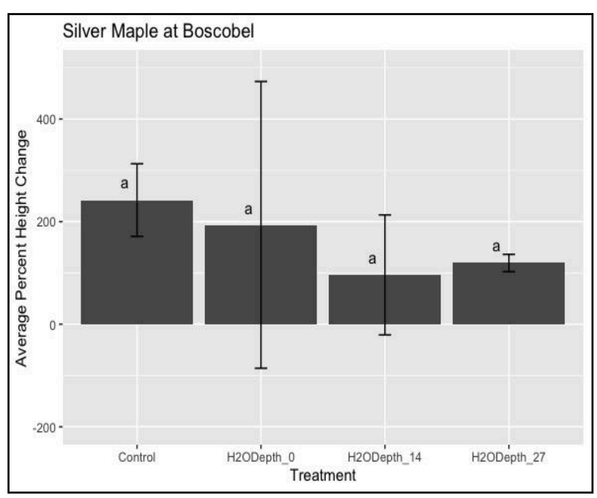


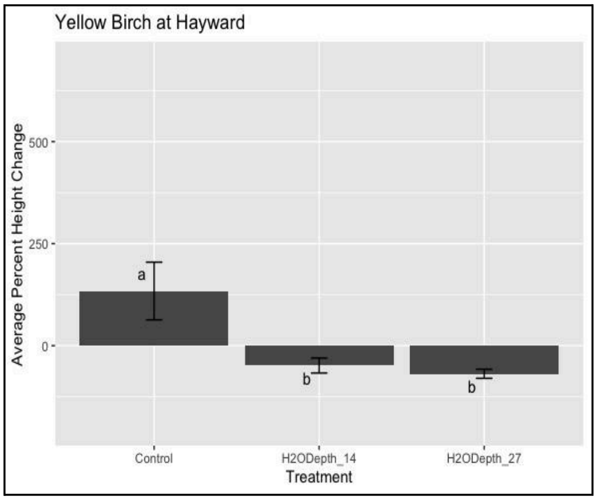


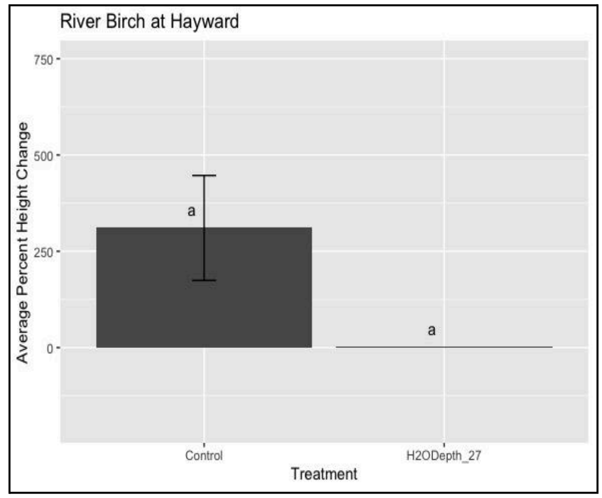


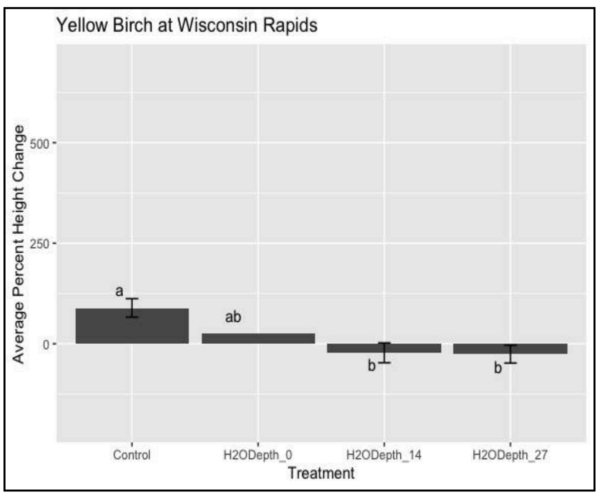


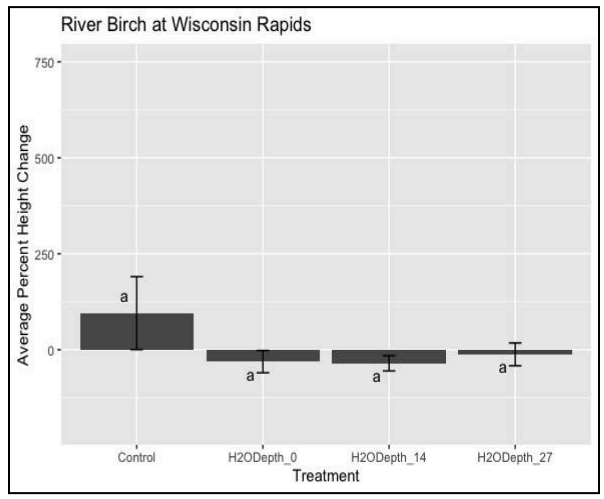


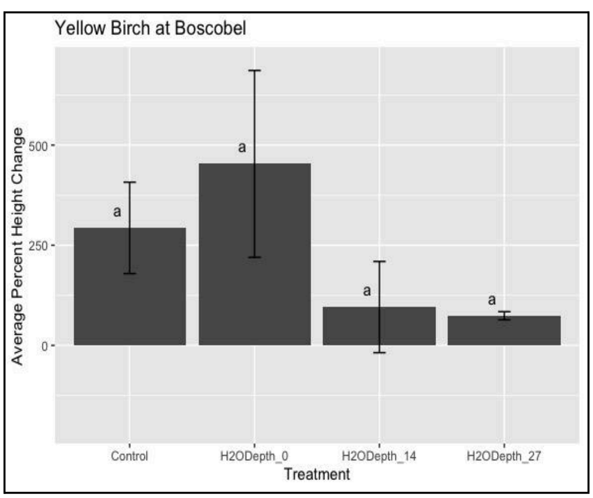


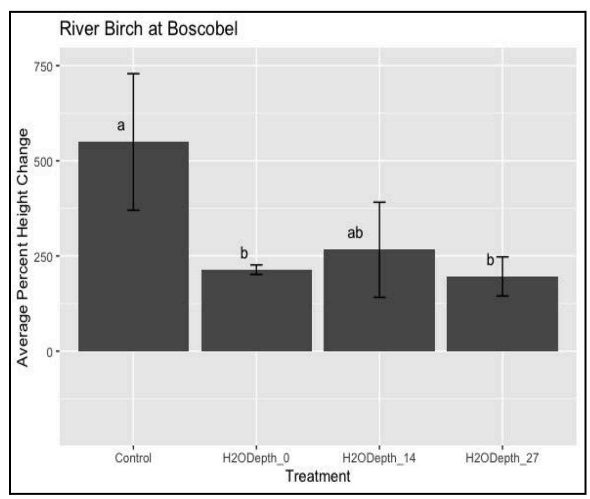

Yellow Birch at Hayward
Average Percent Height Change
500
250
0
a
b
b
Control
H2ODepth_14
H2ODepth_27
Treatment

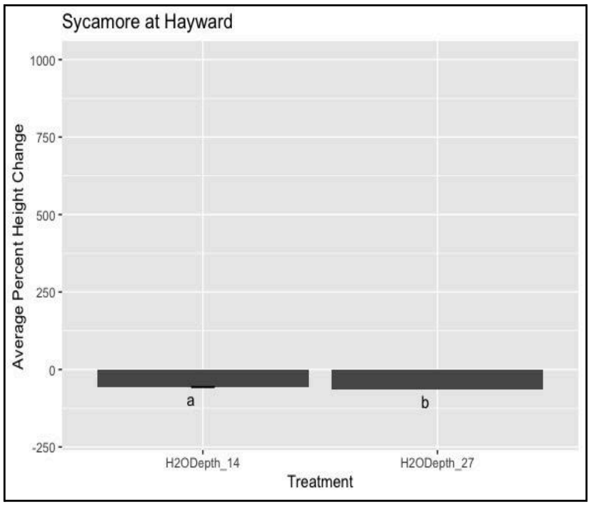

River Birch at Hayward
Average Percent Height Change
750
500
250
0
a
a
Control
H2ODepth_27
Treatment

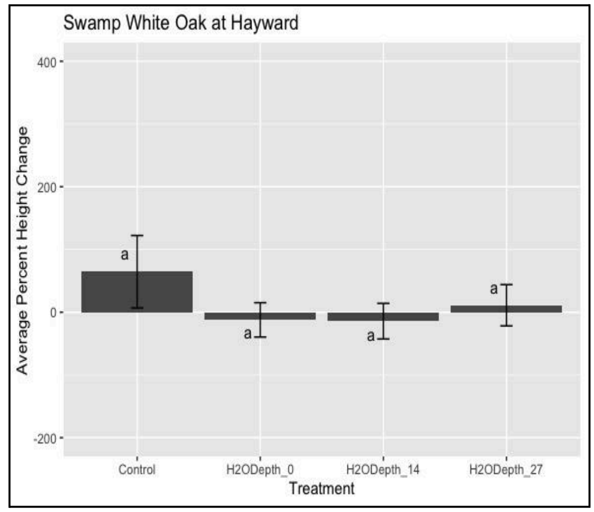

Yellow Birch at Wisconsin Rapids
Average Percent Height Change
500
250
0
a
ab
b
b
Control
H2ODepth_0
H2ODepth_14
H2ODepth_27
Treatment

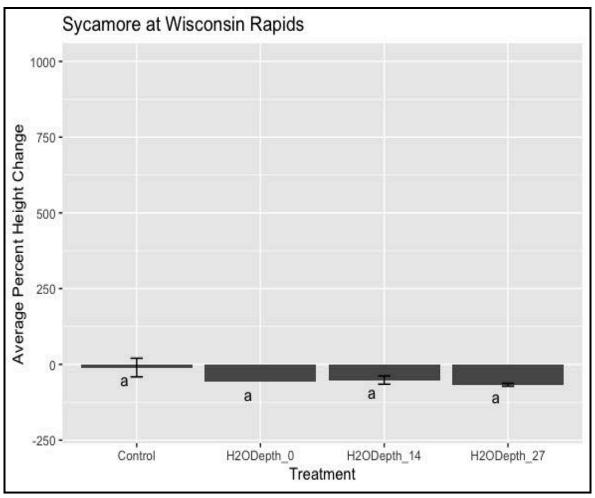

River Birch at Wisconsin Rapids
Average Percent Height Change
750
500
250
0
a
a
a
a
Control
H2ODepth_0
H2ODepth_14
H2ODepth_27
Treatment

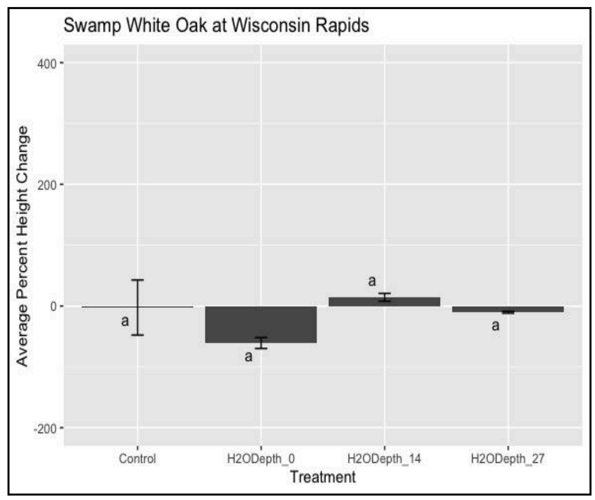

Yellow Birch at Boscobel
Average Percent Height Change
500
250
0
a
a
a
a
Control
H2ODepth_0
H2ODepth_14
H2ODepth_27
Treatment

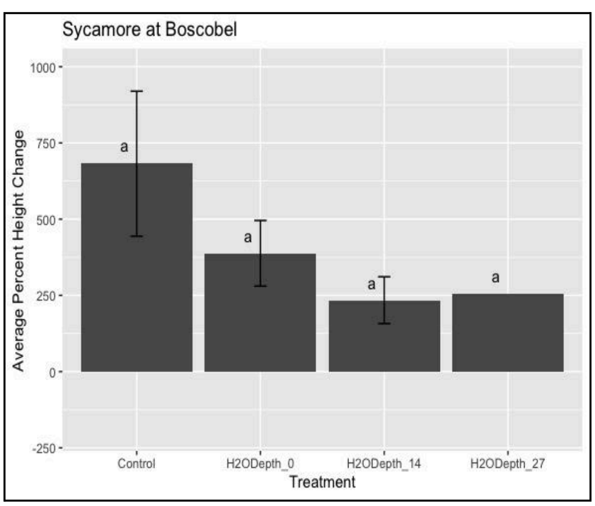

River Birch at Boscobel
Average Percent Height Change
750
500
250
0
a
b
ab
b
Control
H2ODepth_0
H2ODepth_14
H2ODepth_27
Treatment

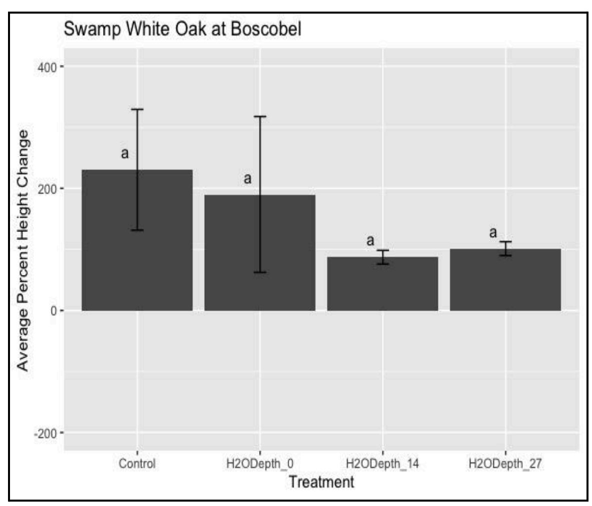

Sycamore at Hayward
1000
750
500
250
0
-250
Average Percent Height Change
a
b
H2ODepth_14
H2ODepth_27
Treatment


Swamp White Oak at Hayward
400
200
0
-200
Average Percent Height Change
a
a
a
a
Control
H2ODepth_0
H2ODepth_14
H2ODepth_27
Treatment


Sycamore at Wisconsin Rapids
1000
750
500
250
0
-250
Average Percent Height Change
a
a
a
a
Control
H2ODepth_0
H2ODepth_14
H2ODepth_27
Treatment


Swamp White Oak at Wisconsin Rapids
400
200
0
-200
Average Percent Height Change
a
a
a
a
Control
H2ODepth_0
H2ODepth_14
H2ODepth_27
Treatment


Sycamore at Boscobel
1000
750
500
250
0
-250
Average Percent Height Change
a
a
a
a
Control
H2ODepth_0
H2ODepth_14
H2ODepth_27
Treatment


Swamp White Oak at Boscobel
400
200
0
-200
Average Percent Height Change
a
a
a
a
Control
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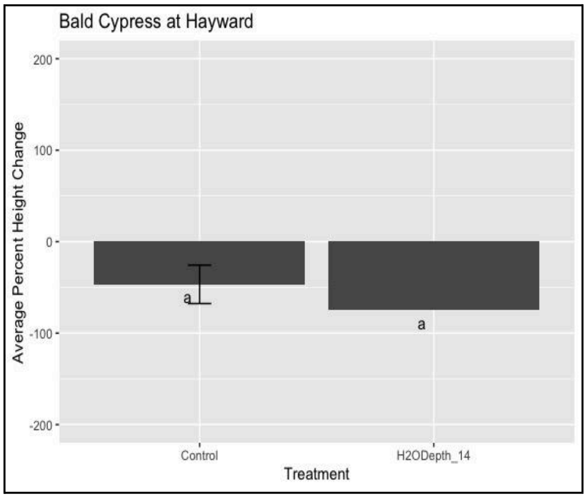
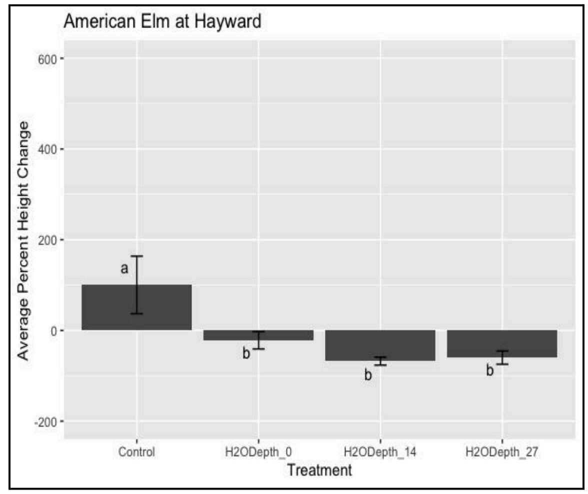
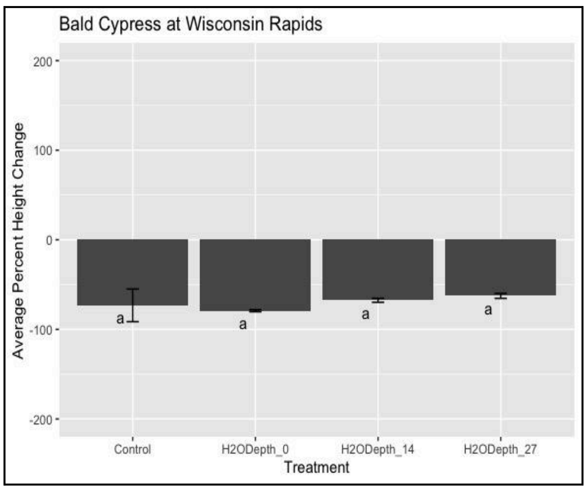
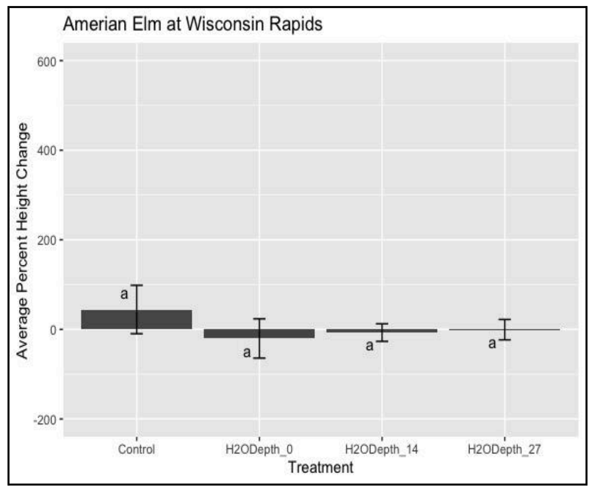
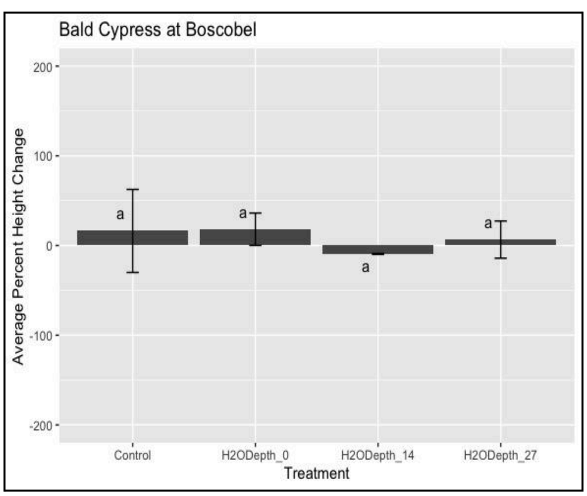
H2ODepth_14
H2ODepth_27
Treatment

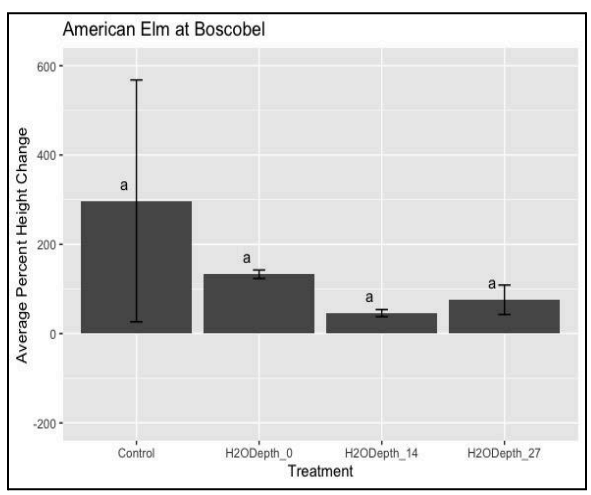

Bald Cypress at Hayward
Average Percent Height Change
200
100
0
-100
-200
a
a
Control
H2ODepth_14
Treatment


American Elm at Hayward
Average Percent Height Change
600
400
200
0
-200
a
b
b
b
Control
H2ODepth_0
H2ODepth_14
H2ODepth_27
Treatment


Bald Cypress at Wisconsin Rapids
Average Percent Height Change
200
100
0
-100
-200
a
a
a
a
Control
H2ODepth_0
H2ODepth_14
H2ODepth_27
Treatment


Amerian Elm at Wisconsin Rapids
Average Percent Height Change
600
400
200
0
-200
a
a
a
a
Control
H2ODepth_0
H2ODepth_14
H2ODepth_27
Treatment


Bald Cypress at Boscobel
Average Percent Height Change
200
100
0
-100
-200
a
a
a
a
Control
H2ODepth_0
H2ODepth_14
H2ODepth_27
Treatment


American Elm at Boscobel
Average Percent Height Change
600
400
200
0
-200
a
a
a
a
Control
H2ODepth_0
H2ODepth_14
H2ODepth_27
Treatment

Appendix 3A

List of stands and polygons visited for survey

Stand Name	Type of Survey/s	Size (Acres)	Number of Plots	Latitude	Longitude
SiAP10	20 Aerial Validation Polygons	24.6	10	45.65661	-90.00166
SiAP17	20 Aerial Validation Polygons	7.2	8	45.65618	-90.02996
SiAP18	20 Aerial Validation Polygons	6.7	7	45.65946	-90.03577
SiAP2	20 Aerial Validation Polygons	21.3	8	45.59216	-89.91056
SiAP21	20 Aerial Validation Polygons	20.9	9	45.6566	-90.02014
SiAP22	20 Aerial Validation Polygons	12.8	5	45.97921	-89.15455
SiAP23	20 Aerial Validation Polygons	46.8	13	45.98435	-89.25676
SiAP26	20 Aerial Validation Polygons	34.6	9	45.65039	-89.82409
SiAP28	20 Aerial Validation Polygons	13.6	5	45.6733	-89.76015
SiAP34	20 Aerial Validation Polygons	60.0	14	45.64604	-90.06407
SiAP35	20 Aerial Validation Polygons	20.9	7	45.58011	-89.59043
SiAP36	20 Aerial Validation Polygons	9.2	10	45.58539	-89.38635
SiAP37	20 Aerial Validation Polygons	10.0	9	45.58699	-89.58511
SiAP43	20 Aerial Validation Polygons	9.5	10	46.07249	-89.86961
SiAP47	20 Aerial Validation Polygons	7.9	8	45.64847	-90.07276
SiAP52	20 Aerial Validation Polygons	5.7	5	45.63961	-89.99187
SiAP60	20 Aerial Validation Polygons	10.9	5	45.57733	-89.59107
SiAP64	20 Aerial Validation Polygons	29.6	10	46.06972	-89.86846
SiAP65	20 Aerial Validation Polygons	7.7	6	45.9968	-89.25264
SiAP7	20 Aerial Validation Polygons	32.9	8	45.94693	-89.53248
WS1057	20 RS Validation Stands	32.2	9	45.81041	-89.54889
WS1146	20 RS Validation Stands	8.8	8	45.68308	-89.39776
WS1165	20 RS Validation Stands	5.0	7	45.57799	-89.91926
WS1188	20 RS Validation Stands	29.4	10	45.59423	-89.9901
WS1189	20 RS Validation Stands	7.8	7	45.56941	-90.00562
WS1331	20 RS Validation Stands	10.8	5	45.70064	-89.80971
WS1436	20 RS Validation Stands	7.8	7	45.63145	-90.1061
WS1719	20 RS Validation Stands	58.9	14	46.04578	-89.4869
WS1736	20 RS Validation Stands	15.5	6	46.04394	-89.50726
WS1737	20 RS Validation Stands	5.7	6	46.04271	-89.50153
WS1923	20 RS Validation Stands	50.2	12	46.08065	-89.91962
WS1962	20 RS Validation Stands	8.7	9	45.97447	-89.30055
WS2024	20 RS Validation Stands	5.1	5	46.01851	-89.28724
WS2039	20 RS Validation Stands	6.6	5	46.06749	-89.35465
WS899	20 RS Validation Stands	5.1	5	45.54062	-89.44025
WS796	20 RS Validation Stands	7.1	8	45.46196	-89.51704
WS1145	20 RS Validation Stands	17.1	6	45.6855	-89.59721
WS572	20 RS Validation Stands	18.0	8	45.46317	-89.41634
WS799	20 RS Validation Stands	12.3	6	45.45932	-89.4664
WS895	20 RS Validation Stands	10.2	5	45.52011	-89.47391
WS1078	10 Random WisFIR	8.4	6	45.84968	-89.43768
WS1163	10 Random WisFIR	20.2	8	45.5902	-89.89089
WS1252	10 Random WisFIR	5.4	5	45.6686	-89.65622
WS1271	10 Random WisFIR	13.3	5	45.81079	-89.22744
WS1308	10 Random WisFIR	7.5	7	45.77477	-89.8696
WS1333	10 Random WisFIR	35.2	13	45.63871	89.68118
WS1826	10 Random WisFIR	10.3	5	46.09509	-89.65472
WS1913	10 Random WisFIR	28.2	17	46.0599	-89.59925
WS798	10 Random WisFIR	17.2	7	45.46545	-89.47842
WS979	10 Random WisFIR	12.2	6	45.52885	-89.77747

Appendix 3B

Field data sheet & summarized protocols for answering each question

TAMARACK/EASTERN LARCH BEETLE & FOREST REGENERATION PROJECT: UMN and WDNR: Validation Data sheet scanned Y / N   Data sheet entered Y / N

Stand or Polygon ID: ________________ **Recon ID(If applicable):** ___________ **Plot:** ________________
Lat/Long: ____ **Date:** _________ **Crew:** ________________________
Survey Type(s): ____
Is the centroid in a polygon of mapped disturbance Y / N ? (Circle one) IF Y, then is it Aerially Detected or Remote Sensing Detected? (Circle all that apply)

GPS Point taken?	Photos taken?
□ Yes!	□ Yes!

Please see protocol for expanded details on how to answer each of these questions

1. Is tamarack present in the overstory Y / N
   a. If no, which tree species are present ____________________
      i. Circle/list up to 3: PIMA, POPULUS*, THOC, ACRU, BEAL, PIRE, PIST, Other ______________
2. Is tamarack component in the overstory over 50% Y / N
   a. If no, which tree species is the most abundant
      i. Circle up to 3: PIMA, POPULUS*, THOC, ACRU, BEAL, PIRE, PIST, Other_______________
3. Is there dead tamarack in the overstory Y / N
   a. If yes, what percent of tamarack are dead
      i. 0-25%
      ii. 25-50%
      iii. 50-75%
      iv. 75-100%
   b. If yes or no, is the first representative (Dead / Alive) tamarack infested with ELB:
      i. None/low
      ii. Moderately
      iii. Highly
   c. If no, are there other dead species in the overstory Y / N
      i. If yes, what percent of trees are dead in the overstory
         1. 0-25%
         2. 25-50%
         3. 50-75%
         4. 75-100%
4. Are there signs (holes, sometimes accompanied by frass and galleries) of eastern larch beetle Y / N
5. Are there invasive species present Y / N
   a. Woody shrubs (circle all that apply)
      i. buckthorn
      ii. honeysuckle
      iii. other__________
   b. Ground Layer
      i. Y / N
6. Is there standing water within the plot (exclude water bodies from question 7)
   a. Y / N
7. Is there a water body (Stream, pond, lake) within visual sight.
   a. Y / N

*All populus species.

Appendix 3C

Table 3.2 Stand summarization of survey values showing percent of plots that fall within each category for the twenty Aerial Sketch Map Survey Polygons.

	Tamarack Present	Tamarack Dominant	Dead Tamarack	Signs of ELB	Standing Water	Water Body	Tamarack Percent Mortality			
Stand							<25	25 - <50	50 - <75	≥ 75
SiAP43	100	90	90	90	0	10	0	20.00	50.00	20.00
SiAP47	100	87.5	75	75	0	0	0	0	12.50	62.50
SiAP2	100	50	12.5	12.5	0	0	12.5	0	0	0
SiAP21	100	33.33	55.56	44	0	0	33.33	11.11	11.11	0
SiAP18	100	14.29	14.29	0	0	0	14.29	0	0	0
SiAP17	100	12.5	0	0	0	0	0	0	0	0
SiAP64	90	80	30	20	0	0	20.00	0	0	10.00
SiAP36	90	50	60	70	10	0	50.00	10.00	0	0
SiAP10	90	20	40	20	0	0	10	20	10	0
SiAP26	88.89	0	0	0	11.11	11.11	0	0	0	0
SiAP52	80	80	40	20	0	0	0	0	0	40.00
SiAP34	85.71	64.29	78.57	71.43	78.57	0	7.14	14.29	0	57.14
SiAP37	77.78	66.67	44.44	33.33	0	0	44.44	0	0	0
SiAP22	60	60	40	40	40	60	0	0	0	40
SiAP23	46.15	46.15	38.46	38.46	15.38	0	7.69	7.69	7.69	15.38
SiAP7	25	0	0	0	12.5	0	0	0	0	0
SiAP60	20	0	0	0	0	0	0	0	0	0
SiAP65	16.67	0	0	0	16.67	0	0	0	0	0
SiAP28	0	0	0	0	0	20	0	0	0	0
SiAP35	0	0	0	0	0	0	0	0	0	0

Table 3.3 Stand summarization of survey values showing percent of plots that fall within each category for the ten WisFIRS tamarack stands

	Tamarack Present	Tamarack Dominant	Dead Tamarack	ELB Signs	Standing Water	Water Body	Tamarack Percent Mortality			
Stand							<25	25 - <50	50 - <75	≥ 75
WS1271	100	100	60	60	0	0	20	20	0	20
WS1333	100	76.92	53.85	46.15	7.69	0	46.15	0	7.69	0
WS1163	100	75	50	37.5	0	0	25	12.5	12.5	0
WS1308	100	42.86	42.86	42.86	14.29	0	14.29	14.29	0	14.29
WS1913	88.24	88.24	17.65	11.76	0	11.76	17.65	0	0	0
WS1078	83.33	33.33	0	16.67	33.33	0	0	0	0	0
WS1826	80	60	80	80	0	20	20	20	0	40
WS798	71.43	14.29	0	0	0	42.86	0	0	0	0
WS1252	60	0	0	0	0	20	0	0	0	0
WS979	50	33.33	33.33	33.33	0	0	16.67	16.67	0	0

Table 3.4 Plot summarization of survey values showing percent of plots that fall within each category for plots in Astrape (remotely sensed) stands for detected (88) and undetected (60) plots. Two plots were missing Tamarack Percent Mortality data.

Plot Status	Tamarack Present	Tamarack Dominant	Dead Tamarack	ELB Signs	Standing Water	Water Body	Tamarack Percent Mortality			
							< 25	25 - < 50	50 - < 75	≥ 75
Detected	80.68	53.41	31.82	29.55	14.77	13.64	10.23	6.82	3.41	10.23
Undetected	71.67	51.67	21.67	13.33	13.33	10	13.33	1.67	0	5

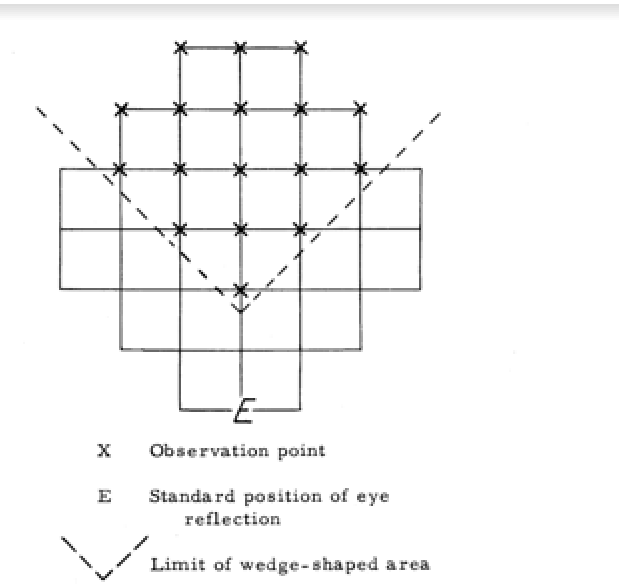
Appendix 4A

List of stands and polygons visited for survey

Stand Name	Type of Survey/s	Size (Acres)	Number of Plots	Latitude	Longitude
WS1078	10 Random WisFIR	8.4	5	45.84968	-89.43768
WS1163	10 Random WisFIR	20.2	8	45.5902	-89.89089
WS1252	10 Random WisFIR	5.4	5	45.6686	-89.65622
WS1271	10 Random WisFIR	13.3	5	45.81079	-89.22744
WS1308	10 Random WisFIR	7.5	6	45.77477	-89.8696
WS1333	10 Random WisFIR	35.2	9	45.63871	89.68118
WS1826	10 Random WisFIR	10.3	5	46.09509	-89.65472
WS1913	10 Random WisFIR	28.2	12	46.0599	-89.59925
WS798	10 Random WisFIR	17.2	7	45.46545	-89.47842
WS979	10 Random WisFIR	12.2	5	45.52885	-89.77747
WS1225	5 Aerial Healthy	17.6	7	45.5257	-89.37994
WS1309	5 Aerial Healthy	23.2	9	45.67235	-89.88264
WS1777	5 Aerial Healthy	7.2	5	46.08037	-89.61291
WS1902	5 Aerial Healthy	28.5	11	46.03458	-89.63409
WS796	5 Aerial Healthy, 5 RS Unhealthy	7.1	6	45.46196	-89.51704
WS1925	5 Aerial Unhealthy	21.8	8	46.07122	-89.86964
WS2002	5 Aerial Unhealthy	13.5	5	45.98345	-89.25514
WS889	5 Aerial Unhealthy	47.1	12	45.53815	-89.8022
WS890	5 Aerial Unhealthy	34.6	8	45.53858	-89.81205
WS978	5 Aerial Unhealthy	10.2	5	45.53893	-89.39305
WS1074	5 RS Healthy Stands	33.0	7	45.85349	-89.335181
WS1144	5 RS Healthy Stands	13.0	6	45.65573	-89.55574
WS1183	5 RS Healthy Stands	19.4	8	45.65054	-89.95724
WS1234	5 RS Healthy Stands	16.3	5	45.57446	-89.97298
WS1275	5 RS Healthy Stands	14.3	6	45.8066	-89.22449
WS1145	5 RS Unhealthy Stands	17.1	6	45.6855	-89.59721
WS572	5 RS Unhealthy Stands	18.0	7	45.46317	-89.41634
WS799	5 RS Unhealthy Stands	12.3	5	45.45932	-89.4664
WS895	5 RS Unhealthy Stands	10.2	5	45.52011	-89.47391

Appendix 4B

Figure demonstrating Strickler method



Appendix 4C

Descriptions to determine the crown class of trees.

Symbol	Crown class	Definition
O	Open grown	Tree crowns receive full light from above and from all sides. In even-aged stands, these trees have their crowns well above the general canopy.

D	Dominant	Crowns extend above the general level of the canopy. They receive full light from above and some light from the sides
C	Codominant	Crowns make up the general level of the canopy. They receive direct light from above, but little or no light from the sides. Generally shorter than the dominant trees.
I	Intermediate	Crowns occupy a subordinate position in the canopy. They receive some direct light from above, but no direct light from the sides. Crowns are generally narrow and/or one-sided, and shorter than the dominant and codominant trees.
S	Suppressed	Crowns are below the general level of the canopy. They receive no direct light. Crowns are generally short, sparse, and narrow.