

SUNY College of Environmental Science and Forestry

Digital Commons @ ESF

Dissertations and Theses

8-15-2019

HABITAT SELECTION, PREY PREFERENCE, AND POPULATION ECOLOGY OF NORTHERN BARRENS TIGER BEETLES IN THE HUDSON VALLEY, NEW YORK

McKenzie Wybron
mckenziwybron@gmail.com

Follow this and additional works at: <https://digitalcommons.esf.edu/etds>



Part of the [Ecology and Evolutionary Biology Commons](#), and the [Environmental Monitoring Commons](#)

Recommended Citation

Wybron, McKenzie, "HABITAT SELECTION, PREY PREFERENCE, AND POPULATION ECOLOGY OF NORTHERN BARRENS TIGER BEETLES IN THE HUDSON VALLEY, NEW YORK" (2019). *Dissertations and Theses*. 115.

<https://digitalcommons.esf.edu/etds/115>

This Open Access Thesis is brought to you for free and open access by Digital Commons @ ESF. It has been accepted for inclusion in Dissertations and Theses by an authorized administrator of Digital Commons @ ESF. For more information, please contact digitalcommons@esf.edu, cjkoons@esf.edu.

**HABITAT SELECTION, PREY PREFERENCE, AND POPULATION ECOLOGY OF
NORTHERN BARRENS TIGER BEETLES IN THE HUDSON VALLEY, NEW YORK**

by

McKenzie Wybron

A thesis
submitted in partial fulfillment
of the requirements for the
Master of Science Degree State University of New York
College of Environmental Science and Forestry
Syracuse, New York
December 2019

Department of Environmental and Forest Biology

Approved by:
Melissa Fierke, co-Major Professor/ Department chair
Dylan Parry, co-Major Professor
Matthew Schlesinger, Steering Committee
Timothy Morin, Examining Committee Chair
S. Scott Shannon, Dean, The Graduate School

© 2019

Copyright

M. D. Wybron

All rights reserved

ACKNOWLEDGEMENTS

First, I thank Matthew Schlesinger for bringing me onto this project and always being so willing to answer every question I came up with. His willingness to let me develop my own methods helped me learn a lot about the importance of patience and paying attention to detail. I appreciated that I could always count on him to help me through any difficult situation I ran into.

I also couldn't have gotten through this without my co-major professors, Melissa Fierke and Dylan Parry. I thank them for taking me on as a student and supporting me throughout my Master's. Whenever I needed a question answer, I could always count on them to help me and appreciated their openness to help even if it was out of their expertise. I appreciated all of their revisions and comments to my conference presentations which helped me become a better scientific presenter. I also would like to thank Jacqueline Frair, who helped me with all my mark-recapture and occupancy modeling. Her willingness to teach me helped me develop a strong understanding of rare species monitoring using these methods.

I am also thankful for all the people who spent their time helping me in the field and in the lab. I appreciated all the help during my third field season from my field technician, Christian Chevalier. His enthusiasm and dedication for this project made for a very successful field season.

I also appreciated all the folks from the Natural Heritage Program who spent some days marking and capturing beetles with me and walking me through any technical difficulty I ran into. I am especially grateful for my Fierke lab mates who supported me throughout my entire Master's

Finally, this research was funded through the New York State Office of Parks, Resources, and Historic Preservation and managed by the New York Natural Heritage Program and funded by the Edna Bailey Sussman Foundation.

TABLE OF CONTENTS

LIST OF TABLES.....	vi
LIST OF FIGURES.....	vii
Abstract.....	viii
CHAPTER 1: LITERATURE REVIEW.....	1
INTRODUCTION.....	2
INSECT DISPERSAL.....	3
MARK-RECAPTURE AND OCCUPANCY.....	4
TIGER BEETLE BIOLOGY.....	6
NORTHERN BARRENS TIGER BEETLE.....	10
FIRE-DEPENDENT HABITAT.....	12
RESEARCH OBJECTIVES.....	14
CHAPTER 2: ESTIMATING ABUNDANCE AND OCCUPANCY OF NORTHERN BARRENS TIGER BEETLE IN AN ENDEMIC NEW YORK POPULATION....	15
ABSTRACT.....	16
INTRODUCTION.....	17
METHODS.....	19
<i>Study area</i>	19
<i>Mark-recapture</i>	19
<i>Population size estimates</i>	22
<i>Occupancy</i>	22
RESULTS.....	23
<i>Mark-recapture</i>	23
<i>Occupancy</i>	27
DISCUSSION.....	28
CONCLUSION.....	31
CHAPTER 3: HABITAT AND PREY PREFERENCE OF NORTHERN BARRENS TIGER BEETLES IN THE HUDSON VALLEY, NEW YORK.....	32
ABSTRACT.....	33

INTRODUCTION.....	34
METHODS.....	36
<i>Study area</i>	36
<i>Habitat survey</i>	38
<i>Prey abundance</i>	38
<i>Statistical analysis</i>	39
RESULTS.....	43
DISCUSSION.....	49
CONCLUSION.....	53
CHAPTER 4: CONCLUSIONS AND FURTHER RESEARCH.....	54
CONCLUSIONS.....	55
LIMITATIONS AND FUTURE DIRECTIONS.....	57
LITERATURE CITED.....	59
RESUME.....	66

LIST OF TABLES

Chapter 2

Table 2.1: Mark-recapture data by sex24

Table 2.2: Relative AIC_c values for mark recapture models.....25

Table 2.3: Relative AIC_c values for occupancy models.....27

Chapter 3

Table 3.1: Random Forest model errors.....42

LIST OF FIGURES

Chapter 2

Figure 2.1: Map of <i>C. patruela</i> studied and new populations in Sam’s Point Preserve.....	19
Figure 2.2: Marking techniques for adult <i>C. patruela</i>	21
Figure 2.3: Capture probabilities for <i>C. patruela</i>	26
Figure 2.4: Detection probabilities for <i>C. patruela</i>	28

Chapter 3

Figure 3.1: Map of <i>C. patruela</i> studied and new populations in Sam’s Point Preserve.....	37
Figure 3.2: Workflow model for habitat score map.....	39
Figure 3.3: Habitat score map for Minnewaska State Park.....	40
Figure 3.4: Variable importance plots.....	45
Figure 3.5: Partial dependence plot for Sam’s Point Preserve model.....	46
Figure 3.6: Partial dependence plot for historical range model.....	47
Figure 3.7: Partial dependence plot for larval burrow model.....	48
Figure 3.8: Habitat where multiple larval burrows found.....	50
Figure 3.9: <i>C. patruela</i> adult with prey item.....	52

ABSTRACT

M. D. Wybron. Habitat Selection, Prey Preference, and Population Ecology of Northern Barrens Tiger Beetle in the Hudson Valley, New York, 66 pages, 4 tables, 13 figures, 2019. Ecology style guide used.

Reliable monitoring of the population ecology of rare species and determination of important habitat variables are critical for determining extinction risk and for development of management plans. The goal of this research was to ascertain basic population ecology and define important habitat and prey variables for the rare northern barrens tiger beetle, *Cicindela patruela patruela* Dejean (Coleoptera: Carabidae), at Sam's Point Preserve, NY. Mark-recapture surveys spanning three years were conducted and habitat use and prey abundance assessed within a single year. Over a two-year period, the population grew from 81 (95% CI 75 to 99) to 109 (95% CI 101 to 128) individuals with an occupancy estimate of 0.79 ± 0.16 . Percent cover of pebbles, moss/lichen, and small sand grains were important variables associated with *C. patruela* presence. Priority should be given to identification of new populations outside the current known distribution in NY as well as further identification and preservation of critical habitat features.

Key Words: *Cicindela patruela*, Northern barrens tiger beetles, rare species, mark-recapture, occupancy, random forest, habitat, Sam's Point Preserve, monitoring

M.D.Wybron

Candidate for the degree of Master of Science, August 2019

Melissa Fierke, Ph.D.

Dylan Parry, Ph.D.

Department of Environmental and Forest Biology

State University of New York College of Environmental Science and Forestry

Syracuse, NY

CHAPTER 1

LITERATURE REVIEW

INTRODUCTION

Northern barrens tiger beetles, *Cicindela patruela patruela* Dejean (Coleoptera: Carabidae), are widely distributed throughout eastern North America pine-oak barren ecosystems, usually only occurring in small localized populations throughout their entire range (Mawdsley 2005). Populations of this beetle are at risk due to habitat destruction and modification, alteration to the frequency and intensity of disturbance, particularly fire, and over-exploitation by collectors (Schlesinger and Novak 2011).

Although the northern barrens tiger beetle historically was widely distributed, it has been extirpated from six known locations in New York and is currently found at only at a single site in the state (Schlesinger and Novak 2011). In 2004, this single population was discovered in a dwarf pitch pine barrens at Sam's Point Preserve on the Shawangunk Ridge, New York (Schlesinger and Novak 2011). Subsequent observations indicate there are actually two discrete populations, one on each side of Lake Maratanza along the northwest and southeast ridges in the preserve (NYNHP 2017). Presence-absence surveys conducted for northern barrens tiger beetles from 2006 to 2008 by The New York Natural Heritage Program indicated the occupied area within Sam's Point Preserve pine barrens was larger than previously estimated (Schlesinger and Novak 2011).

Northern barrens tiger beetles are most commonly associated with open sandy or rocky areas within forested habitat. These openings are used for behavioral thermoregulation, foraging, mating, and oviposition. Decades of fire suppression in the Shawangunk mountains has greatly reduced the extent of existing openings and has curtailed regeneration of pitch-pine forests (Schlesinger and Novak 2011, Beers et al. 2011). In the absence of wild fire or prescribed burns, fire intolerant vegetation is proliferating and encroaching into the barren openings northern

barrens tiger beetle rely on, thereby degrading suitable habitat for the beetles (Mawdsley 2005). Development and implementation of a management plan for northern barrens tiger beetle is necessary for promoting natural openings and native plant assemblages in order for this rare species to persist.

INSECT DISPERSAL

The population size, distribution, community structure, and habitat occupancy of organisms are strongly affected by dispersal patterns (Dieckmann et al. 1999, Ranius and Hedin 2001). Dispersal can be routine or through special movement. Routine movement is the result of daily activities, e.g., foraging or thermoregulation, while special movement is the total of individual displacements away from the natal site leading to colonization of a new area (Van Dyck and Baguette 2005). Dispersal is an inherently risky displacement for any species, as articulated by renowned coleopterist Carl Lindroth: “It can hardly be favorable to an individual insect to fly away from a locality where its life is not threatened, i.e. where feeding, reproduction and hibernation are still possible; the chance of dying before reproduction will generally be much greater in dispersing individuals than in individuals staying in the population, especially when the habitat is rather isolated.” (Den Boer 1990). However, these risks may be offset by negative fitness costs associated with remaining in a habitat with strong competition and resource limitation (Osborne et al. 2002).

Habitat fragmentation, dispersal, and the probability of extinction can function synergistically and should be considered when developing monitoring plans for rare species (Lindenmayer and Possingham 1996). Habitat fragmentation can be defined as a disruption to the spatial distribution of resources (Van Dyck and Matthysen 1999). Fragmented habitat patches are usually less crowded than connected habitat patches due to limitations on dispersal and

colonization (Gadgil 1971). Fragmentation of suitable habitat patches can impact insect dispersal (Hunter 2002). If suitable habitat patches are separated by larger patches of unsuitable habitat, it could make dispersal between suitable habitat patches difficult. Movement of individuals to habitat patches with greater connectivity results in uneven dispersion across the environment (Van Dyck and Matthysen 1999). Replacement of emigrants by immigrants is less likely in isolated fragments due to a lack of suitable resources and distance among suitable habitats. Lack of connectivity among scattered patches of suitable habitat can lead to an overall increase in dispersal loss, especially in species capable of only dispersing short distances (Lindenmayer and Possingham 1996, Dieckmann et al. 1999, Tscharntke et al. 2002). Corridors between patches of suitable habitat have been found to promote colonization of those patches (Hunter 2002).

To understand insect population dynamics, one can quantify insect dispersal using either direct or indirect methods (Osborne et al. 2002). Direct methods typically involve individual mark-release-recapture (MRR) experiments or identifying the path of an insect's movement. Indirect methods include mass mark-recapture, vertical looking radar, and genetic and isotope markers. Out of these methods, MRR is the most frequently used technique used for monitoring insect dispersal (Osborne et al. 2002). Any marking method can be used as long as it does not alter or interfere with the insect's ability to move, forage, and mate, and artificial markers, such as painting marks onto the insect, or natural markers, such as genotype and phenotype have been employed (Hagler and Jackson 2001).

MARK-RECAPTURE AND OCCUPANCY

Reliable and accurate monitoring of rare species abundance in space and time is critical to determine their extinction risk. Abundance is commonly used to characterize the state of a population at a given time (Mackenzie et al. 2004). Conservation managers utilize mark-

recapture methods to monitor species, but they are limited to species which can be marked efficiently and later recaptured with reasonable effort (Lettink and Armstrong 2003). Mark-recapture can estimate population parameters if assumptions are met. Mark-recapture methods can yield high quality population estimates, but are more labor intensive and generally more costly and time-consuming than other methods (Lettink and Armstrong 2003). Estimation of detection probability is a critical consideration prior to developing a management plan based solely on changes in abundance (Mackenzie et al. 2004). If the probability of observing an individual is low, it could result in inaccurate estimation of changes in abundance. Mark-recapture methods give better information and population estimates that are needed to develop a management plan for rare species (Haddad et al. 2008).

Unbiased precise estimates of population size are critical to determine rare species population viability (Haddad et al. 2008). Haddad et al. (2008) found mark-recapture resulted in the most demographic information of the federally endangered St. Francis' satyr butterfly (*Neonympha mitchellii francisci* Parshall & Kral). The ability to track dispersal with mark-recapture is important for determining viability of populations in fragmented habitats. They posited mark-recapture also produced low levels of variability and bias with high sampling frequencies. Haddad et al. (2008) suggested creating a hybrid approach using mark-recapture with another method due to costs of mark-recapture. By combining mark-recapture with other methods, e.g., transect counts, costs are reduced while still providing accurate population estimates.

Occupancy is often utilized in monitoring programs as a surrogate for abundance as it requires less effort (Mackenzie and Bailey 2004). At an appropriate scale, occupancy is positively correlated with abundance, i.e. occupancy increases with increasing abundance

(Gaston et al. 2000). Occupancy is similar to mark-recapture, except it doesn't require marking individuals. Instead, a species is recorded simply as detected or not detected. Care should be taken, however, as non-detection does not always mean a site is not occupied, i.e. a false negative can occur if the organism is there, but was not detected during that survey period. It is often difficult to estimate abundance for rare species, but it is still feasible to estimate occupancy (Chiari et al. 2013). Moreover, wide swings in density may occur without the patch of habitat becoming unoccupied – and loss of the whole subpopulation may be more important than annual fluctuations in abundance for long-term monitoring.

Determining occupancy is less costly and more feasible for long-term management programs and occupancy models produce unbiased estimates of occupancy and other related parameters (USGS 2005). Bried and Pellet (2011) suggest occupancy data may not provide a shortcut to abundance-based conservation of rare species. A recent study by Dibner et al. (2017) found different factors influence occupancy and abundance, i.e. environmental factors influence species occupancy, whereas local characteristics influence species abundance. They posited that occupancy modeling can still be used to estimate abundance, but the best predicting variables of occupancy may not be the best at predicting abundance.

TIGER BEETLE BIOLOGY

Tiger beetles are within the order Coleoptera, Family Carabidae, Subfamily Cicindelinae. Tiger beetles were historically classified within a separate family (Cicindelidae) as they have several distinct characteristics distinguishing them from other related beetles, including: 1) simple teeth arranged along the inner side of the mandible; 2) a long sickle-shaped mandible; 3) long thin running (cursorial) legs; 4) wide head with large bulging eyes; and 5) larval tunnel-building behavior (Pearson and Vogler 2001).

There are > 2,300 tiger beetle species described and they occur worldwide (except in Antarctica, the Arctic above 65° latitude, and some isolated oceanic islands) (Cassola and Pearson 2000, Pearson and Vogler 2001). Tiger beetles frequent a diversity of terrestrial habitats including alpine meadows, dunes, pine barrens, riparian areas such as riverbanks and sandbars, and temperate, boreal, and tropical forests. Although as a taxon, tiger beetles occupy a plethora of habitats, many species have specific habitat requirements and are sensitive to environmental disturbances (Pearson and Cassola 1992). Their global distribution, well-described systematics, and the strict ecological requirements of specialist taxa make tiger beetles good ecological and biodiversity indicator species (Cassola and Pearson 2000).

Since many important behaviors of tiger beetles, like foraging and mating, are controlled by temperature, i.e. about 56% of adult tiger beetle daily activity is dedicated to thermoregulatory behaviors (Pearson and Vogler 2001), they are considered thermal specialists (Dreisig 1980). Tiger beetles use thermoregulatory behaviors such as basking, stilting, body orientation, burrowing, and shuttling among different habitats to maintain body temperatures between 34°C and 38°C (Dreisig 1980, Knisley et al. 1990). To elevate body temperature significantly above ambient, they rely on open areas for basking. By alternating between basking and stilting, they can keep their body temperature relatively constant (Knisley et al. 1990). Being capable of maintaining high body temperatures helps optimize tiger beetles' speed and prey-capturing efficiency (Knisley 2010).

Tiger beetle adults are diurnal predators spending most of their daily activity time searching for small living arthropods, e.g., ants, spiders, flies, in open exposed habitats (Pearson and Mury 1979, Dreisig 1981). Tiger beetles have specialized visual and locomotory adaptations that enable active hunting. They have large prominent eyes, which make their head wider than

their thorax and have stereoscopic (three-dimensional) vision between 40- and 120-degree binocular overlap in forward vision (Pearson and Vogler 2001). Their long and slender cursorial legs are optimal for rapid movement and running down prey. Tiger beetles chase prey in short, fast sprints with brief stops in between. They sprint so fast in their chases that they lose their vision and have to stop to regain their sight again (Pearson and Vogler 2001). As with many insects, food availability is thought to limit adult size, pupation time, and fecundity in tiger beetles. Pearson and Knisley (1985) found low prey abundance significantly decreased the number of eggs females produced, with smaller females producing fewer eggs than larger females at the same prey abundance levels.

Unlike adult tiger beetles, larvae are sit-and-wait predators (Knisley and Juliano 1988). They have limited mobility as they live in a permanent burrow until they pupate. Larvae have a large, darkened, sclerotized head capsule, which helps camouflage the opening of the burrow when it is flush with the burrow opening. They remain in this posture until prey items approach and then they extend their body out to capture prey with their powerful mandibles, anchoring themselves to the burrow with dorsal hooks (Pearson and Knisley 1985). Prey is consumed within the burrow. While serving as preferred prey for adult tiger beetles, ants are an important predator of tiger beetle larvae (Knisley 1987). When danger is present, larvae will retreat into their burrow until it is safe to return to the surface again.

Food limitation is a major factor in larval development rate (Pearson and Knisley 1985) and survival (Meyer 1987). Larvae at low prey abundance levels have a decreased development rate (Meyer 1987). Studies of *Cicindela arenicola* Rumpff reveal larvae given supplemental food took a shorter period of time to reach the third instar relative to larvae not given extra food (Bauer 1991). The minimum development time required for food-supplemented larvae was two

years, while developmental time for non-supplemented larvae was nearly double that (Bauer 1991). Food limitations as larvae can also result in smaller pupae and adults after emergence (Pearson and Knisley 1985).

A significant number of tiger beetles are of conservation concern in North America with 33 of 233 species (15%) experiencing population declines (Cornelisse and Hafernik 2009). Many at-risk species have high fidelity to specific habitats, which are often threatened by both anthropogenic and natural causes. Although extirpations undoubtedly occurred historically, recolonization was more likely with higher connectivity among habitat patches. Knisley (2010) posited that tiger beetle populations are now more isolated due to habitat fragmentation making recolonization and reestablishment more difficult and less likely.

Anthropogenic disturbance has both positive and negative impacts on tiger beetles. For example, increased human foot traffic has been identified as a major cause of decline in the northeastern beach tiger beetle, *Habroscelimorpha dorsalis dorsalis* (Say), which has been extirpated throughout most of its range except for two beach locations on Martha's Vineyard, Massachusetts (Knisley 2010). Development of recreational beaches and the associated increased foot and vehicular traffic compact sand, leaving females unable to oviposit. Increased foot traffic also caused a decrease in the southern subspecies, *H. d. media* (LeConte), along Atlantic coast beaches (Knisley 2010). On the other hand, dirt roads, trails, and paths are exclusive habitat for some 20 different species and subspecies of tiger beetles (Knisley 2010). In fact, the federally endangered Ohlone tiger beetle, *C. ohlone* Freitag and Kavanaugh, has increased due to mountain biking and hiking paths (Knisley 2010). Mountain biking in *C. ohlone* areas was kept at a reduced speed of 8–12 kph to limit adult mortality (Cornelisse et al. 2013). With these bike speed regulations, *C. ohlone* population growth increased by 42–58% (Cornelisse et al. 2013).

Maintenance of dirt roads, trails, and paths create gaps and expose bare soil in areas where vegetation would otherwise proliferate.

Cornelisse et al. (2012) created artificial bare patches to increase habitat for the endangered Ohlone tiger beetle. Vegetation was removed from plots, exposing bare soil to increase suitable habitat, resulting in a significant increase in *C. ohlone* colonization and removal of a major limitation on suitable habitat. Dispersal occurred from a core area into the artificial plots, although dense vegetation may have acted as a dispersal barrier. Implications of the artificial bare plots on *C. ohlone* population viability were not clear, however, as artificial bare plots had the potential to create negative density-dependence as increased oviposition and larval density increased competition, thereby reducing larval survivorship (Cornelisse et al. 2013).

NORTHERN BARRENS TIGER BEETLE

Northern barrens tiger beetles are medium-sized (12–14.5 mm), strikingly metallic green with distinct unconnected lunules (white markings) on their elytra. Their apical and humeral lunules are reduced to two dots and the middle band is continuous into a slightly sinuate line (Canada 2009). Larvae are cream-colored and reach a length of ~25 mm (Canada 2016). They have a large hump with an inner and median hook on their fifth abdominal segment that help maintain their position in the burrow and anchor them as they attack prey (Canada 2009). They live in a permanent burrow for all three instars and eventually pupate within the same larval burrow (Meyer 1987).

Northern barrens tiger beetles have a two-year life cycle. Adults emerge in late summer (early August) to forage for several weeks until early October, when they dig burrows or retreat under objects such as pine litter or rocks to overwinter (Mawdsley 2007). Knisley et al. (2009) documented a higher concentration of beetles in pine stands in September, suggesting that is

where they likely over winter. Pupae require a surface temperature threshold of 19°C before they metamorphose into adults and emerge from their burrows (Knisley et al. 1990). Adults become active again the following spring (early to mid-April), mating and ovipositing (Canada 2009). Adult activity peaks by mid-April and remains high through May until early June. By mid-June most adults die, although some may live until early July.

Availability of potential oviposition sites is an important driver in tiger beetle habitat selection (Cornelisse and Hafernik 2009). Females have hairs covering their ninth and tenth abdominal segments, which are sensitive to soil structure and moisture levels (Cornelisse and Hafernik 2009). Oviposition sites are typically shaded sandy soils with small pebbles (< 2 cm diameter), often with mosses, lichens, and sedges present (Knisley et al. 1990, Mawdsley 2005, 2007). Shade reduces sand surface temperatures and soil moisture helps prevent egg and larval desiccation and heat stress. Oviposition sites are generally located away from areas where adults are typically found (Mawdsley 2007). Females oviposit ~ 50 eggs, each laid singly in 3–5 mm deep holes, 10–14 days after mating (Shelford 1908). Site selection is critical for larval survival as they must persist in a permanent burrow for the duration of their development. If the burrow is disturbed, larvae will construct a new burrow nearby.

Females will not oviposit in areas with high vegetation cover, and if vegetation invades the larval burrow, larvae disappear (Knisley 2010). Since females are highly specific in ovipositional site selection, dispersal and colonization are likely limited to availability of suitable habitat. These strict habitat requirements have made this tiger beetle naturally rare throughout its range (Canada 2016).

FIRE-DEPENDENT HABITAT

Pitch pine (*Pinus rigida* Mill)-scrub oak (*Quercus ilicifolia* Wangenh) communities commonly occur on xeric outwashes, dune deposits, and exposed ridge tops of acidic bedrock throughout the northeast (Motzkin et al. 2002) and support fire-dependent species that have strategies for regeneration or survival after fire (Motzkin et al. 1999). ‘Dwarf’ pitch pine communities occurring at elevations below regional tree-line are extremely rare across the US. These trees have stunted vertical growth reaching ~ 4 m tall, but can grow along the ground. Both of these communities are a priority for conservation since they are uncommon, support high numbers of endemic rare plant and animal species, and are threatened by anthropogenic alteration to the disturbance regimes that maintain this habitat (Motzkin et al. 1999).

Pitch pine communities rely on fairly frequent low intensity fires to open the canopy and decrease the organic surface layer to allow regeneration. Benefits of occasional fire are stem sprouts along the trunk and crown, basal sprouts, rapid initial productivity, and increased seed production (Forman and Boerner 1981). Pitch pine is a polymorphic species with individuals having serotinous or non-serotinous cones. Serotinous cones depend on exposure to high heat so resin sealing the cone scales melts and releases seeds (Givnish 1981).

Oaks are also affected by occasional fires. Exposure to fire causes basal sprouts and rapid initial productivity (Forman and Boerner 1981). Occasional or mild fires are beneficial, but frequent or severe fires injure existing pitch pine or oak, preventing development of “old-growth” stands (Motzkin et al. 1999) that some rare species depend on. On the other hand, higher intensity or frequency fires in these habitats create patches of bare mineral soil that break up the woody plant monoculture and allow shade intolerant herbaceous species to flourish.

Charcoal-to-pollen ratios have shown a dramatic decrease in frequency of fires since World War II (Motzkin et al. 1999). The decrease in fire frequency may result in more fire-intolerant species invading into fire-dependent habitats. White pine and hardwoods such as maple, that are fire intolerant and shade tolerant, encroach on and eventually subsume pitch pine barrens in the absence of fire (Motzkin et al. 1999).

Many agencies are increasingly relying on prescribed fires to restore and maintain barrens habitat. Prescribed burns are controlled to allow for low intensity burns, which slows natural succession and creates open areas (Brown and Smith 2000). Prescribed fire results in different impacts on vegetation than historical fires (Motzkin et al. 1999). Historically, pitch pine forests would burn during dry summer months, but because of “open burning” regulations and dangers associated with nearby human habitations, prescribed burns are conducted in spring and are limited in summer. Prescribed fires in the spring reduce the chances of severe fires, but have little effect on competing hardwood and shrub root systems (Motzkin et al. 1999). This is primarily due to the deep duff layer still being moist from snow melt. Without damage to hardwood and shrub root systems, they are still able to survive in pitch pine forests unless other methods are used alongside fire. An example management regime is that the Albany Pine Bush uses a growing-season mow + herbicide treatment along with their fire + mow treatment to reduce scrub-oak density and restore open barrens (Bried and Gifford 2010). Target sites are mowed during the growing season and all vegetation cut using a Hydro-Ax, leaving debris on site. After the debris dries, prescribed fires are conducted, creating high intensity fires to expose bare mineral soil. After the growing season, target scrub-oak crowns are sprayed with herbicide to create gaps in the canopy. Results indicate scrub-oak density drastically decreased using this new treatment method.

RESEARCH OBJECTIVES

With only a single recognized population in New York State, the northern barrens tiger beetle is at risk of extirpation. To reduce the probability of the loss of this population, to facilitate projections of population dynamics, and to aid in the preservation and restoration of suitable habitat, a more fundamental understanding of northern barrens tiger beetle population size, habitat and prey preferences, and prey abundance is needed. Specific objectives of this research are to:

1. Identify environmental variables associated with suitable habitat for northern barrens tiger beetle adults and larvae.
2. Understand population dynamics and dispersal of northern barrens tiger beetle.
3. Determine prey abundance within and near known northern barrens tiger beetle populations.

Results from this research will support development of a northern barrens tiger beetle management plan and may contribute more broadly to conservation of other rare and endangered organisms.

CHAPTER 2

ESTIMATING ABUNDANCE AND OCCUPANCY OF NORTHERN BARRENS TIGER BEETLE IN AN ENDEMIC NEW YORK POPULATION

ABSTRACT

Reliable monitoring of rare species abundance in space and time is critical to determine their extinction risk. The goal of this research was to determine whether mark-recapture population size estimates or occupancy estimates of *Cicindela patruela patruela* Dejean within Sam's Point Preserve, are feasible for continued long-term monitoring. Mark-recapture surveys were conducted over three years; however, low durability of marks in the first year, restricted population estimates to the last two years. All three years of data were used for occupancy estimates. Over two years, the population grew from 81 individuals (95% CI 75 to 99) to 109 individuals (95% CI 101 to 128) with an occupancy estimate of 0.79 ± 0.16 . Results from this study will be used to determine the best method to use in rare species monitoring.

INTRODUCTION

The northern barrens tiger beetle, *Cicindela patruela patruela* Dejean (Coleoptera: Carabidae) is widely distributed throughout eastern North American pine-oak barren ecosystems, usually only occurring in small discrete populations (Mawdsley 2005). This beetle is at risk of declining due to habitat destruction and modification, alteration of natural processes, e.g., fire, and over-collection (Schlesinger and Novak 2011). Although *C. patruela* occurs over a wide geographic range, it has a patchy distribution. In NY state, it has been extirpated from six historical locations and is currently found at only a single location (Schlesinger and Novak 2011). The sole NY population of *C. patruela* was discovered in the dwarf pitch pine barrens at Sam's Point Preserve on the Shawangunk Ridge, New York in 2004 (Schlesinger and Novak 2011). It continues to occupy this restricted area with populations on either side of Lake Maratanza along the northwestern and southeastern ridges.

Reliable and accurate monitoring of population sizes of rare species abundance in space and time is critical to determine extinction risks. Abundance is a common metric used to characterize a population, while changes in abundance allow managers to assess risk and efficacy of management efforts (Mackenzie et al. 2004). Two common ways to assess populations are abundance estimates and occupancy. Abundance is often extrapolated from mark-recapture methods, which can yield reliable and accurate results, but are usually more costly and time-consuming than occupancy estimates (Lettink and Armstrong 2003). The probability of observing an individual should be considered before developing a management plan based solely on changes in abundance (Mackenzie et al. 2004). Mark-recapture methods can require a high level of effort that is not always feasible for long-term monitoring programs.

Occupancy is often utilized in monitoring programs as a surrogate for abundance as it generally requires less effort (Mackenzie and Bailey 2004). At appropriate scales, occupancy is positively correlated with abundance, i.e. occupancy increases with increasing abundance (Gaston et al. 2000). Dibner et al. (2017), however, found different factors influenced occupancy and abundance, i.e. environmental factors influence species occupancy, whereas local characteristics influenced species abundance. They suggested occupancy modeling can still be used to estimate abundance, but the best predicting variables of occupancy may not be the best at predicting abundance. Occupancy is similar to mark-recapture, except it does not need marked individuals. Instead, the species of interest is recorded as detected or not detected in the sampling unit. Care should be taken, however, as non-detection does not always mean a site is not occupied, i.e. a false negative can occur if the organism is there, but was not detected during that survey period. It is often difficult to estimate abundance for rare species, but it is still feasible to estimate occupancy (Chiari et al. 2013). Moreover, wide swings in density may occur without the patch of habitat becoming unoccupied – and loss of a whole subpopulation may be more important than annual swings in abundance for long-term monitoring. Occupancy models produce unbiased estimates of occupancy and other related parameters (USGS 2005).

The New York Natural Heritage Program (NYNHP) conducted detection surveys for *C. patruela* from 2006 to 2008 at Sam's Point Preserve. Survey results indicated a larger occupied area within Sam's Point Preserve pine barrens than previously thought (Schlesinger and Novak 2011). Due to the rarity of *C. patruela* within the state, it has been identified as a priority species for monitoring and for development of a management plan. The goal of this study was to estimate population size, via mark-recapture and to develop occupancy estimates to allow an evaluation of which method is most feasible for continued long-term monitoring.

METHODS

Study area

Sam's Point Preserve is located at the southwestern end of Minnewaska State Park in the Shawangunk Mountains of New York state (Fig. 2.1). This preserve covers approximately 20.2 km², encompassing Lake Maratanza and a globally rare dwarf pitch pine (*Pinus rigida* Mill.) community. Like other pitch pine barrens communities, Sam's Point preserve supports numerous rare species and is at risk due to fire suppression (Motzkin et al. 2002). Gaps in the dense pitch pine cover are typified by thin soil and a mix of sand, pebbles, cobble, and exposed slabs of, erosion-resistant Silurian quartzite conglomerate (Seischab and Bernard 1996). Sam's Point Preserve is at the highest elevation of the Shawangunk Mountain range and is typified by temperature extremes, high wind speeds, and reduced relative humidity.

Within the preserve, two subpopulations of *C. patruela* are located on the northwestern and southeastern ridges surrounding Lake Maratanza. These subpopulations are ~ 1 km apart with the only possible corridor being the Carriage Road Trail, which extends around the entirety of Lake Maratanza (Fig. 2.1).

The northwestern subpopulation covers 0.1 km along the ridge and is ~144 m from Lake Maratanza. This subpopulation is on the ridge above the carriage road trail and is ~59 m from it. The southeastern subpopulation covers ~316 m along the ridge and is ~386 m from Lake Maratanza. This subpopulation is located off trails and is ~100 m from the Carriage Road Trail and ~34 m from the ice caves trail.

Mark-recapture

Individual beetles were marked, released, and recaptured in the summers of 2017, 2018, and 2019 in Sam's Point Preserve. Each day represented a new marking period and since *C.*

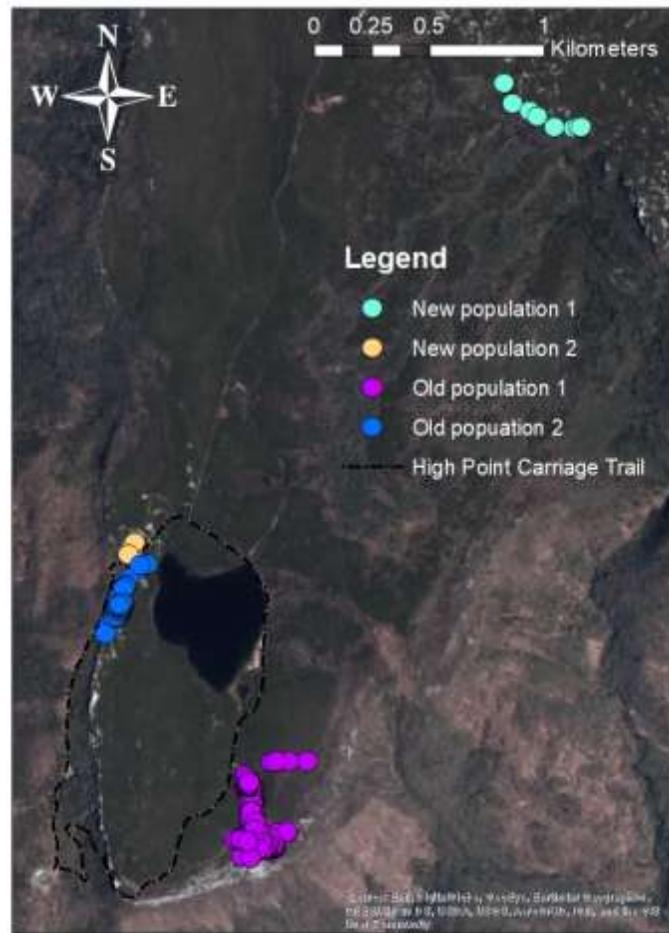


Figure 2.1: Previously identified (2014; purple and blue) and newly discovered (2019; orange and teal) populations of *Cicindela patruela* within Sam's Point Preserve, NY.

patruela die as the summer goes on, it was considered an open system. Twenty-five marking periods were designated in 2017 encompassing 7 weeks (15-May to 23-June and 14 marking periods in 2018 (16-May and 14-June). Adults were captured using a 38 cm flexible net or identified using binoculars from a distance to minimize disturbance to the beetles (Hudgins et al. 2011). On sunny days, a serpentine pattern was walked from the access point across a delineated area of patchy openings bounded by contiguous forest cover resulting in a near complete

assessment of each habitat patch. On cloudy or rainy days, loose rocks were opportunistically lifted to check for hiding beetles encountered during the walk.

In 2017, the first year of the study, captured individuals were sexed and marked with a unique code on the elytra (Fig. 2.2A). I used a white Sharpie™ oil-based extra-fine-point paint pen to create a unique dot code using eight spots. Due to an unusually wet season, the marks faded noticeably over time (Fig. 2.2B). As individuals with fading marks were captured and recorded, they were given a new mark code. Even so, these data were withheld from mark-recapture analysis due to unreliability of marks.

A new marking method was developed for the 2018 field season and used again in 2019. Marks were made from the center of a hole-punched piece of Rite-in-the-Rain® paper (JL Darling LLC, Tacoma, WA) cut in half, with numbers written on it. These labels were attached to individual beetles with cyanoacrylate (super glue) after the elytra was rubbed with a Q-tip soaked in nail polish remover (acetone) then lightly scuffed with 100-grit sandpaper. No marks were lost or faded using this technique (Fig. 2.2C). The new marks did not appear to hinder normal adult behavior as beetles were observed successfully foraging and mating with marks present.

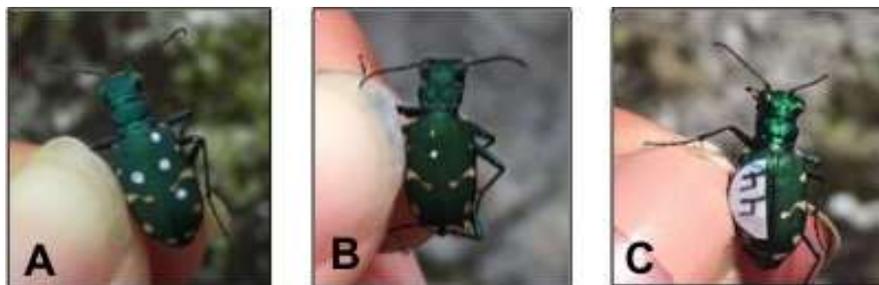


Figure 2.2: Marking techniques for *Cicindela patruela*. A) 2017 technique using oil-based paint pen (code = 0010 1110), B) noticeably faded 2017 mark, and C) 2018 and 2019 technique using water-proof paper.

Population size estimates

Assumptions of the Peterson-Lincoln estimate for closed population mark-recapture programs are as follows: 1) there is no birth, death, or emigration during the study (closed population), 2) all animals have the same probability of being caught (equal catchability), and 3) marks are not lost. In order to meet model assumptions of a closed population, data collection in 2018 ceased when surveys indicated that *C. patruela* abundance was declining. Compliance with this assumption was additionally met by pruning the final survey week from the data set. This ensured the population size remained constant, with no *C. patruela* entering or leaving. The truncated 2019 data set reflected a true closed population since the survey was ended a month before *C. patruela* typically begin to die.

The estimated numbers of adults present May–July 2018 and May–June 2019 were accepted as the population size. Males and females were pooled because sample sizes were relatively small (2018: $n = 72$; 2019: $n = 97$) and there was no evidence that there were sex-based differences in emergence or death. Population size was estimated by capture-mark-recapture data using closed population models. Data from 2018 and 2019 were compared using closed capture models. All models were built using the program MARK (White and Burnham 1999). The program MARK computes the Akaike's Information Criterion (AICc) corrected for small-size samples. The lowest value of AICc was assumed to be the model with the highest support for the capture-recapture data (Chiari et al. 2013).

Occupancy estimate

Mark-recapture data were manipulated in ArcMap to fit occupancy models. A fishnet grid of 25 m² squares was overlaid on the three seasons of mark-recapture data points. Grid size of 25 m² was selected based on the average distance moved by beetles. Average distance was

determined by using the mark-recapture data set and point distance tool in ArcMap. Mark-recapture points showed where *C. patruela* individuals were detected. Squares used as survey sites (n = 78) were determined by areas surveyed with equal effort during the mark-recapture studies. Grid squares with points present were recorded as detected (1) and those without points were recorded as not detected (0) for every survey occasion (2017: n = 23; 2018: n = 14; 2019: n = 9).

Assumptions of occupancy models include: 1) occupancy state is closed, 2) sites were independent, 3) there is no unexplained heterogeneity in occupancy, and 4) no unexplained heterogeneity in detectability. Multiple-season occupancy models yield estimates of occupancy, colonization, extinction, and detection probability. Occupancy has one parameter for probability of occupancy in the first season. Colonization and extinction both have two parameters for the two between season opportunities for species to colonize or become extinct in a location.

Occupancy was estimated by detection/non-detection data using simple multiple-season models. Given violation of closure (survey sites not independent due to using grid overlay), the estimated occupancy parameter was instead interpreted as the probability that a block was used at least once during the survey period. All models were built using the program PRESENCE (MacKenzie et al. 2017). The program PRESENCE computed AIC, and the model with the lowest value of AIC was assumed to be the one with highest support for the occupancy study.

RESULTS

Mark-recapture

In 2018 and 2019, a total of 169 tiger beetles were captured, with 83 (49.1%) being males and 82 (48.5%) being females (2.4% were unknown due to difficulty in sexing some of the

beetles). In 2018, more females were marked than males, unlike in 2019, where more males were marked than females (Table 2.1). Recapture rate of males increased by 12.5% while recapture rate of females decreased 16.6% between seasons. Overall recapture rate remained almost constant between seasons (Table 2.1).

In 2018, 35 of 72 marked beetles were recaptured (Table 2.1). One female was recaptured 231.1 m from the original capture point, while a male was recaptured 161.5 m away. Other recapture distances were less expansive, ranging from 3.2 m to 117.7 m. Recapture distances did not vary between males and females ($t = 0.18$, $df = 26$, $P = 0.43$). In 2019, 47 beetles were recaptured with maximum distances of 165.8 m for a female and 107.6 m for a male. Other recapture distances ranged from 3.8 m to 126.7 m. No individuals were observed to move between the northwest and southeast subpopulations. Averages for recapture distances were not found to be different between males and females ($t = 0.44$, $df = 35$, $P = 0.33$). Mean recaptures did not vary by sex or year (sex: $t = 0.38$, $df = 63$, $P = 0.35$; year: $t = 0.07$, $df = 62$, $P = 0.47$).

Table 2.1: Mark recapture data by sex for northern barrens tiger beetles (*Cicindela patruela*) within Sam’s Point Preserve, NY for 2018 and 2019.

Year	Activity	Male	Female	Unknown	Total
2018	Marked	31	39	2	72
	Recaptured	14	21	—	35
	Percent recaptured	45.2%	53.8%	—	48.6%
2019	Marked	52	43	2	97
	Recaptured	30	16	—	47
	Percent recaptured	57.7%	37.2%	—	48.5%
TOTAL	Marked	83	82	4	169
	Recaptured	44	37	—	82
	Percent recaptured	53.0%	45.1%	—	48.5%

The best-fitting MARK model (Mt f0_s, p) indicated capture and recapture probabilities were the same within and between sexes (p), but capture probabilities varied over time (Mt), and *C. patruela* numbers differed by sex (f0_s). Alternative models included time, behavior, and null models ($\Delta\text{AICc} > 2$; Table 2.2).

Table 2.2: Relative AICc values and Akaike weight (*w*), for each model for population estimates for northern barren tiger beetle (*Cicindela patruela*) captures in 2018 and 2019 at Sam’s Point Preserve, NY.

Year	Model	K*	-2Log (L)**	ΔAICc^{***}	<i>w</i>
2018	Mt f0 _s , p	16	401.3	0.0	0.996
	Mt f0 _s , p _s	30	383.2	11.3	0.003
	Mb f0 _s , p, c _s	5	441.2	17.5	0.0002
	M0 f0 _s , p _s	2	448.0	18.2	0.0001
2019	Mt f0 _s , p	11	220.0	0.0	0.992
	Mt f0 _s , p _s	20	210.7	9.6	0.008
	Mb f0 _s , p	3	277.8	41.7	0.00
	M0 f0, p	2	333.4	95.3	0.00

*K represents number of parameters in the model.

** -2Log (L) is twice the negative log-likelihood value.

*** ΔAICc represents in the difference in AICc value relative to the top model.

In 2018, the population estimate was 46 females (95% CI of 42 to 55) and 35 males (95% CI of 32 to 44), resulting in an adult population estimate of 81 (95% CI 75 to 99) *C. patruela* within Sam’s Point Preserve. As 72 *C. patruela* individuals were marked throughout the entire marking period, this study approached a complete census. Density of *C. patruela* within the area surveyed (2,175 m²) at Sam’s Point Preserve was 0.037 (95% CI 0.034 to 0.046) beetles/m². In 2019, the total population estimate was slightly higher, 109 individuals (95% CI, 101-128), composed of 48 females (95% CI of 44 to 57) and 61 males (95% CI of 57 to 72), which was 11 adults more than the number marked. Density of *C. patruela* within the area surveyed (2,175 m²)

at Sam's Point Preserve was 0.05 (95% CI 0.046 to 0.059) beetles/m². Mean capture probability over the 2018 marking period was 0.13 ± 0.03 (Fig. 2.3) Two survey periods, June 8th and June 11th, had increased capture probabilities of 0.27 (95% CI 0.18 to 0.38) and 0.23 (95% CI 0.15 to 0.34), likely due to greater catch effort as more surveyors participated in the sampling. Average capture probability over the entire 2019 marking period was 0.20 ± 0.03. May 20th, 21st, 29th, and 30th had lower capture probabilities than the subsequent survey periods (Fig. 2.3). Each of these days were characterized by low temperatures and significant precipitation.

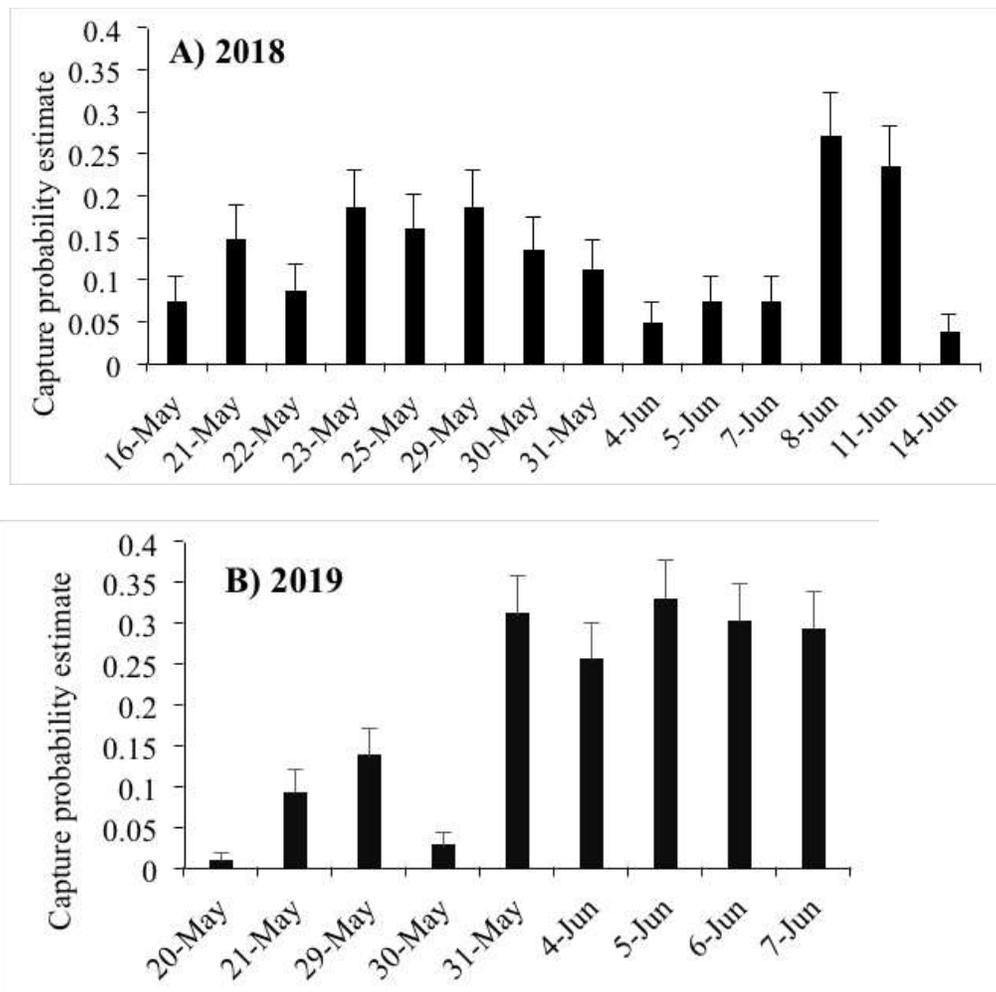


Figure 2.3: Capture probability estimates (± SE) of northern barrens tiger beetles (*Cicindela patruela*) from A) 16-May through 14-June, 2018, and B) from 20-May through 7-June, 2019 at Sam's Point Preserve, NY from closed capture models.

Occupancy

Detection had 1–46 parameters depending on survey occasion. The best fit model (Ψ , γ , ϵ (year), p (year)) indicated the probability that a site was occupied (Ψ) or colonized (γ) remained constant over the three years, while extinction (ϵ) and probability of detection (p) differed among years, but were constant within years (Table 2.3).

Table 2.3: Relative AIC values and Akaike weight (w), for each occupancy model for the northern barrens tiger beetle (*Cicindela patruela*) captured 2017 through 2019 at Sam’s Point Preserve.

Model	K*	-2Log (L)**	Δ AIC***	w
Ψ , γ , ϵ_y , p_y	7	1327.7	0.0	00.452
Ψ , γ_y , ϵ_y , p_y	8	1326.3	0.6	0.331
Ψ_s , γ_y , ϵ , p_y	7	1330.3	2.6	0.122
Ψ , γ , ϵ , p_y	6	1332.9	3.2	0.093
Ψ , γ , ϵ , p	4	1404.0	70.2	0.000

*K represents number of parameters in the model.

** -2Log (L) is twice the negative log-likelihood value.

*** Δ AIC represents in the difference in AICc value relative to the top model.

The occupancy estimate was 0.79 ± 0.16 and colonization was constant among all three years with an estimate of 0.24 ± 0.06 . Extinction changed among years with an estimate of 0.64 ± 0.06 between year one and two and an estimate of 0.33 ± 0.11 between year two and three. The probability of detection was different among, but not within years with the estimate of year one being 0.05 ± 0.01 , year two 0.17 ± 0.02 , and year three 0.23 ± 0.03 (Fig. 2.4).

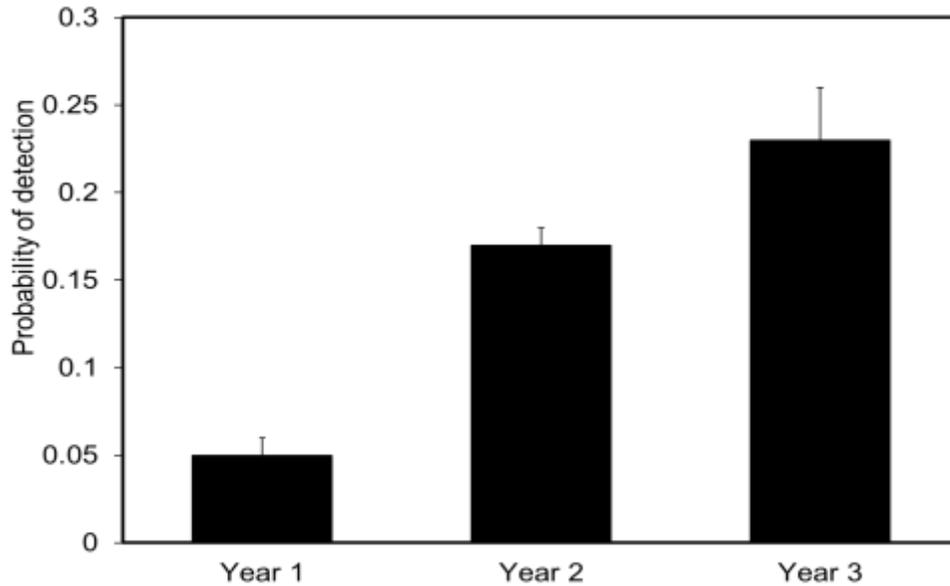


Figure 2.4: Probability of detection (\pm SE) of northern barrens tiger beetles (*Cicindela patruela*) from year 1 (2017) to year three (2019) at Sam's Point Preserve, NY from closed capture models.

DISCUSSION

Mark-recapture models allowed estimation of distances travelled within Sam's Point Preserve. A maximum of 231 m of displacement from the original capture point was recorded with most beetles dispersing mid-May to mid-June. These distances are less than recorded for similar tiger beetle species. Hudgins et al. (2011) observed cobblestone tiger beetles (*Cicindela marginipennis* Dejean) dispersing up to 481 m from their original capture point and Knisley et al. (2016) observed *H. d. doralis* dispersing up to 15 km. Northern barrens tiger beetles were not observed to disperse distances that far, but were observed to disperse along the ridges using bare patches. If an individual was continually flushed from a patch, they would fly over or into thick vegetation to retreat to another patch. This suggests small areas of dense vegetation do not hinder *C. patruela* dispersal. No individuals were observed to disperse between the northwestern and southeastern subpopulations. The distance between subpopulations exceeded the maximum dispersal distance observed and bare patches were not present to serve as corridors between the

ridges. Since there were no observations of *C. patruela* dispersing from one subpopulation to another during the study periods, this suggests connectivity is limited. Spatially isolated small populations are susceptible to a suite of stochastic variables that individually and collectively threaten persistence (Caughley 1994). The low estimates of population size for both subpopulations, indicates they have an elevated risk of extinction and further supports the need for a management plan to increase connectivity between subpopulations.

To put the population size of *C. patruela* at Sam's Point in perspective, the total population was estimated at 81 and 97 individuals in 2018 and 2019, respectively. In Canada, the same species is listed as endangered, yet one population was estimated as 400–1000 beetles (Canada 2016), which is nearly 5–10 times larger than the Sam's Point Preserve metapopulation. A similar species, the endangered northeastern beach tiger beetle (*Cicindela dorsalis dorsalis* Say) had an estimated abundance of ~9,500 individuals in 2014 across 9 regions in the Chesapeake Bay area (Knisley et al. 2016).

Cicindela patruela was detected at 55 of the 78 survey sites (from a grid overlay in ArcMap) over three years. The occupancy estimate (0.79 ± 0.16) does not mean *C. patruela* is absent at the remaining 23 sites, rather they just might not have been detected due to random chance. The estimated probability of detection refers to the probability of detecting *C. patruela* at least once on each survey occasion. Estimates for the probability of detecting *C. patruela* were quite low; year one was 0.05 ± 0.01 , year two 0.17 ± 0.02 , and year three 0.23 ± 0.03 . Over the three seasons, the probability of detection increased. False negatives can lead to an underestimate of the actual number of sites occupied by rare species (Gu and Swihart 2004). After surveying for these tiger beetles for three seasons, it is likely researchers became more adept at detecting *C. patruela* or *C. patruela* abundance increased, thereby increasing the probability of detection.

Although both seasons resulted in near-census mark-recapture surveys with low population estimates, two other small populations of *C. patruela* were found within Sam's Point Preserve in 2019. These two new populations were not included in the mark-recapture or occupancy analyses as these new locations did not have equal search effort between or among years. One new population was found on the other side of the Carriage Road Trail from the northwestern ridge known population (Fig. 2.1). This new population was observed to have at least 10 adult *C. patruela* and > 20 active larval burrows. This subpopulation is assumed to be a part of the known northwestern subpopulation due to their proximity. The second new population was on a ridge on the other side of Sam's Point Preserve. This population had at least 12 adults and 4 active larval burrows. This population is ~3 km from the other subpopulations and is assumed to be isolated from them due to distance and lack of corridors.

Management plans are being developed and implemented to conserve known small populations of *C. patruela*. For example, prescribed fires have been conducted annually in southeastern Canada to restore and manage savannah habitat where *C. patruela* was detected, and surrounding areas of potential habitat were surveyed for new populations as well (Canada 2016). Cornelisse et al. (2012) created artificial bare patches to increase habitat for the endangered Ohlone tiger beetle (*Cicindela ohlone* Morgan). Vegetation was removed from plots, exposing bare soil to increase suitable habitat, resulting in a significant increase in *C. ohlone* colonization, and removal of a major limitation on suitable habitat. This resulted in increased dispersal of *C. ohlone* from core areas to the new artificial bare plots.

CONCLUSIONS

This research suggests *C. patruela* would benefit from a long-term monitoring plan using a combination of both occupancy and mark-recapture methods. Since the area occupied and habitat of *C. patruela* is well known within Sam's Point preserve, ~70% of the suitable habitat is occupied, and adults are flashy, i.e. not cryptic, this population is a good candidate for continued mark-recapture to keep a close account of population size.

Monitoring should be continued at Sam's Point Preserve for this rare species. Since this research was only over two years, mark-recapture should be continued until the annual variation of population size is understood. Once annual variation of *C. patruela* is understood, it would be possible to shift monitoring effort to occupancy surveys. This would allow for park staff to still monitor the population, but with less effort.

CHAPTER 3

HABITAT AND PREY PREFERENCE OF NORTHERN BARRENS TIGER BEETLES IN THE HUDSON VALLEY, NEW YORK

ABSTRACT

Loss or alteration of habitat is among the key drivers of species imperilment worldwide. Identifying critical habitat variables for a given species is fundamental to developing successful management plans for target species. Relatively little is known about the rare northern barrens tiger beetle, *Cicindela patruela patruela* Dejean, in New York State where it currently occurs in a single location. Goals of this study were to understand variables associated with suitable habitat and determine prey abundance and diversity at Sam's Point Preserve. I measured habitat characteristics associated with the presence and nondetection of *C. patruela* adults and larvae. Percent cover of pebbles had a positive relationship with contemporary *C. patruela* distribution as well as with historical range models. Models incorporating larval burrow localities revealed areas with higher percentages of rock slab were not being utilized by ovipositing females. Females selected areas of high pebble percent cover to oviposit. Results of this study will be used to manage for habitat supporting *C. patruela* at its known location and to restore habitat in similar habitats.

INTRODUCTION

There are > 2,300 described tiger beetle (Coleoptera: Cicindelidae) species described worldwide (except in Antarctica, the Arctic above 65° latitude, and some isolated oceanic islands) (Cassola and Pearson 2000, Pearson and Vogler 2001). Of the 22 tiger beetle species recognized in New York, over half are considered rare, at risk, or potentially extirpated from the state (Schlesinger and Novak 2011). Habitat destruction, modification, and alteration of natural succession and disturbance regimes are key factors threatening tiger beetle populations. While identifying large landscape scale habitats necessary for tiger beetle populations is relatively simple, understanding smaller scale habitat features and their relationships to prey abundance, foraging opportunities, key oviposition sites, and larval burrow preferences are critical in developing a management plan to maintain or restore threatened species.

In order to fully understand tiger beetle habitat needs, we must understand composition of their prey. Tiger beetle adults are diurnal cursorial predators, spending most of their daily activity time searching for small living arthropods, e.g., ants, spiders, flies, in open exposed habitats (Pearson and Mury 1979, Dreisig 1981). As with many predators, food availability limits adult size, emergence time, and fecundity. Pearson and Knisley (1985) found low prey abundance significantly decreased number of eggs females produced, with smaller females producing fewer eggs than larger females at the same prey abundance levels. Thus, optimal habitats for foraging are influenced by both prey abundance and quality of prey items.

Unlike adults, tiger beetle larvae are sit-and-wait predators (Knisley and Juliano 1988). They have limited mobility as they live in a permanent burrow from egg hatch through pupation. Food limitation is a major factor in larval developmental rate (Pearson and Knisley 1985) and survival (Meyer 1987). Larvae at low prey abundance levels have a decreased developmental

rate (Meyer 1987). For example, supplemental feeding of *Cicindela arenicola* Rumpff resulted in shorter development times (Bauer 1991). Food limitations as larvae can result in smaller pupae and post-emergence adults (Pearson and Knisley 1985). If food continues to be a limited, tiger beetle populations could begin to decrease over time.

Oviposition site choice is the main driver of tiger beetle habitat selection in laboratory studies of rare *Cicindela hirticollis* Say, 1817 (Cornelisse and Hafernik 2009). Females use sensitive hairs covering their ninth and tenth abdominal segments to evaluate soil moisture and structure prior to oviposition (Cornelisse and Hafernik 2009). Females typically choose to oviposit in shaded sandy soils with small pebbles (< 2 cm diameter) and where mosses, lichens, and sedges are present (Knisley et al. 1990, Mawdsley 2005, 2007). As larvae are constrained to a permanent burrow at the site of oviposition, female site choice is critical to their success. Females will not oviposit in areas with high vegetation cover and if vegetation invades the larval burrow, larval mortality increases (Knisley 2010).

The northern barrens tiger beetle, *Cicindela patruela patruela* Dejean, was formerly found at least six localities in New York State. At present, there is only one known extant population, located at Sam's Point Preserve in the Hudson Valley (Schlesinger and Novak 2001). This population is actually comprised of two small subpopulations located on the northwestern and southeastern ridges in dwarf pitch pine barren habitat. Pitch pine communities likely established in response to frequent fires, and are maintained by occasional fires which are necessary to open the canopy and decrease the organic surface layer to allow regeneration and create natural openings (Forman and Boerner 1981). *Cicindela patruela* rely on these natural openings to forage, mate, and oviposit. Without natural disturbances, vegetation will encroach into these openings, thus decreasing *C. patruela* habitat.

Unfortunately for *C. patruela*, decades of fire suppression in the Shawangunk mountains has limited natural openings and pitch-pine forest regeneration (Schlesinger and Novak 2011, Beers et al. 2011). In the absence of wild or prescribed fire, fire intolerant vegetation will eventually colonize and largely eliminate the open gaps required by this species (Mawdsley 2005). For continued persistence in this locality, as well as potential restoration in formerly occupied areas, it is imperative that a formal management plan be developed and adopted. Any plan must include contingencies for maintaining and expanding natural openings and native plant assemblages. Unfortunately, relatively little is known about the ecology and habitat requirements of this beetle, a necessary prerequisite for developing any conservation program.

In this study, habitat variables are assessed for their importance in predicting presence of adult and larval *C. patruela*. Core objectives were to: 1) evaluate variables associated with adult and larval habitat and 2) determine prey abundance and diversity. Data from this study will provide critical baseline information on basic habitat requirements for this species and information to inform development of a subsequent management plan for the species.

METHODS

Study area

Sam's Point Preserve is located on the highest section, elevation of 698 m, of the Shawangunk Mountains at the southwestern end of Minnewaska State Park (Fig. 3.1). This preserve covers ~ 20.2 km², which includes Lake Maratanza, unique ice cave crevices, and a globally rare dwarf pitch pine community. Like other pitch pine communities (Motzkin et al. 2020), Sam's Point preserve supports numerous rare species and is at risk due to fire suppression. Openings in the dense pitch pine cover are composed of erosion-resistant Silurian

quartzite conglomerate, also termed Shawangunk Grit. Substrate is a mix of sand, pebbles, and cobbles scattered across and between large rock slabs. Sam's Point Preserve is at the highest elevation of the Shawangunk Mountain range, which results in a harsh local climate consisting of extreme temperatures, higher wind speeds, and lower average relative humidity than nearby lower elevation localities.

At Sam's point, beetles are constrained to two discrete areas, one in forest openings along the northwestern ridge overlooking Lake Maratanza and the other on the southeastern ridge. Although populations are separated by the lake, a trail encompasses both habitats and likely provides some population connectivity. Two additional subpopulations were discovered in 2019. One is in close proximity to the known northwestern population while the other was located northeast on another ridge across a valley (Fig. 3.1).

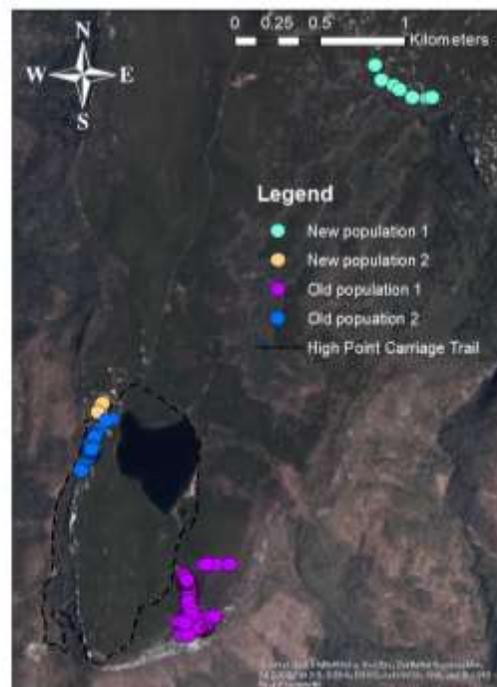


Figure 3.1: Previously identified (2014; purple and blue) and newly discovered (2019; orange and teal) populations of *Cicindela patruela* within Sam's Point Preserve, NY.

Habitat survey

Mark-recapture data points from a 2017 survey (see Chapter 2) were used to randomly select plots where adult tiger beetles were located within Sam's Point Preserve along with confirmed larval burrows. Boundaries were drawn around grouped detected beetle points ($n = 201$) in ArcMap and a random generator placed plots within the boundaries. Plots were placed at least 5 m away from each other. There was a total of 51 plots within the preserve where tiger beetles were present. Within the Preserve, 66 plots were randomly established in habitat with no beetles that was qualitatively similar to habitat where beetles were found (presence of dwarf pitch pine, blueberry, huckleberry, and open areas with varying substrate). I also established 17 randomly located plots in visibly similar habitat outside of Sam's Point Preserve where *C. patruela* was not detected. Some of these were in areas encompassing *C. patruela*'s historic range at Hogencamp Mountain.

Each plot was 5-m radius and percent cover of trees, shrubs, forbs, grasses was estimated, all plants were identified, and substrate categorized using the following classification: sand (< 0.4 cm), pebbles (0.4–15 cm), cobbles (> 15 cm), rock slab. Habitat surveys for larvae were completed at a smaller scale due to the stationary location of burrows. A plot and quadrat were established at every detected larval burrow. Larvae were either observed at the head of the burrow, or the burrow was dug up to ensure it was occupied by a tiger beetle larva. A 1 m² quadrat was placed in each occupied plot with the larval burrow opening in the center. Vegetation and substrate percent cover was recorded for the quadrat.

Prey abundance

At each habitat plot, prey abundance counts (ants) were performed in two 1 m² quadrats. The first was placed 1 m from the center in line with a northerly reading and the second 1 m

south of the center. Any ant entering the quadrat was recorded for 3 min. Ant densities were low, allowing observers to avoid recounting the same ants. Ants within a quadrat were collected over a 1 min time period using an aspirator and later identified to genus using A Field Guide to the Ants of New England (Ellison et al. 2012). Voucher specimens were deposited in the State University of New York College of Environmental Science and Forestry insect museum in Syracuse, New York. Ant species richness, Shannon Wiener (H') diversity, and abundance data were used in random forest models to determine their importance and a t-test was used to evaluate if there were significant differences between plots with and without larvae.

Statistical analysis

A workflow model (Fig 3.2) was developed to use geographic information system (GIS) software (ArcMap 10.6.1) to produce a map of *C. patruela* habitat by canopy cover. For the final map, a canopy raster attribute table was joined to the Minnewaska canopy attribute table, and symbols were added according to tiger beetle presence depending on canopy cover.

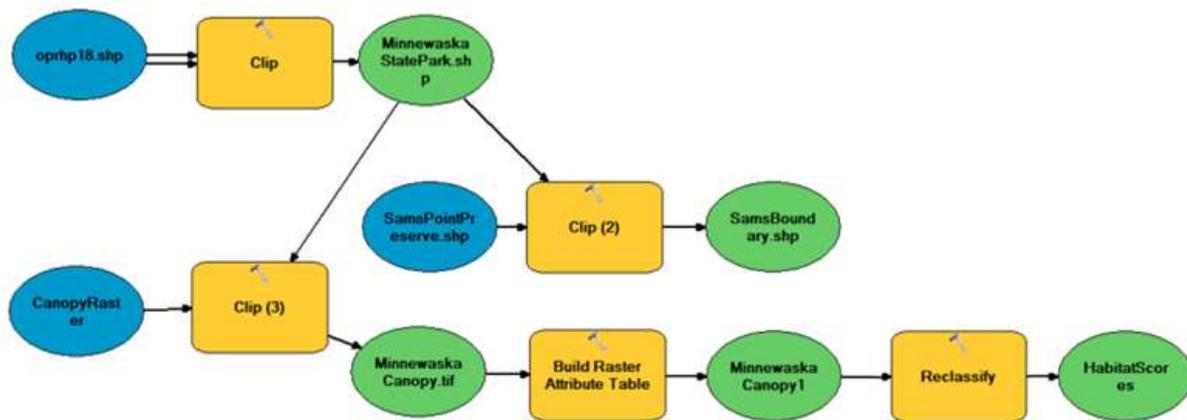


Figure 3.2: Conceptual model showing tools used in ArcMap 10.6.1 to create distance traveled table and habitat scores map.

Percent canopy cover was binned using habitat scores of 0, 1, 2, or 3 (Fig. 3.3). A score of 3 indicates areas with 18% or less canopy cover and represented areas suitable for *C. patruela*. A score of 0 indicated an area with 42% or greater canopy cover and were not likely to have tiger beetles. Scores of 1 or 2 had an intermediate percent of canopy cover where *C. patruela* may be detected, but at lower rates than areas with a score of 3.

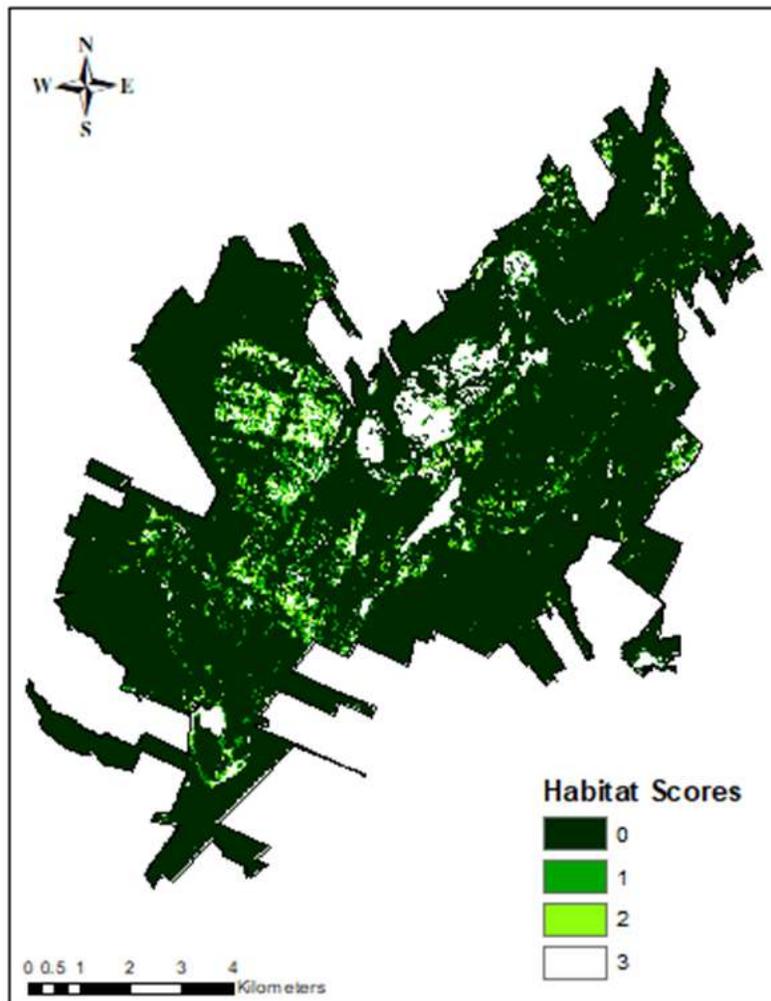


Figure 3.3: Northern barrens tiger beetle (*Cicindela patruela*) habitat suitability scores. Score of 3 indicates suitable habitat and score of 0 indicates low suitability habitat based on canopy cover in Minnewaska State Park.

A large dataset ($n = 134$) was generated containing both binomial (mark-recapture) and continuous (habitat percent cover) variables. Since there were many predictor variables and the interactions among them unknown, the statistical classifier Random Forests (RF, see justification below) was used to model the data instead of generalized linear models. In this model, the predictor variables were ‘vegetation’ and ‘substrate percent cover within each plot’, with presence or absence of tiger beetles used as the response variable. Variable importance plots were generated using Random Forest models indicate how much more accurate a model could be with individual predictor variables. Higher values of mean decrease in accuracy indicate which predictor variables are most important to the classification (Cutler et al. 2007).

Classification procedures are one of the most used statistical methods in ecology (Breiman 2001). A new and powerful machine learner classifier is Random Forests. RF models generate many classification trees (i.e. training the model) and aggregate results to create a final model (Liaw and Wiener 2002). Training of the model is considered stochastic since the same input can result in a different output, but the completed model is considered deterministic since the same input will always give the same outputs, unless the model is re-trained.

Compared to other statistical classifiers, advantages of using RF are: 1) extremely high classification rate, 2) ability to efficiently run large data sets with hundreds of input variables without variable deletion, 3) capability to determine variable importance, and 4) ability to model complex interactions among predictor variables (Breiman 2001, Liaw and Wiener 2002, Cutler et al. 2007, Rodriguez-Galiano et al. 2012). The RF algorithm initiates with many bootstrap samples from the original dataset. The bootstrap dataset does not include all of the original data, but on average it contains ~ 63% of the original data (Cutler et al. 2007). To train the model, a classification tree is fit to each bootstrap sample, but there are only a set number of randomly

selected variables at each node. The number of variables tried at a node, and the number of classification trees used, can be adjusted to increase model accuracy.

Once classification trees are fully grown, they are used to predict out-of-bag observations (Cutler et al. 2007). Out-of-bag observations are from the original dataset, but were not used in the bootstrap sample (Breiman 2001, Cutler et al. 2007). After out-of-bag observations are run through the trees, a majority vote is calculated of how accurate the trees were at predicting observations. Output from the majority vote is the out-of-bag error rate, which estimates accuracy of the completed models. The best model is selected by changing the number of classification trees used and number of random variables used at each node. On the initial run, the standard number of trees was set at 500 and the number of random variables at each node was the square-root of the number of predictor variables. By trial and error, a combination of trees ($n_{tree} = 1000$) and random variables ($m_{try} = 5$) were used to acquire the lowest out-of-bag error combined with the highest accuracy (Table 3.1). Two-sample t-tests assuming unequal variances were used to determine significance of the most important variables from each model.

Table 3.1: Random forests models for northern barrens tiger beetles (*Cicindela patruela patruela*) from 2017–2019 at Sam’s Point Preserve.

Model	Detected class error	Not detected class error	Out-of-Bag estimate of error rate
Historic Habitat	0.100	0.063	8.5%
Sam’s Point Preserve Habitat	0.100	0.157	13.9%
Larval Burrow Habitat	0.375	0.356	36.6%

The first model included sites from Sam’s Point Preserve where *C. patruela* was detected and sites outside of Sam’s Point Preserve encompassing the known historical range of *C.*

patruela, but where no individuals were detected in this study. The sites surveyed within the historical range were located at Hogencamp Mountain in the Hudson Valley. Sites were selected from looking at satellite pictures to locate open patches surrounded by pitch pine. The distances between these sites make it unlikely for the current population of *C. patruela* to disperse to areas outside of Sam's Point Preserve. A lack of corridors connecting these areas also decreases the likelihood of *C. patruela* dispersal. The purpose of this model was to provide potential insight to why *C. patruela* is now extirpated from historic ranges outside of Sam's Point Preserve.

The second model included sites only from Sam's Point Preserve where *C. patruela* were detected and sites where they were not detected, but which had similar vegetation and substrate to sites with *C. patruela*. In other words, these sites had similar environmental conditions and were within a proximity where *C. patruela* could disperse and colonize, but *C. patruela* were not present. The goal of this model was to determine what environmental variables might be important for *C. patruela* in selecting appropriate habitat.

The third and last model included sites only within Sam's Point Preserve where larval burrows were detected and similar areas close to these sites, but which did not have burrows. The goal of this model was to determine what environmental variables were important in habitats that females select for oviposition.

RESULTS

Cicindela patruela were observed almost exclusively in areas characterized as 'open'. Habitat with dwarf pines cover < 18% encompassed 81.6% of *C. patruela* observations, while no beetles were observed in locations where canopy cover was > 42%. The historical range habitat model fits 91.5% (100% - out-of-bag error rate, Table 3.1) of the data while Sam's Point

Preserve habitat model fits 86.1% of the data. The larval burrow habitat model fits 63.4% of the data. Detected class error is the number of times a tree resulted in “beetles not detected” when the result from the boot strap data set was actually “beetles detected”. Not detected class error is the number of times a tree resulted in “beetles detected” when the result from the boot strap data set was “beetles not detected”.

Since tiger beetles were found more often in open areas adjacent to areas suitable for oviposition, percent cover of sand, pebbles, and pine cover were hypothesized to be important predictors of tiger beetle presence. Two of these three variables were in fact identified as important to classifications within Sam’s Point Preserve (Fig. 3.4A), including the most important variable to the model, pebble cover.

Surprisingly, ant abundance and H’ diversity were not important variables in any model. In plots where *C. patruela* adults were detected, there was a mean of 2.3 ± 3.7 ants and in non-detected areas, 2.4 ± 4.9 ($p_{1.66,88} = 0.16$). Six genera of ants were detected; *Camponotus*, *Crematogaster*, *Dolichoderus*, *Formica*, *Formicoxenus*, and *Lasius*. All six genera were present at plots where *C. patruela* was not detected, whereas, only ants in the genera *Camponotus*, *Formica*, and *Lasius* were present at positive *C. patruela* plots.

The most important variables for the outside of Sam’s Point Preserve model were pebble cover, pine/twig, and road percent cover (Fig. 3.4B). Of the 51 plots where *C. patruela* were detected, 88% had pebbles present while only 33% of the 66 plots where *C. patruela* were not detected had pebbles present. Percent cover of pebbles was greater in *C. patruela*-detected plots with a mean of $4.7\% \pm 7.4$ and a mean of $0.8\% \pm 40.4$ in non-detected plots ($p_{1.67,53} = 0.0003$).

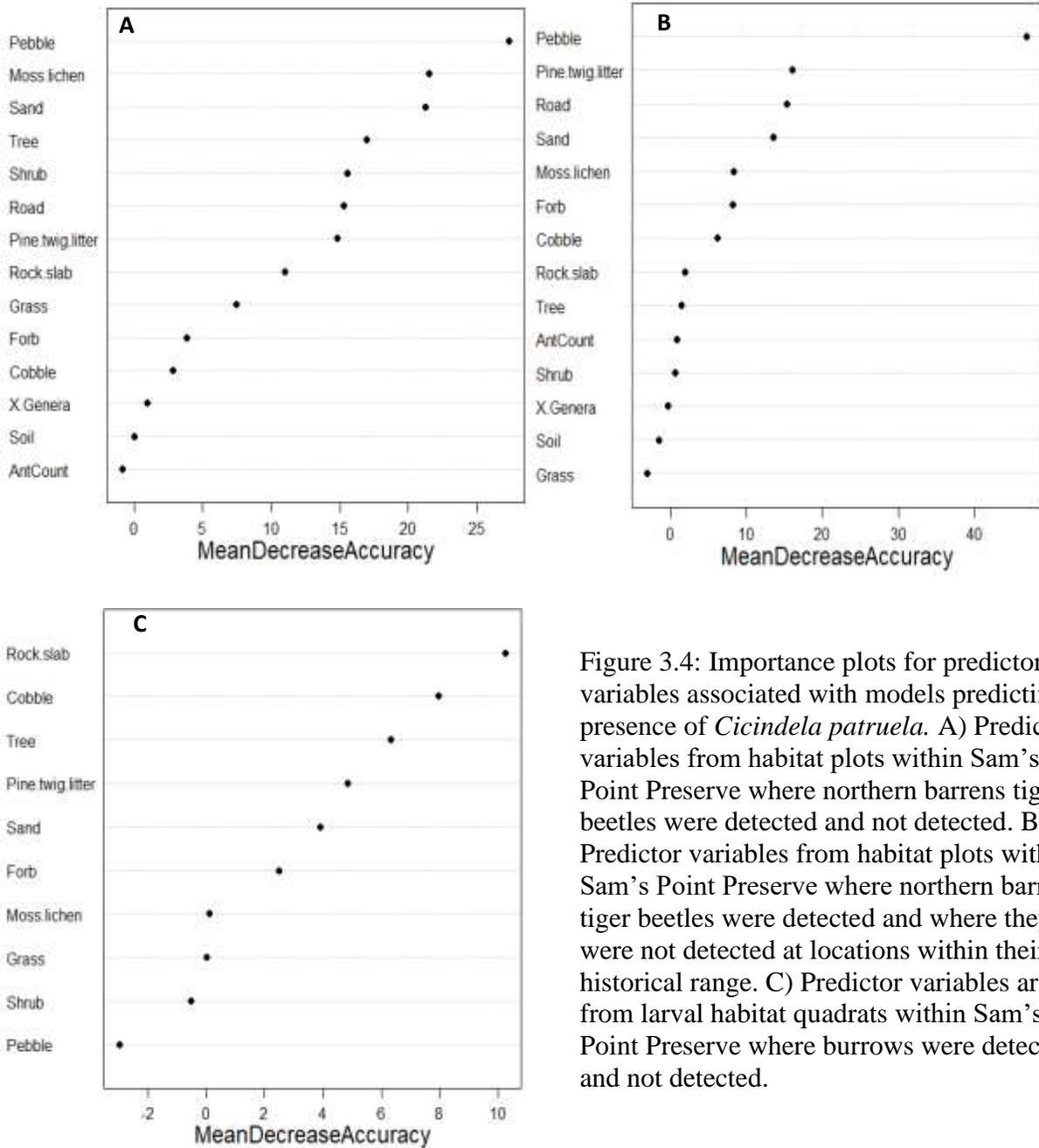


Figure 3.4: Importance plots for predictor variables associated with models predicting presence of *Cicindela patruela*. A) Predictor variables from habitat plots within Sam's Point Preserve where northern barrens tiger beetles were detected and not detected. B) Predictor variables from habitat plots within Sam's Point Preserve where northern barrens tiger beetles were detected and where they were not detected at locations within their historical range. C) Predictor variables are from larval habitat quadrats within Sam's Point Preserve where burrows were detected and not detected.

Outside of Sam's Point locations, only 9% of 17 plots had pebbles present with a mean of 0.1% \pm 3.7 ($p_{1.67, 50} = 2.81E-05$). Both models resulted in a positive relationship of *C. patruela* presence with increased pebble percent cover (Fig. 3.5A and Fig. 3.6A). When the trendline is in the negative region, it means it is unlikely to detect a beetle. As pebble percent cover increased,

likelihood of detecting beetles increased. After 2% pebble cover, there was a constant likelihood of beetle detectability for the Sam's Point Preserve model (Fig 3.5A) and after 5% pebble cover for the historic range model (Fig. 3.6A).

Moss/lichen percent cover was the second most important variable to the Sam's Point Preserve habitat model (Fig. 3.4A). Mean percent cover of moss/lichen at *C. patruela* detected sites was $11.7\% \pm 14.9$ and $2.0\% \pm 3.0$ at non-detected plots ($p_{1.67, 53} = 1.76E-05$). Sand percent cover was the third most important variable to the model. Plots where *C. patruela* was detected (6.1 ± 16.5) had a greater percent cover of sand than non-detected plots ($0.07\% \pm 0.4$; $p_{1.67, 49} = 0.007$). Moss/lichen and sand percent cover had positive relationships with plots where *C. patruela* was detected (Fig. 3.5B and Fig. 3.5C). Sand cover became constant at 3% cover, while moss/lichen continued to increase, though at a much lower rate after ~ 4% cover and leveled off > 25% cover.

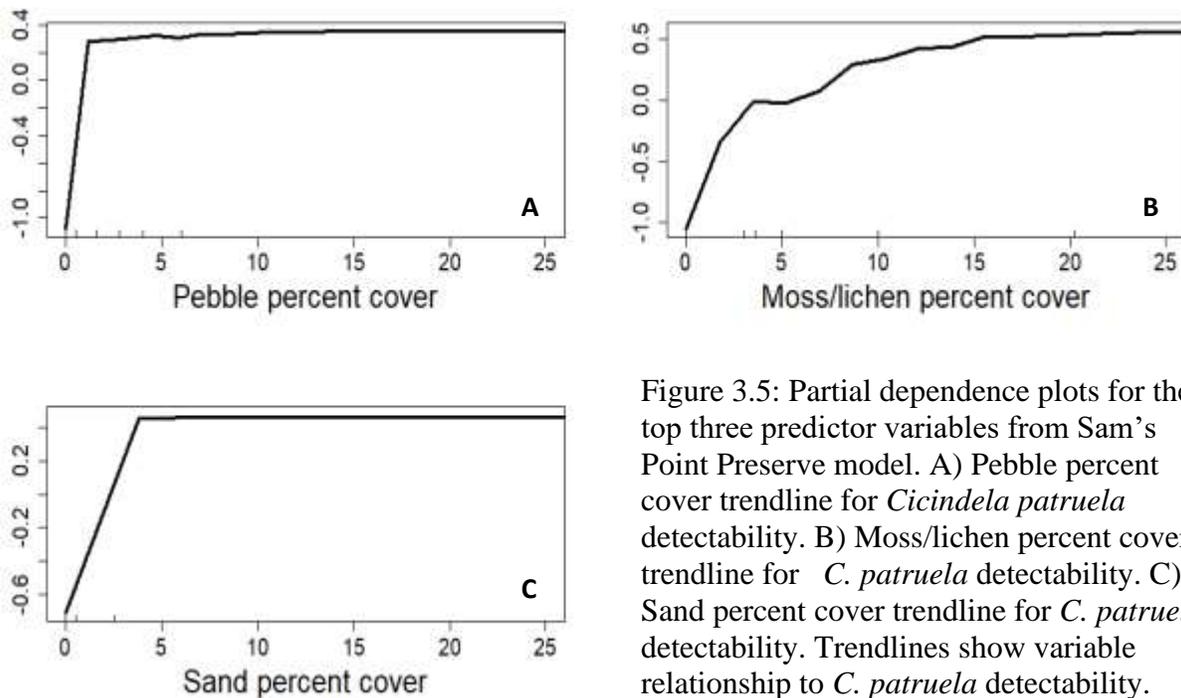


Figure 3.5: Partial dependence plots for the top three predictor variables from Sam's Point Preserve model. A) Pebble percent cover trendline for *Cicindela patruela* detectability. B) Moss/lichen percent cover trendline for *C. patruela* detectability. C) Sand percent cover trendline for *C. patruela* detectability. Trendlines show variable relationship to *C. patruela* detectability.

Pine/twig litter was the second most important variable to the historical range habitat model (Fig. 3.4B) with a mean percent cover of $1.5\% \pm 1.9$ in *C. patruela* detected plots and $8.9\% \pm 17.7$ in historical range *C. patruela* non-detected plots ($p_{1.69, 31} = 0.012$). Increased percent cover of pine/twig litter decreased the likelihood of *C. patruela* detectability (Fig. 3.6B). At 4% pine/twig litter, detection of *C. patruela* becomes unlikely. Road percent cover was the third most important variable. Roads present at sites were not paved and consisted of loose gravel. Plots where *C. patruela* were detected had a mean of $5.5\% \pm 18.2$ road cover while historic range non-detected sites had a mean of $24.9\% \pm 40.4$ ($p_{1.68, 39} = 0.007$). Road percent cover positively influenced *C. patruela* detection until $\sim 80\%$ cover was reached wherein the relationship turned strongly negative (Fig. 3.6C).

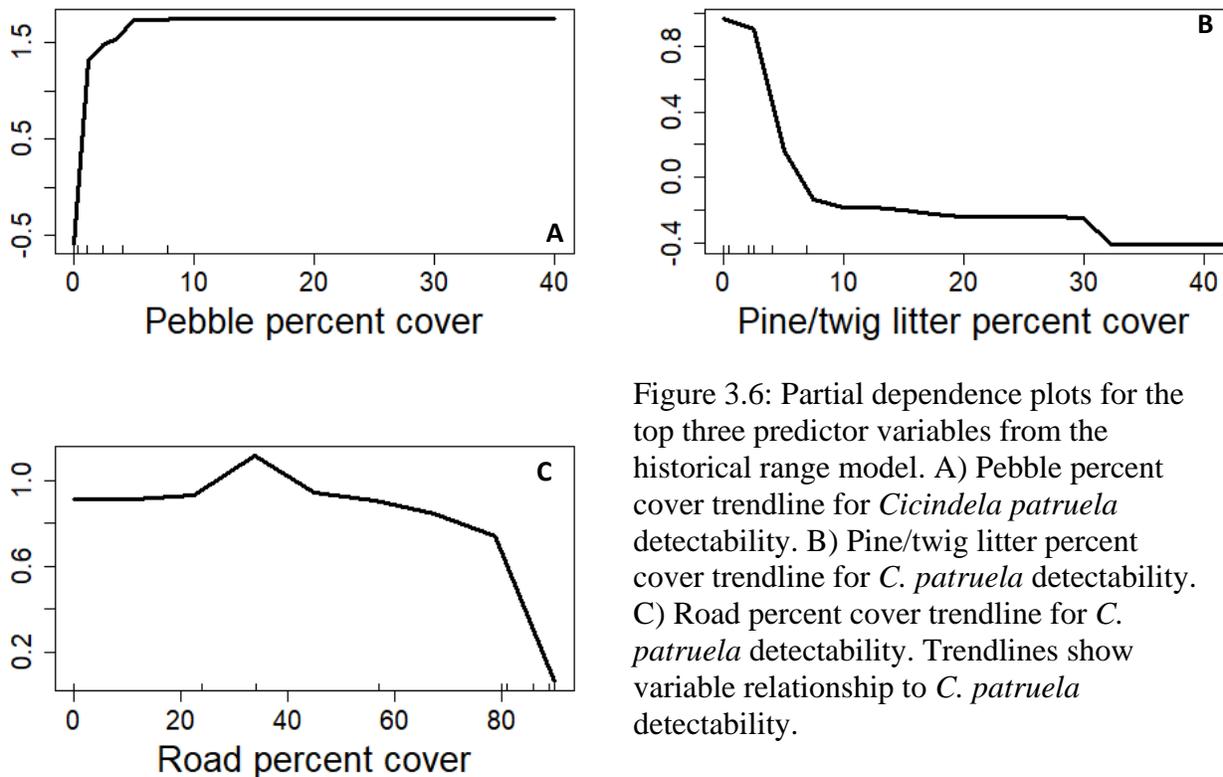


Figure 3.6: Partial dependence plots for the top three predictor variables from the historical range model. A) Pebble percent cover trendline for *Cicindela patruela* detectability. B) Pine/twig litter percent cover trendline for *C. patruela* detectability. C) Road percent cover trendline for *C. patruela* detectability. Trendlines show variable relationship to *C. patruela* detectability.

The top three predictor variables for *C. patruela* larval burrow habitat were presence of rock slab, cobble, and percent tree cover (Fig. 3.4C). Plots where larval burrows were detected had a mean rock slab percent cover of $8.9\% \pm 14.3$, tree percent cover of $6.6\% \pm 11.7$, and cobble cover of $8.0\% \pm 14.5$. Where larval burrows were not detected, mean rock slab cover was $19.1\% \pm 26.1$, tree cover $1.2\% \pm 5.1$, and cobble cover $4.1\% \pm 8.6$. All three variables were found to be significant (rock slab $p_{1.67, 47} = 0.009$; cobble $p_{1.67, 82} = 0.05$; tree $p_{1.67, 69} = 0.002$);). Low percent cover of rock slab resulted in detecting *C. patruela* larval burrows more often than in non-detected *C. patruela* plots until $\sim 5\%$ cover was rock slab. At 5% cover, the relationship between *C. patruela* larval burrow detection and rock slab percent cover became negative (Fig. 3.7A). Tree percent cover had a positive relationship with larval burrow detection (Fig. 3.7B). Cobble percent cover had a positive relationship until $\sim 15\%$ cover wherein the relationship turned negative, but became positive again at $\sim 27\%$ cover (Fig. 3.7C).

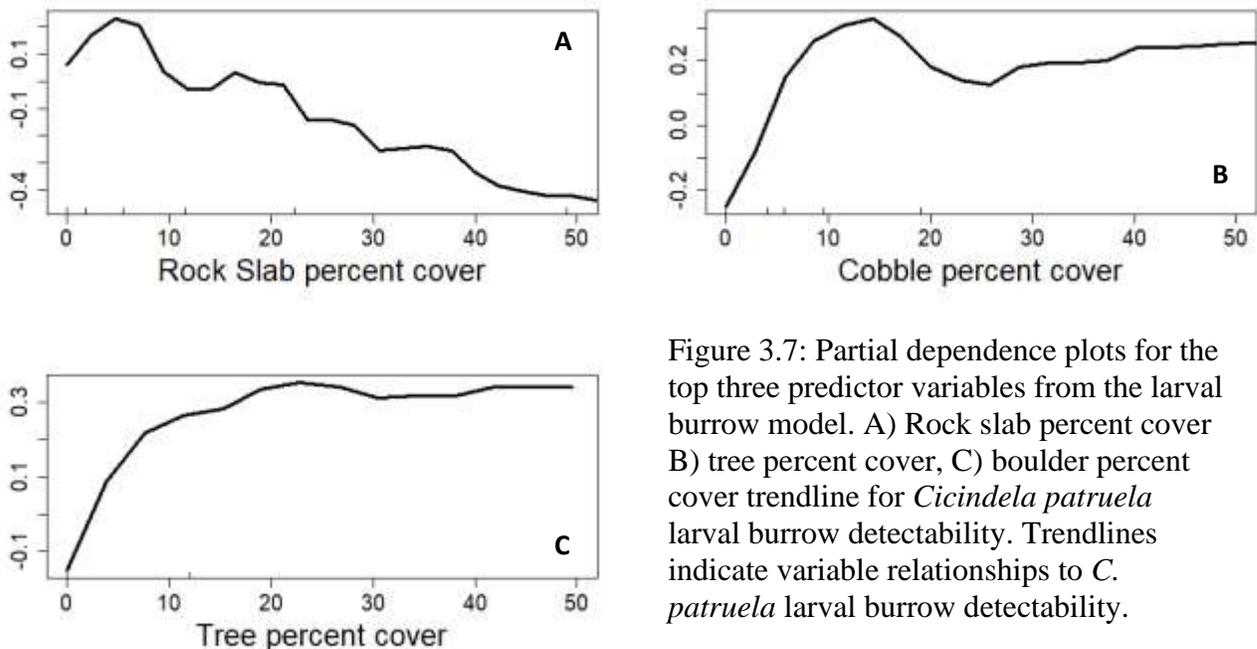


Figure 3.7: Partial dependence plots for the top three predictor variables from the larval burrow model. A) Rock slab percent cover B) tree percent cover, C) boulder percent cover trendline for *Cicindela patruela* larval burrow detectability. Trendlines indicate variable relationships to *C. patruela* larval burrow detectability.

DISCUSSION

Suitable habitat for *C. patruela* was driven by habitat variables rather than prey abundance and diversity. Similar to what Mawdsley (2005, 2007) found, results from the present study indicate adult *C. patruela* were more likely to occur in habitat patches with greater amounts of pebbles, moss/lichen, sand, and less pine/twig litter. Pebble percent cover was the most important variable associated with suitable adult *C. patruela* habitat, however, it was not one of the top three important variables in the larval habitat model. Anecdotal observations from 2019 indicated most larval burrows were in areas of higher average percent cover of pebbles (19.8 ± 23.6). Importance of this habitat is likely connected to female oviposition preference as Mawdsley (2007) found high pebble percent cover was critical to oviposition success. It is likely pebbles provide a looser substrate allowing females to insert their ovipositor to an appropriate depth for oviposition of eggs and larvae to construct a burrow. Increased rock slab and pine and twig litter decreased the likelihood of *C. patruela* selecting an area to oviposit, most likely due to a deeper duff layer that prevents female access to preferred oviposition substrate.

Moss/lichen cover was another important variable associated with suitable habitat for detecting *C. patruela* adults. This could be due to female selection of areas of high moss cover (Mawdsley 2007). Moss has high moisture retention, even in areas with reduced canopy (Tanskanen et al. 2006). Females of another tiger beetle, *C. hirticollis*, were shown to select areas of increased soil moisture, likely to keep larvae from desiccating and allowing for burrow formation (Cornelisse and Hafernik 2009). Although moss/lichen percent cover was not indicated as important in the larval burrow habitat model, anecdotal observations from 2019 indicated most larval burrows were in areas of higher percent cover of moss. One

opportunistically found area (3×2 m) had 10 larval burrows with 90% moss cover (Fig. 3.8) and another area (6×3 m) had 6 larval burrows with 82% moss cover.



Figure 3.8: A) Larval habitat with burrows present. B) Close up of two larval burrows with white arrows indicating burrows.

Larval burrows were detected in areas of with reduced rock slab cover and high dwarf pitch pine and cobble percent cover (Fig. 3.7). Presence of rock slab mechanically prevents oviposition, thus it is not surprising it had a negative relationship with detection of larval burrows. Larval burrows were observed around edges of open areas near pines and shrubs (i.e. mountain and sheep laurel, blueberry, and huckleberry) and so pine percent cover being a significant variable for where females would select to oviposit also made sense. Although pine cover had a positive relationship to larval burrow detection (Fig. 3.7B), too much vegetation cover can result in decreased survival of larvae (Knisley 2010). Shading from vegetation can prevent desiccation and possibly provide larvae with cover from predators (Cornelisse and Hafernik 2009). Cobbles serve a similar purpose as tree cover, providing cover and shade throughout the day.

Knisley and Hill (1992) found loss of suitable habitat affects the distribution and abundance of rare or threatened tiger beetle populations. Loss of suitable habitat of *Cicindela tranquebarica* Herbst, *C. sexgutta* Fabricius, and *C. repanda* Dejean in Virginia resulted in population declines, mainly due to increased foot and vehicular traffic (Knisley and Hill 1992). Sam's Point Preserve does not allow the public to drive vehicles through the park, and *C. patruela* are found along ridges, which are off marked foot trails. As road cover exceeded 80%, *C. patruela* detection was less common, even though the preserve doesn't allow vehicles (Fig. 3.6C). Beetles were observed on a closed trail in 2017, but when the trail reopened in 2018, individuals were not detected that year, nor in 2019. Anecdotal observations indicated the trail substrate was much more compact than in 2017 when *C. patruela* was detected. Cornelisse and Hafernik (2009) documented compaction of preferred substrate deterred *C. hirticollis* females from ovipositing.

Fire suppression is likely a critical driver of habitat loss for *C. patruela* (New York Natural Heritage Program 2017). Active suppression of wildfire throughout this region over a half century or more has resulted in encroachment of vegetation into open areas and buildup of a thick duff layer. My research indicates increased percent cover of pine/twig litter decreased likelihood of *C. patruela* occupancy (Fig. 3.7B). Areas within the historical range of *C. patruela* outside of Sam's Point Preserve had a significantly higher percent cover of pine/twig litter than sites within the preserve where individuals were detected. Incorporation of prescribed fire reduces shrub and pine/twig litter cover and increases moss cover (Bernard 1963) and so should be considered in management for this particular beetle.

Models did not show either prey abundance or diversity as important compared to vegetation and substrate percent cover, although other studies have found tiger beetles can be

limited by prey (Meyer 1987; Bauer 1991; Pearson and Knisley 1985). Although I did not look at abundance and diversity of other prey, ants were the only apparent prey items for this species.

It was curious that *Crematogaster* (2–4 mm), *Dolichoderus* (~ 4 mm), and *Formicoxenus* (2.5–3.5 mm) were not documented in plots with adult beetles. *Formicoxenus* and *Crematogaster* are in the subfamily Myrmicinae and *Dolichoderus* is in the subfamily Dolichoderinae. Some species within Myrmicinae specialize on Collembola (Dejean 1983), which are also a prey item of larval tiger beetles. It may be that adult tiger beetles search for areas where ants in Myrmicinae are not present to reduce competition with their larvae. Species within Myrmicinae have well developed stings, while species in Dolichoderinae lack stings. Instead of a sting Dolichoderinae, uses pungent odors from an anal gland to ward of predators (Welzel et al. 2018). Another possible reason for their not being seen is if they are preferential prey, so that adult tiger beetle presence results in reduced numbers, and/or these ants are exhibiting predator avoidance behaviors.

Ants in the genera *Camponotus* (7–13 mm), *Formica* (4–8 mm), and *Lasius* (5–7 mm) were present at positive *C. patruela* adult plots. These genera of ants share the same subfamily of Formicinae, and these ants have an acidopore (a structure at the end of their seventh abdominal segment that can spray formic acid in defense at their enemies), but have no functional sting (Ward et al. 2016). Ant genera present in areas where adult *C. patruela* was detected are larger than those in areas where *C. patruela* was not detected. Beetles could



Figure 3.9: *C. patruela* individual consuming *Camponotus* sp.

select areas with larger prey items due to increased visibility of the larger prey and increased nutrient intake per prey item for less effort (Fig. 3.9).

CONCLUSIONS

This population of the rare northern barrens tiger beetle, *C. patruela*, is likely limited by habitat variables related to female oviposition behavior. Possible habitat characteristics causing extirpation of *C. patruela* from areas outside of Sam's Point Preserve are decreased cover of pebbles and increased cover of pine/twig litter. Variables potentially keeping *C. patruela* from dispersing to similar nearby areas within Sam's Point Preserve are lack of pebble, moss/lichen, and sand cover. Although there should be plenty of suitable area throughout Minnewaska State Park, the model was for dwarf pitch pine cover only and did not take into account other biotic, e.g., prey availability, predators, and abiotic variables, e.g., moisture, slope, water bodies. These areas could be shown as scores of 3 in Figure 3.2 (i.e. suitable habitat) due to their lack of pine cover but are still not suitable for *C. patruela*.

To ensure this population does not become extirpated, management actions should be taken to decrease pine/twig litter to expose and increase pebble cover. To increase the likelihood of successful dispersal to neighboring areas, Sam's Point Preserve could partner with surrounding areas (e.g., Mohonk, Minnewaska Park) to take management actions to preserve or restore pebble, moss/lichen, and sand cover by creating/supporting natural openings. Patron foot traffic should also avoid areas where there are known larval burrows to decrease the risk of soil compaction. Areas of interest for new trail construction should be scouted to ensure they are not located in *C. patruela* habitat.

CHAPTER 4

CONCLUSIONS AND FURTHER RESEARCH

CONCLUSIONS

Northern barrens tiger beetles, *Cicindela patruela patruela* Dejean (Coleoptera: Carabidae), are one of the 22 rare and at-risk tiger beetle species recognized in New York (Schlesinger and Novak 2011). *Cicindela patruela* has one known population at Sam's Point Preserve. Habitat destruction, modification, and alteration of natural process are key factors threatening this species of tiger beetle. Long-term management programs need to be developed in order to protect this rare population and its associated habitat.

Reliable and accurate monitoring of the abundance of rare species in space and time is critical to determine their possible extinction risk. One method used to estimate population abundance is via mark-recapture surveys. Mark-recapture surveys can be utilized to track small populations by monitoring changes in abundance over survey years (Mackenzie et al. 2004). Occupancy is an alternative to mark-recapture that requires less effort and money to monitor rare/endangered populations (Mackenzie and Bailey 2004). This method, at an appropriate scale, is positively correlated with abundance; i.e. occupancy increases with increasing abundance.

Decades of fire suppression in the Shawangunk mountains has limited creation of natural openings and pitch-pine forest regeneration (Schlesinger and Novak 2011, Beers et al. 2011). As vegetation encroaches on the natural openings northern barrens tiger beetles rely on, area of suitable habitat decreases for the beetles (Mawdsley 2005). Oviposition site choice is one of the main drivers of tiger beetle habitat selection (Cornelisse and Hafernik 2009). Females typically choose to oviposit in shaded (Knisley et al. 1990) sandy soils with small pebbles (< 2 cm diameter) where mosses, lichens, and sedges are present (Mawdsley 2005, 2007), but not dominant as females will not oviposit in areas with high vegetation cover — and if vegetation

invades the larval burrow, larvae disappear (Knisley 2010). The northern barrens tiger beetle needs a management plan to protect and promote natural openings and native plant assemblages for this rare species to continue to persist.

My research took place at Sam's Point Preserve in the Hudson Valley, NY with the goal of determining abundance and population ecology of *C. patruela*, and what habitat/prey variables are selected for by these beetles. Mark-recapture surveys were conducted from May through July for the 2017 and 2018 seasons and from May to June for the 2019 season. Habitat surveys were conducted from Jul–Aug, 2017 and prey abundance surveys were conducted from May–Jun, 2019. Mark-recapture and prey abundance surveys were weather dependent. Mark-recapture data were used for occupancy estimates.

Mark-recapture population estimates increased from ~ 81 individuals in 2018 to ~ 108 individuals in 2019. Colonization probability remained constant among the years 2017, 2018, and 2019, while the extinction probability decreased from 2017–2018 (0.64 ± 0.06) to 2018–2019 (0.33 ± 0.11). Probability of detection also increased each year from 2017 until 2019 (0.05 ± 0.01 , 0.17 ± 0.02 , 0.23 ± 0.03). Increased probability of detection could be a result of surveying for three consecutive years and gaining a search image allowing for quicker detection of *C. patruela* than in my first survey season. Two new populations were discovered in 2019, but were not included in analyses. Continued monitoring of *C. patruela* is necessary to be able to develop a management plan to conserve this small population.

The Sam's Point Preserve model indicated increased amounts of pebble, moss/lichen, and sand are selected for by *C. patruela* adults. The historical range habitat model showed increased amounts of pebble and decreased amounts of pine/twig litter and roads are selected by *C. patruela* adults. The larval burrow model indicated increased cover of dwarf pitch pine

ground cover and cobbles and decreased cover of rock slab was selected by ovipositing *C. patruela* females. It is possible that extirpation of *C. patruela* from locations outside of Sam's Point Preserve was due to increase in pine/twig litter and encroachment of vegetation into open pebbly areas. This could be the result of fire suppression in the Hudson Valley, allowing a buildup of pine and twig litter over pebbles that *C. patruela* selects for.

LIMITATIONS AND FUTURE RESEARCH

Due to *Cicindela patruela* being active only on sunny days $> 19^{\circ}\text{C}$, surveys were limited to days with ideal weather conditions. Years 2017 and 2019 had wet springs that resulted in the study location being flooded, which then resulted in a relatively small number of survey days that could be used in analyses. The extra wet season may have also contributed to fading of marks in the mark-recapture study, which meant that only two years, instead of three, could be used in mark-recapture analyses. The 2017 marking technique was used because of success Hudgins et al. (2011) had throughout their study. It is possible that my field seasons being during the wet spring rather than during the drier summer season, like Hudgins et al. (2011), could have resulted in the mixed success using the same technique.

Sam's Point Preserve is a large park (20.2 km²), which made it impossible to have equal search effort in every possible suitable *C. patruela* habitat. Surveys were thus conducted opportunistically, when weather was ideal. This limited the time spent on each survey as surveys ended in July. Due to this, survey efforts were in two main locations where *C. patruela* was known to be and though we increased the search area to surrounding habitat, we never ventured too far away. Areas of what seemed like suitable habitat, but where no individuals were known to be detected were not searched as often. By chance in 2019, two new subpopulations were found in locations that were previously only lightly searched.

My research did not focus on genetics or relocation of *C. patruela*, but provided information on new subpopulations and habitat requirements that future research can explore. Now that there is a baseline of population numbers and occupancy estimates, along with locating two new subpopulations, continued monitoring and increasing search areas could be undertaken. Additionally, molecular evaluations of all populations could yield interesting results, especially if combined with evaluations of the next closest population of *C. patruela* in New Jersey.

My research resulted in models indicating habitat variables important to, and selected by, *C. patruela* and can be used to search for other locations in New York with suitable habitat. The Albany Pine Bush is within the *C. patruela* historical range and is a strong contender for re-establishing individuals from the Sam's Point Preserve population. Albany Pine Bush has been managing its pitch pine barrens landscape using prescribed fire and hardwood thinning since 1991 (Lee et al. 2019). This has created natural openings, which could yield suitable habitat for *C. patruela*, and thus might be able to support a new population. By re-establishing a population of *C. patruela* in this area, managers will be able to assess if fire management contributes to *C. patruela* establishment and long term abundance and provide insights as to whether or not prescribed fire should be established at Sam's Point Preserve to help manage, and potentially grow, that population.

LITERATURE CITED

- Bauer, K. L. 1991. Observations on the developmental biology of *Cicindela arenicola* Rumpff (Coleoptera: Cicindelidae). *The Great Basin Naturalist* 51:226–23.
- Beers, A., J. Hall, W. Janeway, and T. Dooley. 2011. Northern Shawangunk Ridge fire management plan. Interagency fire management plan for Minnewaska State Park Preserve, Sam's Point Preserve, Mohonk Preserve and Witch's Hole State Forest, New Paltz, New York.
- Bernard, J. M. 1963. Forest floor moisture capacity of the New Jersey Pine. *Ecology* 44:574–576.
- Bijlsma, R., J. Bundgaard, and A. C. Boerema. 2000. Does inbreeding affect the extinction risk of small populations?: predictions from *Drosophila*. *J of Evolutionary Biology* 13:502–514.
- Breiman, L. 2001. Random Forests.
- Bried, J. T. and N. A. Gifford. 2010. Mowing and herbicide of scrub oaks in pine barrens: Baseline data (New York). *Ecological Restoration* 28(3): 245–248.
- Bried, J. T. and J. Pellet. 2011. Optimal design of butterfly occupancy surveys and testing if occupancy converts to abundance for sparse populations. *Journal of Insect Conservation* 16(4): 489–499.
- Brown, J. K. and J. K. Smith. 2000. Wildland fire in ecosystems: effects of fire on flora. US Department of Agriculture, Forest Service, Rocky Mountain Research Station, General Technical Report RMRS-GTR-42 (2): 1–257.
- Canada, E. 2016. Species at risk act recovery strategy series recovery strategy for the Northern barrens tiger beetle (*Cicindela patruela*) in Canada. Ottawa.
https://www.sararegistry.gc.ca/virtual_sara/files/plans/rs_northern_barrens_tiger_beetle_e_proposed.pdf
- Canada, I. 2009. COSEWIC assessment and status report Northern barrens tiger beetle *Cicindela patruela*.
https://www.sararegistry.gc.ca/virtual_sara/files/plans/rs_northern_barrens_tiger_beetle_e_proposed.pdf
- Cassola, F., and D.L. Pearson. 2000. Global patterns of tiger beetle species richness (Coleoptera: Cicindelidae): their use in conservation planning. *Biological Conservation* 95:197–208.

- Chiari, S., A. Zauli, A. Mazziotta, L. Luiselli, P. Audisio, and G.M. Carpaneto. 2013. Surveying an endangered saproxylic beetle, *Osmoderma eremita*, in Mediterranean woodlands: a comparison between different capture methods. *J of Insect Conservation* 17:171–181.
- Cornelisse, T.M., and J.E. Hafernik. 2009. Effects of soil characteristics and human disturbance on tiger beetle oviposition. *Ecological entomology* 34:495–503.
- Cornelisse, T.M., M.C. Vasey, K.D. Holl, and D.K. Letourneau. 2012. Artificial bare patches increase habitat for the endangered Ohlone tiger beetle (*Cicindela ohlone*). *Journal of Insect Conservation* 17(1): 17–22.
- Cornelisse, T.M., M.K. Bennett, and D.K. Letourneau. 2013. The implications of habitat management on the population viability of the endangered Ohlone tiger beetle (*Cicindela ohlone*) metapopulation. *PloS one* 8(8): e71005
- Cutler, D.R., T.C. Edwards, K.H. Beard, A. Cutler, K.T. Hess, J. Gibson, and J.J. Lawler. 2007. Random forests for classification in ecology. *Ecology* 88:2783–2792.
- Den Boer, P.J. 1990. The survival value of dispersal in terrestrial arthropods. *Biological Conservation* 54:175–192.
- Dibner, R.R., D.F. Doak, and M. Murphy. 2017. Discrepancies in occupancy and abundance approaches to identifying and protecting habitat for an at-risk species. *Ecology and Evolution* 7: 5692–5702.
- Dieckmann, U., B. O’Hara, and W. Weisser. 1999. The evolutionary ecology of dispersal. *Trends in Ecology & Evolution* 14(3): 88–90.
- DiTomaso, J.M., M.L. Brooks, E.B. Allen, R. Minnich, P.M. Rice, and G.B. Kyser. 2006. Control of invasive weeds with prescribed burning. *Weed technology* 20:535–548.
- Dreisig, H. 1981. The rate of predation and its temperature dependence in a tiger beetle, *Cicindela hybrida*. *Oikos* 36:196–202.
- Dreisig, H. 1980. Daily activity, thermoregulation and water loss in the tiger beetle *Cicindela hybrida*. *Oecologia* 44:376–389.
- Driscoll, D.A., D.B. Lindenmayer, A.F. Bennett, M. Bode, R.A. Bradstock, G.J. Cary, M.F. Clarke, N. Dexter, R. Fensham, G. Friend, and M. Gill. 2010. Fire management for biodiversity conservation: Key research questions and our capacity to answer them. *Biological Conservation* 143(9):1928–1939.
- Van Dyck, H., and M. Baguette. 2005. Dispersal behavior in fragmented landscapes: Routine or

- special movements? *Basic and Applied Ecology* 6:535–545.
- Van Dyck, H., and E. Matthysen. 1999. Habitat fragmentation and insect flight: a changing ‘design’ in a changing landscape? *Trends in Ecology & Evolution* 14:172–174.
- Ellison, A.M., N.J. Gotelli, E.J. Farnsworth, and G.D. Alpert. 2012. A field guide to the ants of New England. Yale University Press, New Haven, Connecticut, USA.
- Forman, R.T.T., and R.E. Boerner. 1981. Fire frequency and the pine barrens of New Jersey. *Bulletin of the Torrey Botanical Club* 108:34–50.
- Gadgil, M. 1971. Dispersal: Population consequences and evolution. *Ecology* 52:253–261.
- Gaston, K.J., T.M. Blackburn, J.J. Greenwood, R.D. Gregory, R.M. Quinn, and J. H Lawton. 2000. Abundance-occupancy relationships. *J of Applied Ecology* 37: 39–59.
- Givnish, T.J. 1981. Serotiny, geography, and fire in the pine barrens of New Jersey. *Evolution* 35:101–123.
- Gu, W., and S.K. Swihart. 2004. Absent or undetected? Effects of non-detection of species occurrence on wildlife-habitat models. *Biological Conservation* 116:195–203.
- Haddad, N.M., B. Hudgens, C. Damiani, K. Gross, D. Kuefler, and K. Pollock. 2008. Determining optimal population monitoring for rare butterflies. *Conservation Biology* 22(4): 929–940.
- Hagler, J.R., and C.G. Jackson. 2001. Methods for marking insects: Current techniques and future prospects. *Annu. Rev. Entomol* 46:511–543.
- Harvey, A., and S. Zukoff. 2011. Wind-powered wheel locomotion, initiated by leaping somersaults, in larvae of the southeaster Beach Tiger Beetle (*Cicindela doralis media*). *PloS one* 6(3): e17746.
- Hudgins, R.M., C. Norment, M.D. Schlesinger, and P.G. Novak. 2011. Habitat selection and dispersal of the cobblestone tiger beetle (*Cicindela marginipennis* Dejean) along the Genesee River, New York. *American Midland Naturalist* 165:304–318.
- Knisley, B., and S.A. Juliano. 1988. Development, and size of larval tiger beetles: effects of food and water. *Ecology* 69:1983–1992.
- Knisley, B.C. 1987. Habitats, food resources, and natural enemies of a community of larval *Cicindela* in southeastern Arizona (Coleoptera: Cicindelidae). *Canadian J Zoology* 65:1191–1200.

- Knisley, C.B. 2010. Anthropogenic disturbances and rare tiger beetle habitats: benefits, risks, and implications for conservation. *Terrestrial Arthropod Reviews* 4:41–61.
- Knisley, C.B., and J.M. Hill. 1992. Effects of habitat change from ecological succession and human impact on tiger beetles. *Virginia J Science* 43:134–142.
- Knisley, C.B., T.D. Schultz, and T.H. Hasewinkel. 1990. Seasonal activity and thermoregulatory behavior of *Cicindela patruela* (Coleoptera: Cicindelidae). *Annals of the Entomological Society of America* 83:911–915.
- Knisley, C.B., M. Drummond, and J. McCann. 2016. Population trends of the Northeastern beach tiger beetle, *Cicindela dorsalis dorsalis* Say (Coleoptera: Carabidae: Cicindelinae) in Virginia and Maryland, 1980s through 2014. *The Coleopterists Bulletin* 70(2): 255–271.
- Lee, C.S., G.R. Robinson, I.P. Robinson, and H. Lee. 2019. Regeneration of pitch pine (*Pinus rigida*) stands inhibited by fire suppression in Albany Pine Bush Preserve, New York. *J Forestry Research* 30(1): 233–242
- Lettink, M., and D.P. Armstrong. 2003. An introduction to using mark-recapture analysis for monitoring threatened species. Department of conservation technical series 28:5–32.
- Liaw, A., and M. Wiener. 2002. Classification and regression by randomForest. *R news* 2:18–22.
- Lindenmayer, D.B., and H.P. Possingham. 1996. Modelling the inter-relationships between habitat patchiness, dispersal capability and metapopulation persistence of the endangered species, Leadbeater’s possum, in south-eastern Australia. *Landscape Ecology* 11:79–105.
- Mackenzie, D.I., and L.L. Bailey. 2004. Assessing the fit of site-occupancy models. *J of Agricultural, Biological, and Environmental Statistics* 9:300–318.
- Mackenzie, D.I., D.I. Mackenzie, and J.D. Nichols. 2004. Occupancy as a surrogate for abundance estimation. *Animal Biodiversity and Conservation* 27:461–467.
- Mawdsley, J.R. 2005. Extirpation of a population of *Cicindela patruela* Dejean (Coleoptera: Carabidae: Cicindelini) in suburban Washington, D.C., USA. *Proc. Entomol. Soc. Wash.* 107:64–70.
- MacKenzie, D.I., J.D. Nichols, G.B. Lachman, S. Droege, J.A. Royle, and C.A. Langtimm. 2002. Estimating site occupancy rates when detection probabilities are less than one. *Ecology* 83:2248–2255.

- MacKenzie, D.I., J.D. Nichols, J.A. Royle, K.H. Pollock, L.L. Bailey, and J.E. Hines. 2017. Occupancy estimation and modeling : inferring patterns and dynamics of species occurrence. Elsevier.
- Mawdsley, J.R. 2005. Extirpation of a population of *Cicindela patruela* Dejean (Coleoptera: Carabidae: Cicindelini) in suburban Washington, D.C., USA. Proc. Entomol. Soc. Wash. 107:64–70.
- Mawdsley, J.R. 2007. Ecology, distribution, and conservation biology of the tiger beetle *Cicindela patruela consentanea* Dejean (Coleoptera: Carabidae: Cicindelinae). Proc. Entomol. Soc. Wash. 109:17–28.
- Meyer, E.J.M. 1987. Asymmetric resource use in two syntopic species of larval tiger beetles (Cicindelidae). Oikos 50:167–175.
- Motzkin, G., D.A. Orwig, and D.R. Foster. 2002. Vegetation and disturbance history of a rare dwarf pitch pine community in western New England, USA. J Biogeography 29:1455–1467.
- Motzkin, G., W.A. Patterson III, and D.R. Foster. 1999. A historical perspective on pitch pine-scrub oak communities in the Connecticut Valley of Massachusetts. Ecosystems 2:255–273.
- Murphy, P.A., and G.J. Nowacki. 1997. An old growth definition for xeric pine and pine-oak woodlands. US Department of Agriculture, Forest Service, Southern Research Station, General Technical Report SRS-7:1–11.
- New York Natural Heritage Program. 2017. Online conservation guide for *Cicindela patruela patruela*. <http://acris.nynhp.org/guide.php?id=37144>.
- Osborne, J.L., H.D. Loxdale, and I.P. Woiwod. 2002. Monitoring insect dispersal: methods and approaches. Dispersal Ecology:24–49.
- Pearson, D., and F. Cassola. 1992. World-wide species richness patterns of tiger beetles (Coleoptera: Cicindelidae): Indicator taxon for biodiversity and conservation studies. Conservation Biology 6:376–391.
- Pearson, D.L., and C. B. Knisley. 1985. Evidence for food as a limiting resource in the life cycle of tiger beetles (Coleoptera: Cicindelidae). Nordic Society Oikos 45:161–168.
- Pearson, D.L., and E.J. Mury. 1979. Character divergence and convergence among tiger beetles (Coleoptera: Cicindelidae). Ecology 60:557–566.
- Pearson, D., and A. Vogler. 2001. Tiger Beetles: The evolution, ecology, and diversity of the Cicindelids. Cornell University Press, Cornell.

- Ranius, T., and J. Hedin. 2001. The dispersal rate of a beetle, *Osmoderma eremita*, living in a tree hollow. *Oecologia* 126:363–370.
- Rodriguez-Galiano, V.F., B. Ghimire, J. Rogan, M. Chica-Olmo, and J.P. Rigol-Sanchez. 2012. An assessment of the effectiveness of a random forest classifier for land-cover classification. *ISPRS Journal of Photogrammetry and Remote Sensing* 67:93–104.
- Seischab, F.K. and J.M. Bernard. 1996. Pitch pine (*Pinus rigida* Mill.) communities in the Hudson Valley region of New York. *American Midland Naturalist* 1: 42–56.
- Schlesinger, M.D., and P.G. Novak. 2011. Status and conservation of an imperiled tiger beetle fauna in New York State, USA. *Journal of Insect Conservation* 15:839–852.
- Shelford, V.E. 1908. Life-histories and larval habits of the tiger beetles (Cicindelidae). *J of the Linnean Society of London, Zoology* 30:157–184.
- Tscharntke, T., I. Steffan-Dewenter, A. Kruess, and C. Thies. 2002. Characteristics of insect populations on habitat fragments: A mini review. *Ecological Research* 17:229–239.
- Tanskanen, H., A. Granström, A. Venäläinen, and P. Puttonen. 2006. Moisture dynamics of moss-dominated surface fuel in relation to the structure of *Picea abies* and *Pinus sylvestris* stands. *Forest Ecology and Management* 226:189–198.
- Ward, P.S., B.B. Blaimer, and B.L. Fisher. 2016. A revised phylogenetic classification of the ant subfamily Formicinae (Hymenoptera: Formicidae), with resurrection of the genera *Colobopsis* and *Dinomyrmex*. *Zootaxa*, 4072(3), 343–357.
- Welzel, K.F., S.H. Lee, A.T. Dossey, K.R. Chauhan, and D.H. Choe. 2018. Verification of Argentine ant defensive compounds and their behavioral effects on heterospecific competitors and conspecific nestmates. *Scientific reports* 8.1:1477.
- White, G.C. and K.P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. *Bird Study* 46:120–139.
- USGS. 2005. Occupancy Models to Study Wildlife. www.cnr.colostate.edu/~gwhite/soft-

McKenzie D. Wybron

585-443-5257 mckenziwybron@gmail.com

Education

Expected December 2019 – State University of New York, College of Environmental Science and Forestry, Syracuse, NY

Master of Science in Ecology; GPA: 3.65

December 2017 – The College at Brockport-State University of New York, Brockport, NY

Bachelor of Science in Environmental Science and Ecology; Chemistry minor; GPA: 3.65

Professional Experience

September 2019-Present – Natural Resources Technician, Fort Drum, Watertown, NY

- Conducted pollinator surveys along roadside in flowering meadows
 - Collected and preserved pollinators captured during targeted sweep netting
 - Identified pollinators to genus
- Chemically and mechanically controlled invasive plant species
- Maintained and repaired equipment and heavy machinery

July 2019–August 2019 – Native Pollinator Research Assistant, Fort Drum, Watertown, NY

- Conducted pollinator surveys across wetlands, meadows, roadsides, and forests
 - Collected and preserved pollinators captured during targeted sweep netting
 - Placed bowl traps along transects in alternating color order
- Worked near military equipment and firing ranges

May 2017–August 2018 – Natural Heritage Program Research, Sam’s Point Preserve, Cragmoor, NY

- Researched habitat, prey, and population dynamics of the rare northern barrens tiger beetle in pitch pine heath
 - Conducted mark-recapture, occupancy, vegetation, and prey abundance surveys
 - Supervised one technician
- Conducted public outreach about the pitch pine ecosystem
- Presented oral and poster presentation at Northeastern Natural History Conference, 2019

Aug 2018–May 2019 – Emerald Ash Borer Research Assistant, College of Environmental Science and Forestry, Syracuse, NY

- Helped to control invasive species using biocontrol
- Recorded canopy cover, woodpecker damage, and basal sprouts to assess ash tree health
- Conducted emerald ash borer surveys by mechanical removal and examination of ash tree branches
 - Scraped and peeled bark of branches to quantify emerald ash borer infestation and parasitoid establishment in Syracuse

Jan 2018–Apr 2018 – Ecological Restoration Crew Georgia DNR, Mauk, GA

- Worked as part of a prescribed burn/ wildland fire crew for ecological restoration and habitat improvement
 - Operated/maintained UTVs, ATVs, type 7 fire engine, and hydraulic and 4 stroke engine pumps
 - Worked in multiple fuel types (old growth montane longleaf pine, restored grasslands, sandhills, and bogs)

- Prepared/marked timber sales using aerosol cans and backpack sprayers
- Monitored and surveyed endangered species using drift fences and other methodologies
- Developed maps of boundaries, roads and other features

May 2016–Aug 2016 – Undergraduate Research, The College at Brockport, Brockport, NY

- Researched possible methods of biocontrol for terrestrial invasive species
 - Determined if goat herbivory reduced invasive species biomass in the short-term and long-term
 - Mechanically removed and chemically treated invasive species
- Identified native and invasive plants using various resources and field guides including dichotomous key
- Poster presentation at Northeastern Natural History Conference, 2017

Aug 2015–Dec 2017 – Research Assistant, The College at Brockport, Brockport, NY

- Hiked long distances, often over 10 miles through backcountry scouting for invasive species
- Entered field collected data into excel data sheets for future analyses
- Trapped small mammals, weighed, sexed, and marked them with an ear tag
- Updated, sorted, and analyzed data gathered for the Brockport herbarium
- Quantified native plant growth with multiple methods

Volunteer Experience

2018–Present – Albany Pine Bush Fire Crew, Albany, NY

2017–2018 – New York State Hemlock Initiative, Cornell University, Ithaca, NY

2016–2018 – Finger Lakes Trail Maintenance, Allegany County, NY

2015–2017 – New York State Department of Environmental Conservation Raptor Survey, Avon, NY

Skills and certifications

R-Studio

GIS: ArcMap

Program Mark

Program Presence

Program Distance

ATV Rider Course

Hunter Safety Course (rifle and archery)

Fire Fighter Type 2 Certified- Red Card Eligible

Sawyer Level 1 Wildland Fire Chainsaw S-212 ASI