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Bog Turtle (Glyptemys muhlenbergii) Population Dynamics and Response to Habitat Management in Massachusetts

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BOG TURTLE (*Glyptemys muhlenbergii*) POPULATION DYNAMICS AND RESPONSE TO HABITAT MANAGEMENT IN MASSACHUSETTS

A Thesis Presented

by

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ABSTRACT

BOG TURTLE (*Glyptemys muhlenbergii*) POPULATION DYNAMICS AND RESPONSE TO HABITAT MANAGEMENT IN MASSACHUSETTS

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The Bog Turtle (*Glyptemys muhlenbergii*) is a federally threatened species that occupies isolated pockets of open-canopy fens. This long-lived species is susceptible to habitat loss and degradation; thus, the conservation of known populations and management of their habitat is critical to the species' survival. Long-term (multi-decadal) assessment is important for determining population trends and responses to ongoing habitat management. I assessed population demographics (abundance, survival) and spatial distribution (home range) of two Bog Turtle populations in Massachusetts that have been managed since the late 1990s by treating invasive species, thinning woody vegetation, and mitigating flooding. The results of this study were compared to two previous studies conducted in 1994–1997 and 2005–2009 to evaluate the response to habitat management. Estimates of adult population abundance increased from the first study period (Site 1 $\bar{X} = 37.3 \pm 10.4$, Site 2 $\bar{X} = 36.2 \pm 3.2$) to the last study period (Site 1 $\bar{X} = 65.1 \pm 17.9$, Site 2 $\bar{X} = 42.5 \pm 10.9$) across both sites. Estimates of annual survival across all study periods remained above 90% at Site 1 and were 100% for two years at

Site 2. I constructed 95% minimum convex polygon (MCP) and 95% kernel density estimation (KDE) home ranges for 71 turtles. At Site 1 there was no significant influence of the study period on home range estimates. The increase in abundance estimates, high survival, and stable home range sizes at Site 1 suggest that ongoing management has maintained quality habitat. At Site 2, the average home range size decreased by approximately half after the first study period in response to flooding but increased in the current study. Fluctuations in population abundance, and home range size at Site 2 throughout the study period reflect the cycles of habitat degradation and habitat management. My results indicate that habitat management efforts implemented since the late 1990s have provided quality habitat for the two Bog Turtle populations in Massachusetts while also mitigating long-term negative impacts on the populations. This further supports the need for long-term analysis of Bog Turtle populations, especially at sites where active habitat management is occurring.

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CHAPTER 1

INTRODUCTION

1.1 Biodiversity Declines

We are currently experiencing global climate and biodiversity crises (Butchart et al. 2010, IPBES 2019, WWF 2022, IPCC 2023). The rate at which species are going extinct is at least tens to hundreds of times higher than it has been over the past 10 million years (IPBES 2019). Nearly 2.5% of reptiles, amphibians, mammals, fish, and birds have already gone extinct (WWF 2022). Approximately one million plants and animals currently face extinction (IPBES 2019, WWF 2022). Half a million terrestrial species of animals and plants are threatened with extinction due to habitat loss and degradation (IPBES 2019).

Turtles and tortoises (chelonians) are a particularly vulnerable group. In the past 280 years, seven species of turtles and tortoises have gone extinct (Rhodin et al. 2018, Stanford et al. 2020). More than half of 360 living species and 482 taxa (species and subspecies combined) of chelonians are nearing extinction (Rhodin et al. 2018, Stanford et al. 2020). The main threats to chelonians are loss of habitat, international pet trade collection, increased prevalence of infectious disease, consumption by humans, use in traditional medicines, increase of invasive species, and habitat alteration resulting from climate change (Bowne et al. 2018, Rhodin et al. 2018, Stanford et al. 2020).

1.2 Study Species: Bog Turtle

The Bog Turtle (*Glyptemys muhlenbergii*) is one of 14 federally listed turtle species in the family Emydidae in the United States. The Bog Turtle was federally listed as an endangered species in 1997 due to the rapid decline of the northern population, comprised of populations that occur in Connecticut, Delaware, Maryland, Massachusetts, New Jersey, New York, and Pennsylvania (USFWS 1997, 2010). While the southern population (Georgia, North Carolina, South Carolina, Tennessee, and Virginia) is separated from the northern population by 250 miles and is not experiencing the same degree of population decline, both the northern and southern populations are protected under the same threatened listing due to similarity of appearance to reduce the risk of illegal trade of the species (USFWS 1997).

In 2001 a recovery plan was developed by the United States Fish and Wildlife Service (USFWS) that defined four recovery criteria for Bog Turtles (USFWS 2001). These criteria identified the need for long-term protection of at least 185 populations, monitoring populations every five years over 25 years, reducing illegal collection, and understanding long-term (>15 years) habitat dynamics within populations (USFWS 2001). In 2019 a conservation plan was implemented to guide in the achievement of these recovery criteria (Erb 2019). This thesis complements the recovery plan and the regional conservation plan, by analyzing population metrics and investigating the effectiveness of habitat management on Bog Turtles.

The Bog Turtle is a highly cryptic semi-aquatic species that occupies open-canopy fens (Arndt 1977, USFWS 1997, Whitlock 2002). This species is regarded as one of the rarest turtles in North America due, in part, to their ability to blend into the habitat in which they occupy. The shells of Bog Turtles are light to dark brown with subtle cream or yellow areas along the plastron, allowing them to easily camouflage with organic silt (USFWS 1997). Their most distinguishing feature, patches of bright yellow, orange, or red patches located on either side of the head behind the eye, are commonly hidden when observed from above (USFWS 1997). Adding to their cryptic nature is their small body size. Straight line carapace lengths for northern populations average 90–100 mm for adults of both sexes (Arndt 1977, USFWS 2001, Whitlock 2002). The growth rate is rapid during the first 5–8 years and reaches maximum size around 12 years (Arndt 1977, Whitlock 2002). Research has demonstrated that sexual maturity is dependent upon body size rather than age (Ernst 1977, Whitlock 2002). Sex can be distinguished by plastron and tail shape with males having a concave plastron and a longer, thicker tail (Barton and Price 1955, USFWS 1997, Whitlock 2002). Females deposit one clutch per year and the number of eggs ranges from two to six (Barton and Price 1955, Arndt 1977, USFWS 1997, Whitlock 2002). A study of diet found a variety of small invertebrates, including beetles, caddisfly larvae, and snails, along with plant material including seeds present in fecal samples (Melendez et al. 2017). The survival of Bog Turtles has been aided by their long lifespans and limited dispersal, reducing the potential for road mortality. As of 2020, the longest living Bog Turtle known was 62 years of age with other individuals in the same population known to be 40–50 years of age (USFWS 2022).

Bog Turtles are habitat specialists that reside in fens, unique wetlands that are composed of groundwater-dependent hydrology (Bedford and Godwin 2003, Feaga et al. 2012). This groundwater discharge results in uncommon water and soil chemistry, vegetation composition, and communities of wildlife that depend upon the seepage (Bedford and Godwin 2003, Feaga et al. 2012). Bog Turtles depend upon the stable water table, constant groundwater flow, and soil composition found in fens (Feaga et al. 2012). Throughout the active season (approximately April through September; Figure 1.1) Bog turtles are commonly found in open-canopy areas. During hibernation (October through

March) Bog Turtles reside in forested or shrub-scrub portions of fens (Whitlock 2002, Byer et al. 2018, Knoerr et al. 2021) with deep pockets of saturated soil that allow the turtles to thermoregulate (Sirois 2011, Feaga and Haas 2015).

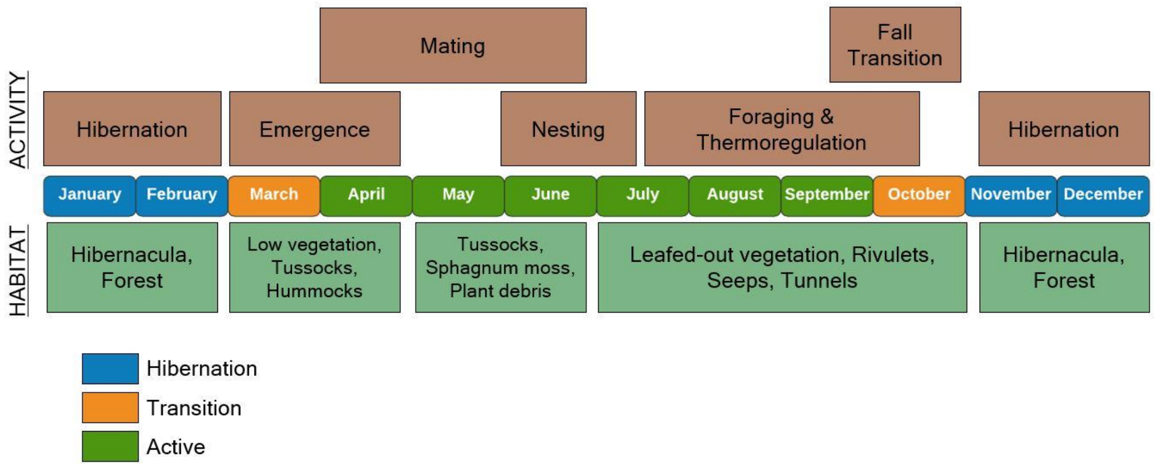


Figure 1.1: Annual Bog Turtle activity and habitat use defined by biological season.

These pockets are created by root systems of trees and shrubs, particularly alders (*Alnus* spp.), willows (*Salix* spp.), tamaracks (*Larix laricina*), and red maple (*Acer rubrum*) (Feaga and Haas 2015). Bog Turtles also use burrows created by crayfish, small mammals, and even livestock hoofprints for hibernation (Pittman and Dorcas 2009, Feaga and Haas 2015). After emergence from hibernacula in March and April, the turtles use areas of low vegetation, hummocks, and tussocks to bask (Sirois 2011, USFWS 2022). Mating occurs after emergence from March to June with nesting to follow in June and into early July (Sirois 2011, Zappalorti et al. 2015, USFWS 2022). Nest sites require grass or sedge tussock vegetation, sphagnum moss, or plant debris for nesting materials (Zappalorti et al. 2015, 2017). In the hottest months of the summer, Bog Turtles rely on shade provided by leafed-out vegetation, rivulets, and tunnels under vegetation for thermoregulation (Sirois 2011). In September and October, Bog Turtles begin moving towards the hibernacula and move around between overwintering locations before settling into one location for the winter (Sirois 2011).

1.3 Habitat Threats and Management

Fens are declining globally due to alterations in the landscape including increased urbanization, alteration of waterways, succession of vegetation, encroachment of non-native invasive vegetation, imbalanced grazing by livestock, and draining of wetlands (Morrow et al. 2001*a*, Tesauro 2001, Whitlock 2002, Smith and Cherry 2016, Hájek et al. 2020). In the United States fens are located throughout the Northeast and Midwest, as well as the mountainous West and sections of the Appalachian Mountains (Bedford and Godwin 2003). Fens have declined throughout the Holocene due to higher nutrient availability and lack of extreme habitat conditions (Hájek et al. 2020). Increased nutrient availability and lack of environmental disturbance can result in the influx of more generalist woody and vascular vegetation (Hájek et al. 2020). An increase in woody vegetation has been found to decrease the water level within the wetland initiating a positive feedback loop in which a lower water table allows for greater growth of woody vegetation, further reducing the water level through transpiration, resulting in more woody vegetation (Stratmann et al. 2020). This exacerbated decline in habitat quality can result in the rapid loss of one of the most biodiverse habitats and the species that depend upon fens (Bijkerk et al. 2022). In addition to Bog Turtles, grass-of-Parnassus (*Parnassia glauca*), slender cottongrass (*Eriophorum gracile*), and shrubby cinquefoil (*Dasiphora floribunda*) are commonly found in these imperiled fens. Of these species, slender cottongrass is one of many fen indicator species that are state-listed as threatened in Massachusetts (NHESP 2019).

Research has shown that fluctuations in the water table influence the behavior of freshwater turtles. Studies of Bog Turtle response to drought have demonstrated altered behavior as the turtles sought refuge in streams and ditches (Pittman and Dorcas 2009, Feaga 2010, Feaga et al. 2012), but did not exhibit larger than normally average movements (Carter et al. 1999). Streams and ditches are not commonly used habitats by Bog Turtles, but the course substrate and slightly lower elevation allowed those areas to remain saturated throughout the dry period (Feaga 2010, Feaga et al. 2012). However, the effects of water table fluctuations on the survival of nests are perhaps most influential to Bog Turtle populations. During drought years eggs have been found to show signs of dehydration (Zappalorti et al. 2017) and altered incubation periods (Whitlock 2002), while flooding results in the loss of eggs from drowning (Sirois 2011, Zappalorti et al. 2017). The loss of eggs is detrimental to the already low rates of recruitment of the species and has the potential to eliminate a generation from the population (Zappalorti et al. 2017, Stratmann et al. 2020).

The threat of vegetation altering the open-canopy habitat that Bog Turtles depend upon is commonly regarded as the largest threat to protected populations (Morrow et al. 2001*a*, Sirois 2011, Byer et al. 2018, Stratmann et al. 2020). Habitat management is critical for mitigating this risk of degradation and providing long-term viability (Erb 2019). To prevent further declines in populations, native woody vegetation species such as alders, red maple, and willows in the fen must be managed so that they do not reduce the availability of sunlight and ultimately alter water levels in the fens (Morrow et al.

2001*a*, Stratmann et al. 2020). Non-native invasive vegetation that commonly encroaches upon Bog Turtle habitat includes common reed (*Phragmites australis*), multiflora rose (*Rosa multiflora*), honeysuckle (*Lonicera* spp.), purple loosestrife (*Lythrum salicaria*), and barberry (*Berberis thunbergia*) (Morrow et al. 2001*a*, Sirois 2011, Angoh et al. 2021). Grazing, cutting of woody vegetation, and treatment of invasive species through herbicide application have been used to reduce the spread of these species. In some wetlands prescribed fire has been used to thin large areas of thick vegetation while the turtles are hibernating. However, with any habitat management technique, there must be preventative measures in place to minimize negative impacts on the Bog Turtles.

1.4 Massachusetts Populations

There are two Bog Turtle sites in Massachusetts that have been studied since 1994. The populations are located in the western part of the state in the Appalachian Highlands region and are on permanently protected land. These populations were intensively studied in 1994–1997 (Whitlock 2002) and 2005–2009 (Sirois 2011). The two previous studies consisted of mark-recapture studies using radio telemetry to analyze population demographics and habitat use of the populations. The first study also investigated the breeding ecology of the species. Over the two decades, the population abundance and survival remained stable at Site 1 corresponding to the stability of suitable habitat (Sirois 2011). At Site 2 the availability of suitable habitat decreased, which may explain the decline in Bog Turtle abundance and survival over the same period (Sirois 2011).

Over the last three decades, the habitat at each site has fluctuated with the encroachment of invasive vegetation, natural succession of woody vegetation, and habitat management to maintain an open canopy mosaic. Site 1 has a greater density of woody vegetation than Site 2 and is hydrologically stable due to runoff and an adjacent perennial stream. Habitat management at this site has focused on preserving the open canopy core use area and low density of invasive species while creating higher quality habitat along the perimeter of the core use area. Site 2 is impacted by flooding by beavers and encroachment of invasive species, and management has emphasized reducing the impact of beavers through the active removal of dams and beavers to stabilize hydrology.

1.5 Thesis Goals and Approach

This thesis investigated the response of Bog Turtles to long-term habitat management in the two Massachusetts sites. In Chapter 2, I estimated the current abundance and survival of Bog Turtles. I also compared the current population demographics to those in the past studies (1994–1997, 2005–2009) to analyze the long-term impacts of habitat management at both sites. Information on the status of the two populations will guide ongoing and future habitat management.

Chapter 3 evaluated habitat use by Bog Turtles at the two sites and how the spatial distribution of habitat use relates to habitat management over the three decades. Specifically, I estimated yearly home range sizes for the current study period. I also calculated home ranges within each study period and compared the shift in home range centroid for turtles found across multiple studies. The results of this study will enable managers to identify locations to focus on continued and future habitat management to maintain and expand habitat.

In Chapter 4 I discussed the management implications of these results and how they can be used to better understand the long-term impacts of habitat management on

Bog Turtles. By highlighting the need for continued habitat management, the results of this thesis will inform future planning for the conservation of Bog Turtles.

CHAPTER 2

POPULATION DEMOGRAPHICS OF BOG TURTLES IN MASSACHUSETTS: A CURRENT ANALYSIS AND COMPARISON TO PREVIOUS STUDIES

2.1 Introduction

Bog Turtles (*Glyptemys muhlenbergii*) face many pressures that threaten their survival including loss of wetlands, habitat succession, and climate change (USFWS 2022). The impacts of these threats are enhanced due to habitat fragmentation, which disconnects populations, thereby limiting gene flow and reducing dispersal, further enhancing the danger of local extinctions (Chase et al. 1989, Barron II 2021, USFWS 2022). In the northern range of the species, from New York down to Maryland, 224 of the 330 known populations are single, isolated populations, including two within Massachusetts (MA) that are the focus of this study (USFWS 2022). Despite decades of protection, it is estimated that the northern population has experienced a range reduction of 39% within the past 30 years (USFWS 2022).

Bog Turtles are commonly found in fens, wetlands with groundwater-dependent hydrology (Chase et al. 1989, Bedford and Godwin 2003, Rosenbaum and Nelson 2010, Stratmann et al. 2020). Saturation by groundwater provides pockets of well-oxygenated mud that maintain a consistent temperature range for hibernation and provide refuge for turtles escaping predators (Feaga 2010, Feaga et al. 2012, Myers and Gibbs 2013). Constant groundwater saturation also results in a signature of relatively short vegetation with indicator, obligate species that are used throughout the active season (Carter et al. 1999, Sirois 2011). This presence of short vegetation provides an open canopy that helps Bog Turtles regulate their body temperature through basking (Sirois 2011, Myers and

Gibbs 2013, Feaga and Haas 2015, Stratmann et al. 2020). The reliance on open canopy vegetation greatly limits the potential distribution of Bog Turtles and increases the need for habitat management to prevent natural habitat succession (Sirois 2011).

While the populations in MA are both located on protected land, they are not sheltered from habitat degradation via the natural succession of vegetation and influx of invasive species that alter the vegetation communities within the wetlands. To combat these pressures, the sites have been actively managed since the 1990s by removing invasive vegetation, thinning native woody vegetation, implementing prescribed fire, and managing beavers to maintain the hydrology. Many studies have highlighted the importance of active management to maintain known populations of Bog Turtles (Morrow et al. 2001*a*, Tesauro and Ehrenfeld 2007, Zappalorti et al. 2015, Travis et al. 2018); however, only a few studies have assessed the impacts of habitat management on the population demographics over multiple decades (Sirois et al. 2014, Holden 2021).

For species such as freshwater turtles that have long lifespans, delayed sexual maturity, low recruitment, and high adult survival, long-term studies are crucial for understanding trends in the populations (Zappalorti et al. 2017, Knoerr 2018, Stratmann et al. 2020, USFWS 2022). Bog Turtles can live more than 60 years in the wild and do not reach sexual maturity until around five to nine years of age, depending on body size (Congdon et al. 1993, Carter et al. 1999, Whitlock 2002, Browne and Hecnar 2007, USFWS 2022). These life history traits can lead to serious declines in populations being masked by the long lifespan of adults (Browne and Hecnar 2007). For this reason, the two populations in MA have been studied on an approximately 10-year frequency, providing enough time for the juveniles in the previous study to sexually mature and contribute to the current population.

Previous studies of the Bog Turtle populations in MA were completed in 1994–1997 (Whitlock 2002) and 2005–2009 (Sirois 2011). At Site 1, estimates of abundance (1996: $\bar{x}$= 38 ± 7.02; 2009: $\bar{x}$= 35 ± 6.62) and survival (1996: 99% ± 2%; 2008: 96% ± 6%) remained stable across the two studies (Sirois et al. 2014). The site underwent extensive habitat management throughout the early 2000's including two prescribed fires within the wetland, restoration of a pond at the edge of the wetland, as well as non-native invasive species treatment throughout the site. In contrast, at Site 2 the population declined between 1996 ($\bar{x}$=38 ± 6.44) and 2009 ($\bar{x}$=20 ± 4.32) (Sirois et al. 2014). Estimated survival also decreased at Site 2 (1997: 96% ± 3%; 2006: 72% ± 19%) over the decade (Sirois et al. 2014). The declines were due, in part, to high predation in addition to invasive species expansion resulting from shifting hydrology at the site (Sirois et al. 2014). Continued study is needed to analyze the long-term impacts of ongoing habitat management on these populations.

We investigated the impacts of habitat management on the population demographics of Bog Turtles in MA over nearly three decades. Specifically, the objectives of this research were to 1) assess the current population abundance and survival rates for Bog Turtles in MA and 2) compare the current population demographics to those in past studies to analyze the long-term impacts of habitat management at both sites. Habitat management efforts since the 2009 study have focused on maintaining high-quality habitat while also thinning the vegetation in new areas within the wetland at Site 1 and restoring the historically known use area at Site 2 through flood mitigation and thinning of native and invasive vegetation.

2.2 Methods

2.2.1 Study Area

The only two known extant populations of Bog Turtles in MA reside within two wetlands in Berkshire County. The wetland complexes are classified as palustrine systems with a mosaic of shrub/scrub, forested, and primarily emergent classes (USFWS 2011). Both sites have groundwater-dependent hydrology and vegetative communities that are constrained to wetlands with springs rising through limestone bedrock on karst topography, typical of calcareous fens (Whitlock 2002, Bedford and Godwin 2003). The area of wetland used by Bog Turtles at Site 1 is approximately 10 ha and composed of emergent, shrub/scrub, and forested wetland. At Site 2 the area used by Bog Turtles is around 4.5 ha and primarily emergent wetland with some shrub/scrub along the perimeter. As recorded from 1901–2000, the average annual temperature in Berkshire County is 6.7˚C and the county receives an annual average of 1,155.5 mm inches of precipitation (NOAA 2023). The average annual precipitation is 116.8 cm and the average annual temperature is 6.9°C for Berkshire County, calculated from 1895 to 2023 (NOAA 2023). During the four years of this study (2019–2022) average annual temperature was 7.4°C, 8.9°C, 8.7°C, and 8.2°C, respectively (NOAA 2023). The average annual precipitation throughout the four study years was 125.2 cm, 102.7 cm, 145.1 cm, and 110.8 cm, respectively (NOAA 2023). Site 1 is primarily surrounded by protected natural lands consisting of wetland, meadow, and marsh. Site 2 is primarily surrounded by residential development and agricultural land.

The wetlands provide habitat for many fauna and flora, some of which are state-listed rare species (Lowenstein et al. 1996). In addition to the Bog Turtles, a variety of amphibians and reptiles are commonly observed including frogs (*Lithobates* spp.), eastern newts (*Notophthalmus viridescens*), and Dekay's Brownsnake (*Storeria dekayi*). Many avian species use the wetlands throughout the seasons including common wetland species such as red-winged blackbirds (*Agelaius phoeniceus*), American woodcock (*Scolopax minor*), and Virginia rail (*Rallus limicola*). White-tailed deer (*Odocoileus virginianus*), raccoons (*Procyon lotor*), American beaver (*Castor canadensis*), and black bear (*Ursus americanus*) are present.

Vegetation varies between the two sites; however, alders (*Alnus* spp.), willows (*Salix spp.*), and red maple (*Acer rubrum*) are common in both. Non-woody vegetation includes shrubby cinquefoil (*Dasiphora fruticosa*), skunk cabbage (*Symplocarpus foetidus*), sphagnum moss, and many species of *Carex*. Across the sites, an abundance of rare flora has been documented including a variety of orchids, sedges, and woody taxa (Lowenstein et al. 1996). Scattered among both sites are invasive species including cattail (*Typha* spp.), purple loosestrife (*Lythrum salicaria*), and non-native phragmites (*Phragmites australis*). Both sites are actively managed by conservation partners who maintain the mosaic of wetland vegetation and stable hydrology that Bog Turtles are dependent upon.

2.2.2 Turtle Capture

I captured Bog Turtles by one of five methods: 1) during a visual survey, 2) in passive interruption traps, 3) with radio telemetry, 4) incidentally while doing radio telemetry, and 5) completely incidentally. The elusive behavior of Bog Turtles coupled with the temporal limitations of this study required the use of all capture types to optimize the number of observations (Somers and Mansfield-Jones 2008, Stratmann et al. 2020). The frequency of visual findings, whether through surveys or incidental captures, depends upon the ability of the surveyor to spot Bog Turtles within the wetland (Somers and Mansfield-Jones 2008). The experience of the surveyor in addition to the height and density of vegetation obstructing the view of the turtles can influence the effectiveness of visual captures (Lovich et al. 1992). Radio telemetry and trapping both eliminate the reliance on visual observations. Incidental finds are common when searching for a turtle with a radio as Bog Turtles congregate for mating and often share hibernacula (Ernst et al. 1989, Sirois 2011). Trapping eliminates surveyor detection variability and temporal limitations by capturing turtles continuously when deployed, but this method is the most time-intensive as traps must be checked daily (Somers and Mansfield-Jones 2008).

Visual surveys were completed following the U.S. Fish and Wildlife Service (USFWS) regional survey guidelines (USFWS 2020). We surveyed on May 8th, May 21st, and June 1st in 2019; April 29th, May 6th, May 12th, May 18th, and May 26th in 2022. No surveys were completed during 2020 and 2021 due to the COVID-19 pandemic. Surveyors (n = 3–11 during each survey) meandered throughout the designated area while searching for individual turtles, signs of turtles, and suitable habitat features. Survey duration depended on the quality of the habitat and the size of the site. Throughout the 2022 visual surveys, 57.25 person hours were spent at Site 1, and 63.67 person hours at Site 2.

Passive interruption traps (hereafter, traps) were constructed based on those developed by Whitlock (2002) and used by Sirois (2011) (**Error! Reference source not**

found.) using 1.27 cm hardware cloth. Each trap consisted of a main body (45 cm x 14 cm x 12 cm) with two doors (14 cm x 12 cm), one at each end, which fall behind the turtle when it enters inhibiting escape from the trap. Extending off each end were two arms (40 cm x 10 cm). In 2019 trapping was completed in May with 1,449 total trap nights across both sites. Trapping occurred from May through September in 2021 (515 total trap nights) and May through July in 2022 (2,425 trap nights total). Traps were deployed in rivulets and other potential passageways, with the arms extended to guide the turtles into the traps by molding them flush to rivulet edges, hummocks and tussocks, and intersecting game trails. To protect captured turtles from heat stress, each trap was covered with vegetation collected from the immediate area to provide shade. The bottoms of traps were submerged in approximately 1 cm of water to allow for cooling but were supported so that they would not fall into water deep enough to flood the trap. The density of traps was dependent upon the individual sites and number of available traps but followed the regional protocol of 10–20 traps per hectare dependent upon wetland type (USFWS n.d.).

During 2019 20 individual traps were set at Site 1 and 25 traps at Site 2, each for 22 nights total. No trapping was completed during 2020 due to the COVID-19 pandemic. In 2021, 20 traps were set at Site 1 and 30 traps at Site 2. In 2022 there were two rounds of trapping at each site with 50 set in the first round and 25 in the second. Traps were checked daily (within 24 hrs.) following the Northeastern regional protocol (USFWS n.d.) and were pulled from the wetlands before any major rain event (> 2.54 cm) to prevent rising water from flooding the trap.

Figure 2.1: Passive interruption trap set with vegetation covering to provide shade to turtles. Photo: J. Vineyard

Radio telemetry was completed at each of the two sites from 2019–2022. Ten adult turtles at each site were equipped with ATS R1680 VHF micro-radio transmitters (Advanced Telemetry Systems, Isanti, MN). Radios were affixed using a two-part quick-set epoxy (WaterWeld: J-B Weld, Marietta, Georgia) on the right rear of the carapace so that any interference with mating would be minimized. Throughout 2019 and 2020 turtles were tracked once a week from early spring emergence until return to hibernacula. The frequency of tracking increased to twice a week during the most active seasons of 2021 and 2022, after spring mating until turtles returned to hibernacula. Locations were recorded once per month from December 2021–March 2022 to confirm that no radio signals were lost over the winter.

Incidental captures of turtles were broken up into two categories: incidental while doing radio telemetry, or completely incidental. Incidental turtles that were found while completing radio telemetry include those that were visual finds while moving through the wetland, those that were pulled out of silt pockets while searching for a turtle with a radio, and those heard or seen while processing a radio telemetry capture. Any capture that did not occur during a visual survey or radio telemetry event was recorded as a completely incidental capture. These captures primarily consisted of observations by those implementing habitat management within the wetlands.

2.2.3 Turtle Measurements

Each captured turtle was evaluated for identifying notches that had been previously scored into the marginal scutes of the plastron. These notches are used for the identification of individuals by coding out to a unique value using the system developed by Ernst (Ernst et al. 1974). If a turtle was not previously notched and was 50 g or larger, it was marked by filing the corresponding marginal scutes with a sanitized triangular file.

Each turtle was examined for overall health at each capture. Weight and any scute morphology irregularities were recorded once per year. Any external injuries, discharge, or changes in overall health were noted at each capture. Photographs were taken of each turtle to aid in later identification if marks were lost. The mode of capture and the health metrics were recorded along with the location of the turtle using ArcGIS Collector (Environmental Systems Research Institute, Inc., Redlands, CA, USA).

2.2.4 Data Analysis

2.2.4.1 Objective 1

We calculated sex ratios, population abundance, and survival estimates using data resulting from all capture types. Any turtles that could not be positively identified whether due to loss of notches or the animal was too small to mark with notches (<10 g), were removed from the dataset to confirm to model assumptions (Pollock and Alpizar-Jara 2010). Analysis was limited to adult turtles (> 70 mm carapace length; Ernst 1977) as the small body size of juveniles is likely to reduce detection probability. Sex ratios were calculated for each year at each site, and across years for both sites combined. The sex ratios for each site were tested for statistical significance between years using an ANOVA and Tukey Honest Significance Difference (HSD) test.

Population abundance and survival were estimated using Cormack-Jolly-Seber (CJS) models. This open population model enabled analysis of the temporal change in abundance by allowing the population to expand and contract through emigration, immigration, birth, and death throughout the 4 years (Manly et al. 2005, Iijima 2020). CJS models use the first capture occasion as the reference level so estimates are created for the subsequent years only; in this, case estimates were produced for 2020–2022 (McDonald et al. 2018). The last survival parameter does not exist in CJS models, limiting estimates to 2019–2021 (McDonald et al. 2018). Program R was used with the package ‘mra’ as it can incorporate known deaths into the analysis of mark-recapture data (McDonald et al. 2018, R Core Team 2021).

2.2.4.2 Objective 2

Data from study period 1 (1995–1997; Whitlock 2002) and study period 2 (2005–2009; Sirois 2011) were incorporated with the current dataset (study period 3; 2019–2022) to compare population demographics over nearly three decades. The previous studies used weekly radio telemetry, visual surveys, and trapping at equal levels of effort to capture and monitor turtles (Whitlock 2002, Sirois 2011). During both study periods, researchers conducted visual surveys from late March through October and implemented trapping from April through June (Whitlock 2002, Sirois 2011).

We limited data to turtles found in the geographic area surveyed during all three periods to conform to model assumptions. We excluded 2006 at Site 2 due to a lower level of effort reflected in the total number of captures. Capture histories were limited to only adult turtles and those that could be positively identified. Sex ratios were calculated for each of the three study periods at each site. Survival and abundance estimates were calculated using the same CJS modeling as described in Objective 1 with R package 'mra' (McDonald et al. 2018, R Core Team 2021). Differences in sex ratios, population abundance, and survival estimates among the three periods were tested using an ANOVA for each variable at each site. Tukey's HSD tests were used post hoc to identify pairwise differences among studies.

2.3 Results

2.3.1 Objective 1: Current Population Demographics

Throughout the current study period (2019–2022) 143 adult turtles were observed across both sites. More adults were found at Site 1 (n= 90) than at Site 2 (n= 53) (Table 2.1, Figure 2.2). As the study spanned multiple years some turtles were found in both adult and juvenile age classes (Site 1 n= 4, Site 2 n=3). Most adult turtles were seen multiple times (Site 1 = 57, Site 2 = 41) while only 24 turtles were seen once at Site 1, and 12 turtles were seen once at Site 2 during the four years. Throughout the study period, one dead turtle was recovered at Site 2 while three were recovered at Site 2. At both sites male turtles were observed more often than female turtles; however, the number of female individuals (n=79) was higher than males (n=64). When looking at sites individually, male:female sex ratios did not vary significantly by year (ANOVA: Site 1 F=0.065, p=0.823; Site 2 F=0.652, p=0.504).

Table 2.1: Number of Bog Turtles observed at two sites in Massachusetts throughout the current study period (2019–2022). The total number of observed turtles across the years is higher than the total number of individuals since the individual turtles were found in multiple years.

Site	**Sex**	**Age Class**	**2019**	**2020**	**2021**	**2022**	**Total Observed**	**Total Individuals**
1	Male	Adult	16	12	20	34	82	39
	Female	Adult	17	21	19	39	96	51
	Unknown	Juvenile	7	0	6	6	20	17
2	Male	Adult	12	12	18	19	61	25
	Female	Adult	13	9	15	16	53	28
	Unknown	Juvenile	3	2	4	7	17	15

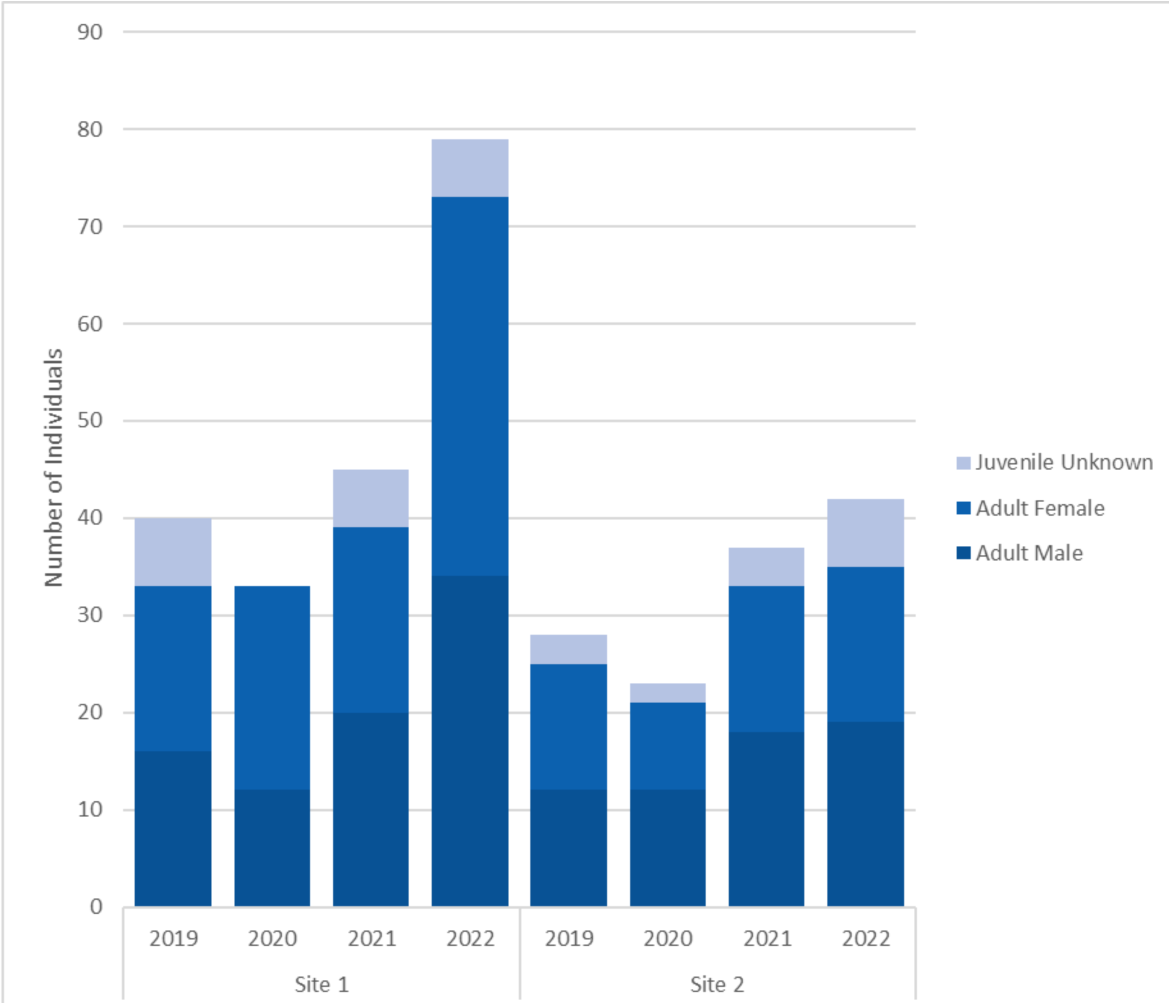


Figure 2.2: Age class and sex class distribution of positively identified Bog Turtles observed at two sites in Massachusetts throughout 2019–2022.

At Site 1, yearly estimated population abundances were 65.5 ± 27.0 in 2020, 68.4 ± 21.0 in 2021, and 80.8 ± 18.8 in 2022 (Figure 2.3). At Site 2 estimates of population abundance were 42.0 ± 14.0 in 2020, 39.2 ± 9.0 in 2021, and 46.2 ± 9.8 in 2022 (Figure 2.3). CJS models of population abundance are not able to calculate the first capture probability (McDonald et al. 2018).

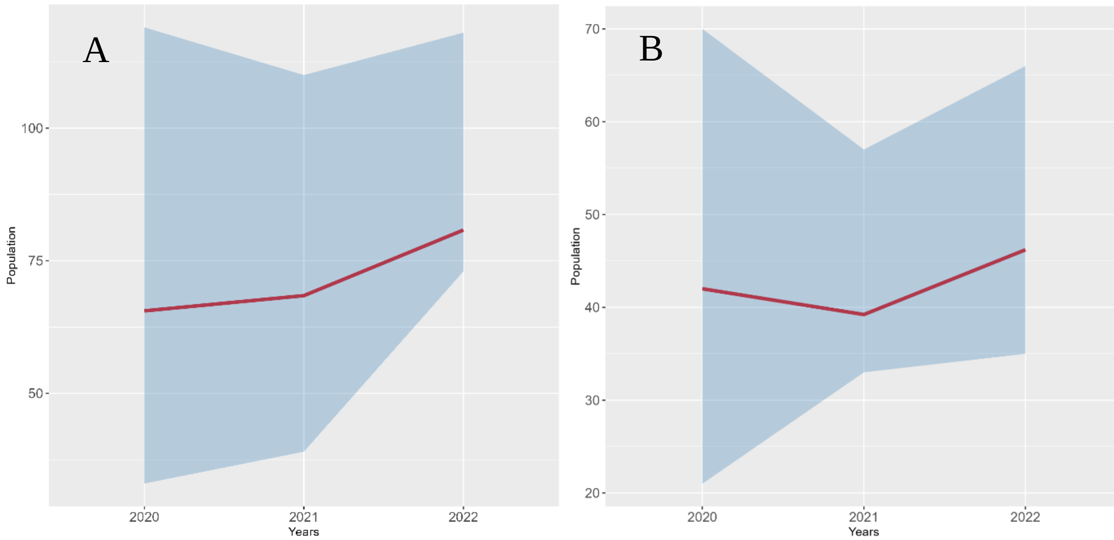


Figure 2.3: Abundance estimates for each year of the current study of Bog Turtles within Massachuestts at Site 1 (A) and Site 2 (B). Estimated abundance is displayed by the red line with standard error surrounding in blue.

At Site 1 survival estimates were 90% ± 15% in 2019, 95% ± 15% in 2020, and 90% ± 15% in 2021 (Table 2.2, Figure 2.4). At Site 2 survival was 100% ± 2% in 2019, 78% ± 12% in 2020, and 100% ± 2% in 2021 (Table 2.2, Figure 2.4). CJS models cannot calculate survival for the last year of data (McDonald et al. 2018).

Table 2.2: Survival estimates for Bog Turtles at both sites in Massachusetts.

Site	Year	Survival Estimate	Std. Error
1	2019	90%	15%
	2020	95%	15%
	2021	90%	15%
2	2019	100%	2%
	2020	78%	12%
	2021	100%	2%

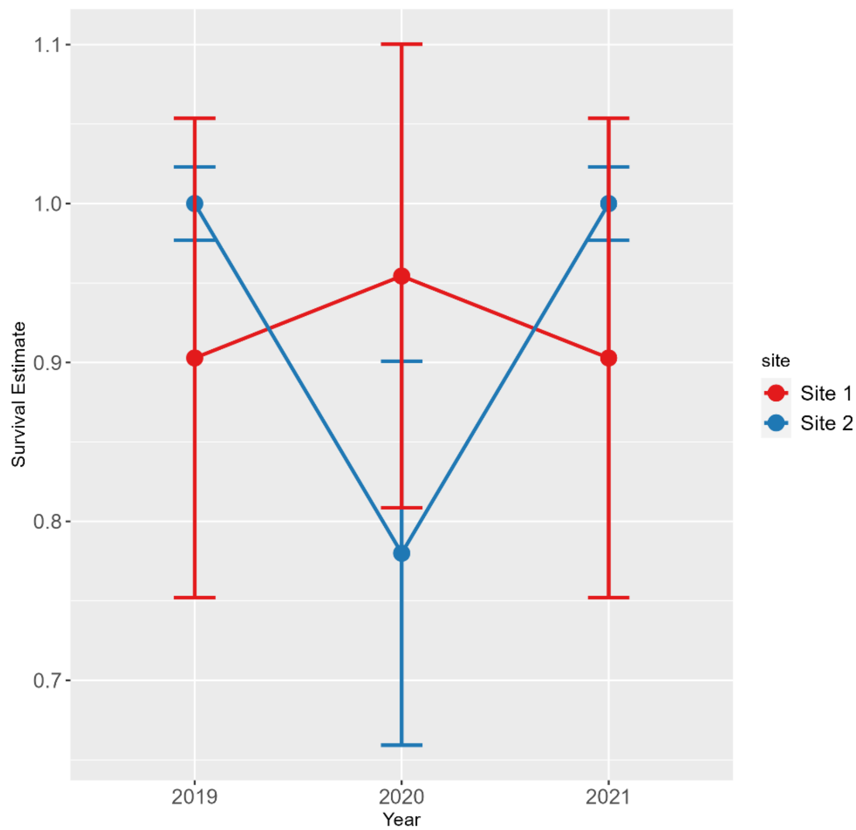


Figure 2.4: Survival estimates for both sites of Bog Turtles in Massachusetts across three years.

2.3.2 Objective 2: Comparison of Population Demographics Across Decades

The number of adult Bog Turtles observed during each of the three study periods ranged from 75 in study period 1 (1994–1997) to 69 in study period 2 (2005–2009) and 134 in study period 3 (2019–2022; Table 2.4). The number of adults observed in the current study differs by 9 from the value reported previously due to the removal of observations within new areas of Site 1. The total area that Bog Turtles are known to occupy at Site 1 increased with the discovery of a new area in 2018 and another new pocket of habitat in 2019, both due to tracking turtles on radio. The area discovered in 2018 is 0.18 ha and southeast of the known core habitat, and the habitat found in 2019 is

0.75 ha and northeast of the core habitat. These pockets of habitat increased the overall footprint in which turtles were found in the two previous studies.

Site 1 sex ratios (male:female) increased from 0.65 in study period 1 to 0.76 in period 2 to finally 0.80 in period 3 (Table 2.3). Site 2 had a similar increase in the sex ratio with 0.54, 0.60, and 0.89, respectively. The sex ratio differed by study period for Site 1 (ANOVA; $p = 0.006$) and Site 2 (ANOVA; $p = 0.035$).

Table 2.3: Number of Bog Turtles observed by sex and male:female sex ratios for study period 1(1994–1997), period 2 (2005–2009), and period 3 (2019–2022) at each site.

Site	Study Period	Males	Females	Sex Ratio
1	1	15	23	0.65
	2	16	21	0.76
	3	36	45	0.80
2	1	13	24	0.54
	2	12	20	0.60
	3	25	28	0.89

At Site 1, estimated population abundance differed among study periods (ANOVA: $F=28.45$, $p<0.001$), with the highest abundance in period 3 ($\bar{X}= 65.1 \pm 17.9$), and lower abundances in period 1 ($\bar{X}= 37.3 \pm 10.4$) and period 2 ($\bar{X}= 30.3 \pm 4.7$) (Table 2.4, Figure 2.5). Population abundance at Site 2 also statistically differed among the three study periods (ANOVA: $F=21.56$, $p=0.004$). The highest abundance was in study period 3 ($\bar{X}= 42.5 \pm 10.9$), followed by period 1 ($\bar{X}= 36.2 \pm 3.2$), and period 2 ($\bar{X}= 24.2 \pm 22.1$). CJS models are unable to estimate the first capture probability (McDonald et al. 2018).

Table 2.4: Population estimates for adult Bog Turtles throughout three study intervals (1997–1997, 2005–2009, 2019–2022).

Site	Year	Individuals	Estimate	Std. Err.
1	1995	30		
	1996	31	36.7	4.0
	1997	33	37.9	16.7
	2005	12		
	2006	20	26.7	5.4
	2007	17	27.9	6.3
	2008	27	33.9	4.8
	2009	29	31.4	2.3
	2019	28		
	2020	33	58.9	20.9
	2021	35	59.8	17.0
	2022	66	76.7	15.8
2	1995	25		
	1996	32	36.4	3.5
	1997	33	36.1	3.0
	2007	10	28.5	12.7
	2008	21	23.8	2.8
	2009	20	20.2	51.2
	2019	25		
	2020	21	42.0	14.0
	2021	33	39.2	9.0
	2022	35	46.2	9.8

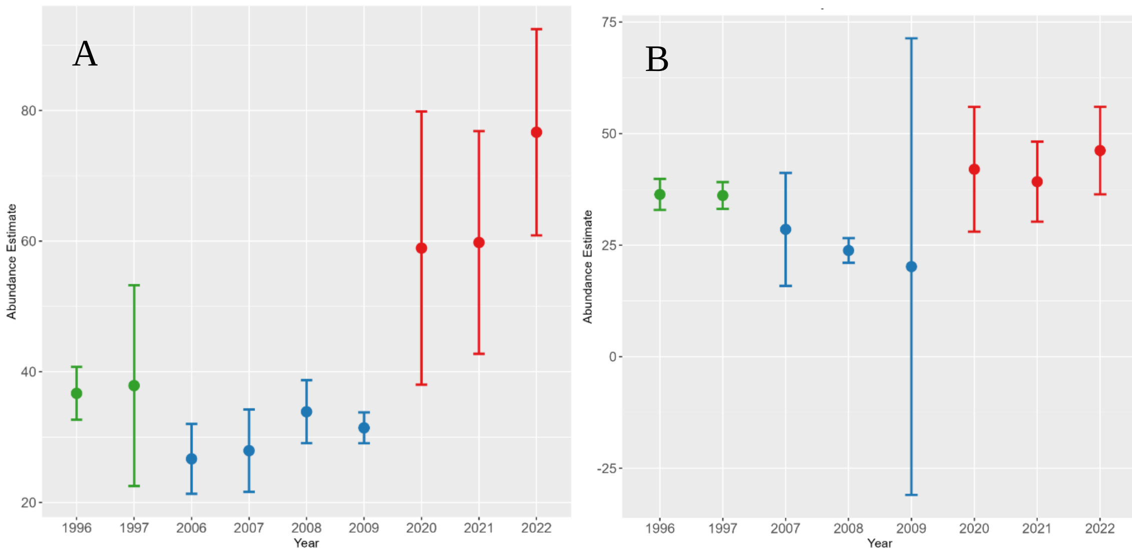


Figure 2.5: Population estimates (mean ± standard error) of Bog Turtles at Site 1 (A) and Site 2 (B) in Period 1 (green), Period 2 (blue), and Period 3 (red).

Estimated survival was consistently above 90% at Site 1. At Site 2, survival was 100% in five of the years, but was lower in 2005 (89%), 2008 (76%), and 2020 (78%) (

Table 2.5). There was no significant effect of the study period on the survival of Bog Turtles at Site 1 (ANOVA: $F=2.223$, $p=0.190$) or Site 2 (ANOVA: $F=0.658$, $p=0.558$).

Table 2.5: Cormack-Jolly-Seber (CJS) survival estimates and standard error (in parentheses) of adult Bog Turtles for each year by site. NA = Not Applicable.

Study Period	Year	Site 1	Site 2
1	1995	99% (4%)	100% (0%)
	1996	100% (42%)	100% (5%)
2	2005	100% (0%)	89% (6%)
	2006	100% (0%)	NA
	2007	93% (6%)	100% (0%)
	2008	100% (2%)	76% (197%)
3	2019	96% (13%)	100% (2%)
	2020	92% (14%)	78% (12%)
	2021	96% (13%)	100% (2%)

2.4 Discussion

The results of this study indicate that continued habitat management efforts have provided habitat sufficient for increased estimates of population abundance over the last nearly three decades. The survival rates of adults remained around 90% at both sites, apart from three years at Site 2. While the estimated survival at both sites dips below the 96% threshold suggested by a stable population of Bog Turtles in New York (Shoemaker et al. 2013), the overall trend in abundance does not reflect a decline in the current population size.

The cumulative knowledge of the areas used by the Bog Turtles at the sites in MA has grown since study period 1. This increase in knowledge has inadvertently led to potentially greater rates of detection across the sites since the first study, resulting in possible bias in the increased abundance estimates reported in this study. However, variation in the surveyors throughout the years of the three study periods and the use of multiple capture types in each study likely limits this bias. Bias was minimized by using multiple capture types throughout the three study periods. Variety in capture types reduced high rates of recapture of individual turtles and allowed equal probability of detection among male and female turtles (Hanscom et al. 2020). Site fidelity has been demonstrated in the overwintering habitat of Bog Turtles (Sirois et al. 2014) leading to potential biases in individuals captured via visual surveys completed shortly after emergence or before hibernation, as turtles return to known hibernacula. A movement-based capture method, such as trapping, has the potential to bias sex ratios as some studies have demonstrated a relationship between sex and movement (Lovich et al. 1992, Pittman and Dorcas 2009). By using multiple capture types, including radio telemetry, visual surveys, trapping, and incidental observations, I was able to reduce the bias in the turtles captured and the overall sex ratio.

The sex ratio was found to significantly fluctuate between the three study periods at both sites, with an overall increase in the number of males to females observed. These results are not due to climatic variation altering the sex ratio of hatchlings as with other species of freshwater turtles (Parrott and David Logan 2010, Roberts et al. 2023, Vogt and Flores-Villela 2023) due to Bog Turtles having genetic sex determination (Literman et al. 2017). Studies have demonstrated that species of freshwater turtles with genetic sex determination sex ratios have considerable variation across populations and time (Georges et al. 2006, Hanscom et al. 2020), but the significance of changing sex ratio across the study periods is most likely driven by the focus of the first study period on breeding ecology (Whitlock 2002). Study period 1 focused on capturing female Bog Turtles and understanding their use of the wetland for nesting (Whitlock 2002), potentially explaining why fewer males were captured. During the most recent study period, I prioritized having an equal number of male and female Bog Turtles as part of the radio telemetry effort to reduce potential bias in observed sex ratios.

Ultimately, the increased estimates of adult population abundance at both sites throughout the last 2.5 decades support the minimum viable population size suggested by Shoemaker et al. (2013) that populations of less than 50 individuals can persist when the quality of habitat is maintained, and the conservation of the species is put at the forefront of any activities. However, small populations are exceptionally vulnerable to many factors including the loss of quality habitat, loss of genetic variability, and loss of individuals through a natural catastrophe or illegal collection (Shoemaker 2011, Reed and Mccoy 2014, Erb 2019). In the northeast range of the Bog Turtle, the risk posed by these threats is heightened as most populations are estimated to have less than 30 individuals (Erb 2019). The continued study of the two sites in MA provides knowledge on the true viability of isolated Bog Turtle populations and how populations fluctuate throughout years of alternating habitat quality.

The influence of habitat quality on population abundance estimates has been demonstrated many times in the degradation of habitat resulting in decreased population abundances (Sirois et al. 2014, Stratmann et al. 2020, Holden 2021), but research has been limited in demonstrating how habitat management can positively influence population demographics of Bog Turtles (Sirois et al. 2014). This study demonstrates that within two populations in MA habitat management efforts tailored to the individual needs of the sites have provided habitat in which the populations can continue to exist for nearly three decades. These results further support the implementation of habitat management efforts—with caution to reduce any negative impacts to the current population—for the conservation of the species.

CHAPTER 3

HOME RANGE AND SPATIAL DISTRUBITION OF BOG TURTLES IN MASSACHUSETTS

3.1 Introduction

The importance of habitat management to maintain high-quality habitat for Bog Turtles (*Glyptemys muhlenbergii*) has been recognized since the species' federal listing in 1997 (USFWS 1997). However, only recently have studies evaluated the species' long-term response to habitat management. Understanding the influence of habitat management activities, such as invasive species control, low-intensity livestock grazing, and hydrological restoration, on known populations of Bog Turtles at long-term time scales will allow us to better protect and manage this imperiled species.

The primary threats known to Bog Turtle populations are habitat loss, anthropogenic fragmentation, natural succession, and degradation through eutrophication and invasive plant invasions within wetland systems (USFWS 1997, Carter et al. 1999, Myers and Gibbs 2013, Smith and Cherry 2016, Erb 2019). It was estimated that the northern population of the species (i.e., Connecticut, Delaware, Maryland, Massachusetts, New Jersey, New York, and Pennsylvania) experienced a 39% reduction in the occupied range from 1989 to 2019 within the last 30 years (USFWS 2022). Conservation planning for the species has focused on the entire northern population to achieve recovery criteria that were established in the 2001 Bog Turtle Northern Population Recovery Plan (Erb 2019). However, Bog Turtles regularly exist in small, disjunct populations of functionally reproductive groups of individuals that are particularly susceptible to local threats due to their isolation(Whitlock 2002, Pittman et al. 2011, Erb 2019). An understanding of how these small, isolated populations function and fluctuate with changes in habitat is needed to protect the species as a whole.

The life history of Bog Turtles and their unique habitat requirements limit their distribution and increase the risk of extirpation. Like most species of turtles, Bog Turtles are long-lived with lifespans >60 years in the wild and have delayed sexual maturity, reaching maturity around seven to nine years of age depending upon the size of the individual (Congdon et al. 1993, Whitlock 2002, Browne and Hecnar 2007, USFWS 2022). The species is a habitat specialist, relying on the stable hydrology, soil composition, and vegetation provided by fens (Chase et al. 1989, Bedford and Godwin 2003, Pittman and Dorcas 2009, Feaga and Haas 2015, Stratmann et al. 2020). The constant discharge of groundwater limits the growth of vegetation, resulting in distinctively short vegetation and fen indicator species (Bedford and Godwin 2003, Feaga et al. 2012, Myers and Gibbs 2013). This stunted vegetation provides an open canopy habitat that is crucial for Bog Turtles to regulate their body temperature through basking (Sirois 2011, Myers and Gibbs 2013, Feaga and Haas 2015, Stratmann et al. 2020). Reliance on groundwater-driven systems greatly limits the potential distribution of Bog Turtle populations and increases the need for habitat management where populations are known to occur (Sirois 2011).

Open canopy habitat is considered ephemeral due to the constant threat of natural vegetative succession by native woody vegetation in addition to the potential encroachment of invasive species (Ernst and Lovich 2009). Succession of vegetation alters the water table which in turn creates a positive feedback cycle allowing more woody vegetation to invade and continue to dry the wetland (Ernst and Lovich 2009, Feaga et al. 2012, Stratmann et al. 2020). To combat this natural succession, disturbance—such as prescribed fire, removal of invasive vegetation, or thinning of woody vegetation—is necessary to maintain the wetland hydrology and vegetation (Ernst and Lovich 2009, Stratmann et al. 2020). However, a balance must be maintained in the amount of disturbance to avoid inadvertently destructive effects on the biota that encompass Bog Turtle ecosystems. The mosaic of vegetation, flow of groundwater through the system, and silt that Bog Turtles require cannot be quickly nor easily recreated (Ernst and Lovich 2009). Planned disturbances within wetlands to support Bog Turtle populations must consider not only the short-term benefits but also the long-term influence on the populations. Any habitat management must minimize potential mortality within the population and not restrict access to areas of non-disturbed habitat (Dodd et al. 2006). With adequate precautions in place, populations of turtles may benefit from targeted habitat management (Dodd et al. 2006).

While studies have identified that habitat management is crucial for the potential recovery of Bog Turtles (Morrow et al. 2001*a*, Tesauro and Ehrenfeld 2007, Sirois et al. 2014, Stratmann et al. 2020), very few studies have been able to assess the relationship between habitat alteration and the spatial distribution of Bog Turtles on a long-term basis (Sirois et al. 2014, Barron II 2021). Barron (2021) analyzed data recorded over 32 years and found evidence of Bog Turtles moving out of a wetland where a beaver impoundment made the habitat unsuitable for Bog Turtles. Similarly, Sirois et al. (2014) found that Bog Turtles at one site shifted home ranges to avoid areas of degraded habitat due to beaver flooding. At a site where the habitat had been restored through the treatment of invasive species, the Bog Turtles shifted home ranges to use the restored area (Sirois et al. 2014).

In this study, I assessed the influence of habitat management—treatment of invasive vegetation, thinning of woody vegetation, mitigation of flooding—on two populations of Bog Turtles that have been actively managed since the 1990s. Specifically, the objectives of this research were to 1) investigate the current spatial distribution of the Bog Turtles across two occupied wetland complexes in Massachusetts, and 2) compare Bog Turtle wetland use (i.e., location, spatial extent) to habitat use in past decades. The results of this study will enable managers to identify areas where ongoing habitat management is necessary and determine areas where essential habitat can be increased. By investigating how the habitat management activities influence the spatial distribution of turtles throughout the wetland we can better understand how the turtles are directly impacted by these disturbances.

3.2 Methods

3.2.1 Study Area

The research took place in southern Berkshire County, Massachusetts. The average annual temperature for Berkshire County is 6.9°C and the average annual precipitation is 116.8 cm from 1895 to 2023 (NOAA 2023). Average annual temperatures during the four years of the study period (2019–2022) were 7.4°C, 8.9°C, 8.7°C, and 8.2°C, respectively. Average annual precipitation throughout the study years was 125.2 cm, 102.7 cm, 145.1 cm, and 110.8 cm, respectively.

I studied two populations of Bog Turtles located in Core Habitat and Critical Natural Landscape, a state-level conservation planning designation, wetlands (Commonwealth of Massachusetts and The Nature Conservancy 2022). Site 1 is an 11-ha wetland that is classified as primarily freshwater forested/shrub wetland with some areas of freshwater emergent wetland (USFWS 2011). Site 2 consists of 4.5 ha of wetland that is classified as a combination of freshwater emergent wetland and freshwater forested/shrub wetland (USFWS 2011). Both sites are calcareous sloping fens with groundwater-dependent hydrology and vegetative communities that are constrained to wetlands with springs rising through calcareous bedrock including calcite marble with dolomite masses (Whitlock 2002, Bedford and Godwin 2003, MacDougall 2016). Both sites are permanently protected by non-profit and/or state agencies and managed for the Bog Turtle. Site 1 is primarily surrounded by protected natural lands consisting of wetland, meadow, and marsh. Site 2 is constrained by residential development and agricultural use land. These sites provide refuge for many fauna and flora including state-listed rare species. Both sites also contain invasive species, including Japanese barberry (*Berberis* thunbergia), purple loosestrife (*Lythrum salicaria*), and non-native phragmites (*Phragmites australis*).

Both sites are actively managed to maintain the biodiversity associated with calcareous sloping fens. Numerous studies have been conducted since the 1990s to understand the vegetation composition, hydrologic status, and soil composition of the wetlands (Lowenstein et al. 1996, Stevens 1996, Lowenstein 1998, Morgan 2008, MacDougall 2016). The information obtained through these studies has been directly applied to management strategies implemented since the late 1990s. Site 1 has been managed since 1999 through mechanical and chemical treatment of invasive species, thinning of woody vegetation, removal of trees and shrubs, and prescribed fire. These activities have been focused on maintaining the open canopy habitat in the core use area of the fen. In 2018 and 2019 two new areas were identified as used habitats by tracking a single turtle to each area. Habitat management shifted in the year 2020 from focusing on managing the core use area of the wetland to thinning woody vegetation in the first new area. The new area identified in 2019 is on the periphery of an area that has received treatment for phragmites but is otherwise a suitable habitat. At Site 2, controlling flooding by beavers has been the focus of habitat management efforts, to reduce the density of cattail and invasive phragmites in the wetland. Phragmites was chemically treated before the beavers moved into the site and although it is still being treated, it has been controlled within the core use area. Likewise, beavers are still present within the system, but flooding events were rarely observed during the current study due to actively removing dams and managing individuals. The primary focuses of habitat management at Site 2 are maintaining a low level of invasive species throughout the Bog Turtle use area, balancing the open canopy habitat with the necessary structure provided by native woody vegetation, and reclaiming periphery habitat that has degraded.

3.2.2 Turtle Capture

At each of the two sites, 10 turtles were equipped with ATS R1680 VHF micro-radio transmitters per year throughout 2019–2022 (Advanced Telemetry Systems, Isanti, MN). Turtles were chosen from the pool of those that were already fitted with radios, originally deployed during 2018, and those that were included in the previous studies. If a radio fell off a turtle it was placed back onto the same turtle. If the same turtle was not able to be found, the radio was affixed to the next turtle found without a radio. Priority was given to any turtle who had been part of previous radio telemetry efforts.

Radios were secured to the posterior portion of the carapace centered on the seam between the pleural scutes and the marginal scutes near the right pleural scute #4 (i.e., just above the right hind foot on the outer surface of the carapace). A two-park quick-set epoxy (WaterWeld; J-B Weld, Marietta, Georgia) was used to affix the radios in the field. The total weight of the radio and epoxy did not exceed 5% of the individual turtle's body weight, or one-half the standard recommended by the Society for the Study of Amphibians and Reptiles (Beaupre et al. 2004). This transmitter model had an estimated battery life of over one year, but I changed the radios yearly to ensure that no radio was left remaining on a turtle past the battery life.

Turtles were tracked from February 2019 until December 2022. The frequency at which tracking was completed was variable depending on the relative activity level of the turtles. The location of each turtle was obtained through tracking once a week during the early spring (April) to capture emergence. I transitioned to tracking twice a week during the 2021 and 2022 seasons when greater distance movements were observed in preparation for mating (typically mid-May) and locations were collected twice a week throughout the active season. I reduced the frequency of tracking to once a week as movement slowed during the return to hibernacula in late September to October. Locations were collected once a week for the active seasons of 2019 and 2020. I completed radio telemetry once per month from December 2021–March 2022 to confirm that no signals were lost over the winter and that the turtles remained in the same hibernacula.

I collected data at each turtle observation including the GPS coordinates, time of observation, behavior (basking in open, cryptic basking, feeding, moving, mating, nesting, hibernating, submerged, or other), shell condition, health (skin pigmentation, presence of ectoparasites, signs of infection), and description of the turtle's micro-location (on vegetation, under vegetation, underground, in a rivulet, under mud, in water, in a tunnel, or other). I also recorded habitat characteristics, including the wetland type (forested, shrub-scrub, or emergent) and dominant vegetative species within a 1-m and 5-m radius from each turtle. The number of cut stumps greater than 5 cm in diameter that were visible within an approximately 2-m radius from the turtles was also recorded.

Additional habitat data were collected to assess habitat preferences by comparing the characteristics of habitat at turtle locations with paired random locations without a turtle, following the methodology described by Compton et al. (2002). I identified a random subset of 30 locations where turtles were found (3 per turtle) at each site from June through August 2022. I returned to each of the 30 selected known locations in November 2022 and identified a paired random point in November of 2022. This point was chosen by moving on a random bearing, by looking at the current time in seconds and moving along the corresponding bearing with zero seconds aligning with North and thirty seconds being South. The distance traveled between known locations and the paired random locations was based on the average daily movement for each sex of Bog Turtle reported from a comparable population in Maryland (Morrow et al. 2001*b*). I created a uniform distribution of values that averaged the reported value for each sex and randomly selected from this distribution the distance between the known and paired random locations. At each known location and paired random point I assessed the same habitat variables that were recorded at each turtle location during radio telemetry monitoring: wetland type, dominant species, and number of visible cut stumps. Additionally, I measured the average basal area of standing trees surrounding the locations using a Cruz All. I recorded the number of trees that were counted as “in” on a Cruz All at the basal area factor of 10 and 5 using a variable radius plot for each known and paired random location (Henning 2009).

3.2.3 Data Analysis

3.2.3.1 Objective 1

Yearly home ranges of turtles at both sites were constructed using data from 2019–2022. Estimated home range size tends to increase with the number of locations in a year up until an animal delineates its home range at which point the estimated size stabilizes (Harris et al. 1990, Morollón et al. 2022). To identify the number of locations needed for home range estimates to stabilize, I plotted the 95% minimum convex polygon (MCP) home range size against the number of relocations used to calculate the home range size for each turtle. The asymptote of these accumulation plots revealed that the home range size stabilized at approximately 20 locations. I, therefore, limited the dataset to turtles with at least 20 observations per year and used all locations to generate home ranges.

I used the R package ‘adehabitatHR’ to construct home ranges using both 95% MCP and kernel density estimates (KDE) (Clement 2022). Studies have demonstrated that MCP may overestimate home range size; however, MCP is still frequently used, especially in studies that draw comparisons to older studies (Lawson and Rodgers 1997, Carter et al. 1999, Börger et al. 2006, Nilsen et al. 2008, McCoy et al. 2021). The smoothing parameter of each kernel density estimate was calculated using Silverman's rule of thumb due to the small sample sizes (Harpole et al. 2014). Home ranges were created for individual turtles each year that they were tracked, as well as a site composite home range for all turtles for each year of study. I tested whether 95% MCP and 95% KDE home range size differed by sex, site, and year using generalized linear models and mixed effects modeling through the 'lme4' R package (Bates et al. 2015).

I compared habitat variables collected at known turtle locations and paired random locations using chi-square tests. The four variables (dominant vegetation, wetland type, number of visible cut stumps, and the number of trees) were analyzed independently, with comparisons being between the same site and spatial scale, where relevant.

3.2.3.2 Objective 2

I compared the distribution of home ranges among three study periods: 1= 1994–1997 (Whitlock 2002), 2= 2005–2009 (Sirois 2011), and 3= 2019–2022 (current study). The datasets were limited to individual turtles that had at least 20 observations per study period, as the historic datasets were not as robust as the current dataset. Both 95% MCP and 95% kernel density estimate home ranges were generated using the 'adehabitatHR' package in R (Clement 2022). Silverman's rule of thumb was used to calculate the smoothing parameter for each kernel density estimate (Harpole et al. 2014). Generalized linear models and mixed effects models were created with the 'lme4' package in R to test 95% MCP and 95% KDE home range size by sex, site, and study period (Bates et al. 2015).

To test whether individual turtle’s home ranges shifted among the three study periods, I calculated the centroids of the 95% MCP home ranges of individual turtles in each study using the ‘rgeos’ package in R (Bivand and Rundel 2017). I then used the ‘sp’ package in R to find the Euclidian distance between the centroid points of consecutive studies (Pebesma and Bivand 2023). This allowed me to evaluate the shift in home range centroids among study periods for individual turtles.

3.3 Results

3.3.1 Objective 1

At Site 1 I constructed home ranges for 15 individual turtles (9 females and 6 males). We obtained sufficient relocations to create home ranges for 8 turtles in 2019, 6 in 2020, 10 in 2021, and 10 in 2022. The average number of relocations varied each year with 22.8 in 2019, 27.2 in 2020, 43.5 in 2021, and 47.8 in 2022 (Table 3.1). Limiting the data to turtles with a minimum of 20 relocations resulted in 5 turtles having home ranges constructed each of the 4 years while only 2 years of home ranges were constructed for 4 turtles (Appendix B). The remaining 6 turtles only had the minimum number of relocations within 1 year of the study period.

At Site 2 11 individual turtles (5 females and 6 males) had enough relocations through telemetry to construct home ranges. I was able to estimate home ranges for 6 turtles in 2019, 8 in 2020, 9 in 2021, and 10 in 2022. The average number of relocations at Site 2 also increased each year with 23.7 in 2019, 25.4 in 2020, 46.3 in 2021, and 51.2 in 2022 (Table 3.1Table 3.1). Out of the 11 turtles found I was able to produce a home range each year for 5 individuals, 2 individuals had home ranges constructed for 3 years, 3 turtles had 2 years, and 1 turtle only had 1 year with a home range constructed (Appendix A).

Table 3.1: Minimum, maximum, and average number of relocations of Bog Turtles per year at each site.

Site	Year	Minimum	Maximum	Average
1	2019	20	25	22.8
	2020	26	29	27.2
	2021	30	47	43.5
	2022	34	51	47.8
2	2019	22	25	23.7
	2020	20	28	25.4
	2021	43	48	46.3
	2022	49	52	51.2

At Site 1 95% MCP home range area ranged from a minimum of 0.01 ha to a maximum of 2.02 ha (

Table 3.2, Figure 3.1, Appendix B). Across all study years at Site 1, males had an average 95% MCP home range of 0.64 ha and the average for females was 0.61 ha (

Table 3.2). Kernel density home range size ranged from 0.34 ha, recorded in 2021, to 5.08 ha, recorded in 2020, at Site 1 (Table 3.3, Figure 3.2, Appendix C). For males, the average KDE home range was 1.66 ha (Table 3.3). Females had an average KDE home range size of 1.65 ha across all years.

At Site 2 the range of 95% MCP home ranges were from a minimum of 0.04 ha to a maximum of 0.69 ha (

Table 3.2, Figure 3.1, Appendix B). The average 95% MCP home range size for males was 0.24 ha and females had an average of 0.30 ha across all study years (

Table 3.2). KDE home range sizes varied from 0.29 ha to 2.32 ha (Table 3.3, Figure 3.2, Appendix C). For males, the average KDE home range size was 0.56 ha (Table 3.3). Females at Site 2 had an average KDE home range of 0.67 ha.

Table 3.2: Minimum, maximum, and average 95% MCP home range size of Bog Turtles by site, sex, and year with the standard deviation.

Site	Sex	Year	n	Min	Max	Average (ha)	Standard Deviation
1	Male	2019	5	0.01	0.62	0.23	0.26
		2020	4	0.12	1.92	0.74	0.85
		2021	5	0.20	1.84	0.87	0.85
		2022	5	0.20	1.66	0.73	0.72
	Female	2019	3	0.07	0.72	0.46	0.35
		2020	2	0.34	0.52	0.43	0.13
		2021	5	0.10	0.89	0.50	0.36
		2022	5	0.05	2.02	0.88	0.81
2	Male	2019	4	0.04	0.20	0.11	0.07
		2020	4	0.12	0.53	0.25	0.19
		2021	5	0.09	0.47	0.29	0.16
		2022	6	0.13	0.57	0.28	0.16
	Female	2019	2	0.11	0.40	0.26	0.21
		2020	4	0.14	0.42	0.27	0.12
		2021	4	0.15	0.69	0.39	0.22
		2022	4	0.16	0.43	0.25	0.12

Table 3.3: 95% Kernel density estimation (KDE) minimum, maximum, average, and standard deviation by site, sex, and year of study. All KDE values are reported in hectares.

Site	Sex	Year	Minimum	Maximum	Average	Standard Deviation
1	Female	2019	1.57	3.34	2.62	0.93
		2020	2.58	4.09	3.34	1.07
		2021	0.34	1.62	0.81	0.50
		2022	0.57	1.75	1.24	0.55
	Male	2019	1.16	2.73	1.90	0.65
		2020	1.82	5.08	3.19	1.52
		2021	0.51	1.00	0.68	0.19
		2022	0.78	1.97	1.19	0.57
2	Female	2019	0.37	0.47	0.42	0.07
		2020	0.39	0.66	0.51	0.11
		2021	0.38	0.75	0.59	0.16
		2022	0.42	2.32	1.03	0.88
	Male	2019	0.29	0.56	0.39	0.12
		2020	0.37	0.75	0.51	0.16
		2021	0.29	0.69	0.52	0.16
		2022	0.39	1.52	0.75	0.42

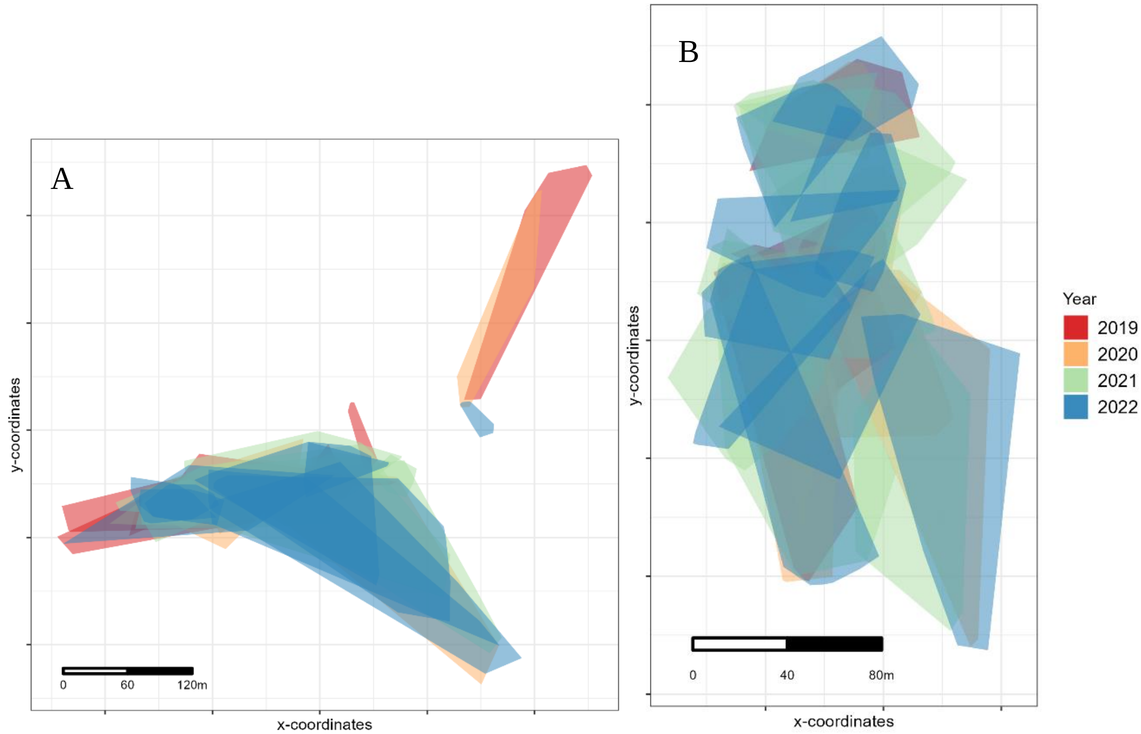


Figure 3.1: Site composite of 95% minimum convex polygon (MCP)home ranges for A) Site 1, and B) Site 2 in 2019–2022.

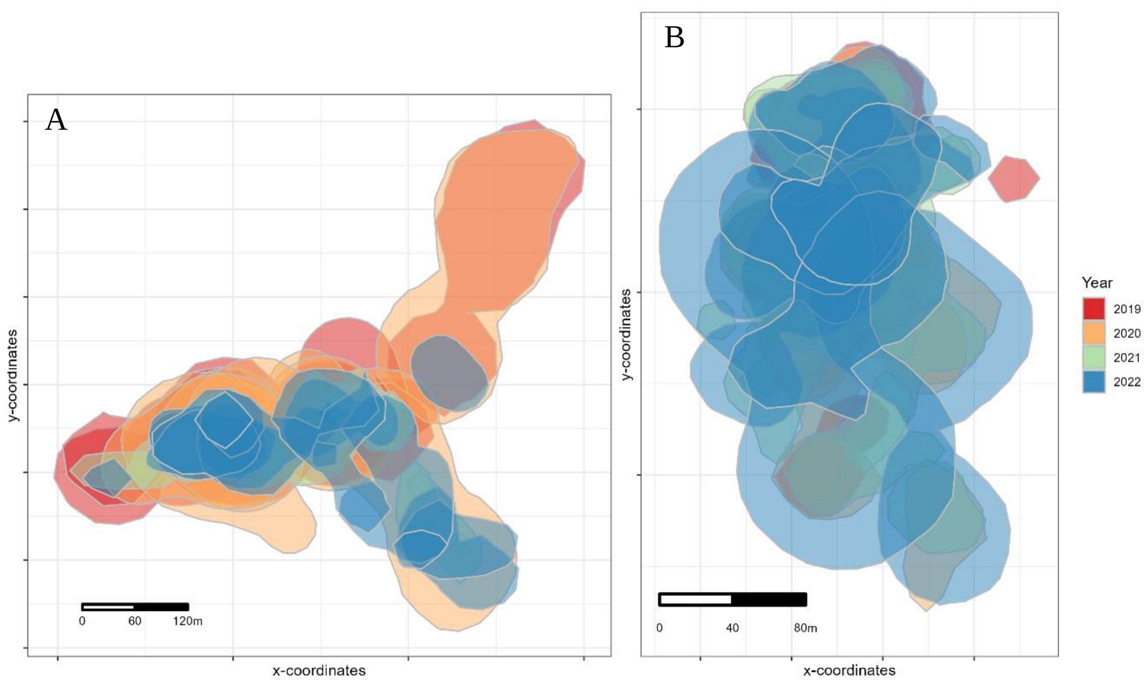


Figure 3.2: Site composite of 95% kernel density estimate (KDE) for all turtles at A) Site 1, and B) Site 2 based on the year of capture.

Using generalized linear models and mixed effects models to analyze the relationship between 95% MCP home range size and the variables of sex and year resulted in 8 models per site (Table 3.4). In the mixed effects models, individual turtle ID was used as the random effect since some turtles were found throughout multiple years. At Site 1 no models were found to be significantly different from each other through likelihood ratio testing using ANOVA. Analysis of AIC values showed that the mixed effects models with year only and the addition of year and sex were the best (Table 3.4). At Site 2 likelihood ratio testing revealed that the mixed effects model that does not include year was significantly different than the full model (p=0.315), indicating that the full model is better at explaining the home range size than the reduced model. The analysis of each model's fit through AIC showed that the same models as Site 1 were best supported (Table 3.4).

Table 3.4: Generalized linear models (glm) and mixed effects models (lmer) with turtle ID as the random effect were used to test for significance in 95% minimum convex polygon (MCP) home range during Study Period 3. Akaike information criterion (AIC), delta AIC, relative likelihood (*L*), and model weight (ω_i) values are reported for each site independently.

	Site 1				Site 2			
Candidate Models	AIC	Δ AIC	*L*	ω_i	AIC	Δ AIC	*L*	ω_i
lmer(area ~ year + (1|id))	63.66	0.00	1.00	0.50	-37.82	0.00	1.00	0.58
lmer(area ~ sex + year + (1|id))	65.65	1.99	0.37	0.19	-36.07	1.75	0.42	0.24
lmer(area ~ sex + (1|id))	65.84	2.18	0.34	0.17	-32.96	4.86	0.09	0.05
lmer(area ~ sex * year + (1|id))	67.94	4.28	0.12	0.06	-34.80	3.02	0.22	0.13
glm(area ~ sex)	68.54	4.88	0.09	0.04	-24.40	13.42	0.00	0.00
glm(area ~ year)	69.41	5.75	0.06	0.03	-24.03	13.79	0.00	0.00
glm(area ~ sex + year)	71.30	7.64	0.02	0.01	-22.98	14.84	0.00	0.00
glm(area ~ sex * year)	75.50	11.84	0.00	0.00	-18.77	19.05	0.00	0.00

The same structure of candidate models was used to test for statistical significance in 95% KDE home range sizes (Table 3.5). Likelihood ratio testing revealed that at Site 1 the mixed effects model containing the interaction of sex and year was significantly better at explaining the home range size than the generalized linear model with year only ($p<0.001$). The full model is also better at explaining the data than the year component of the glm with the addition of sex and year ($p<0.001$), and the year component of the glm with the interaction of sex and year ($p<0.001$). This means that the interaction of sex and year in a mixed effects model is better at explaining the home range size than the glms and adding in the interaction of sex and year is stronger than having only sex. Analysis of the AIC results revealed that the mixed effects model with year only was most supported (Table 3.5) at Site 1. Site 2 likelihood ratio testing revealed that the mixed effects model that has the addition of sex and year was significantly better than the mixed effects model with sex only ($p=0.034$), indicating that adding in the year strengthens the model. The AIC analysis showed that the best supported model was the generalized linear model with year as the only explanatory variable (Table 3.5).

Table 3.5: Generalized linear models (glm) and mixed effects models (lmer) with turtle ID as the random effect were used to test for significance in 95% kernel density estimation (KDE) home range during Study Period 3. Akaike information criterion (AIC), delta AIC, relative likelihood (*L*), and model weight (ω_i) values are reported for each site independently.

	Site 1				Site 2			
Candidate Models	AIC	Δ AIC	*L*	ω_i	AIC	Δ AIC	*L*	ω_i
lmer(area ~ year + (1|id))	81.19	0.00	1.00	0.33	32.72	1.03	0.60	0.21
glm(area ~ year)	81.38	0.20	0.91	0.30	31.68	0.00	1.00	0.36
glm(area ~ sex + year)	82.34	1.16	0.56	0.18	32.87	1.19	0.55	0.20
lmer(area ~ sex + year + (1|id))	82.61	1.43	0.49	0.16	34.21	2.53	0.28	0.10
glm(area ~ sex * year)	87.11	5.92	0.05	0.02	37.91	6.22	0.04	0.02
lmer(area ~ sex * year + (1|id))	87.33	6.14	0.05	0.02	38.70	7.01	0.03	0.01
glm(area ~ sex)	111.00	29.81	0.00	0.00	34.88	3.20	0.20	0.07
lmer(area ~ sex + (1|id))	112.55	31.36	0.00	0.00	36.88	5.20	0.07	0.03

Chi-square analysis of the point habitat variables revealed that the dominant vegetation at known turtle locations was different than random locations at the 1m scale at Site 1 (p=0.003, Table 3.6). Skunk cabbage (*Symplocarpus foetidus*) and ferns were recorded only at the known locations, while blueberry (*Vaccinium spp.*), red-osier dogwood (*Cornus sericea L. ssp. sericea*), and sphagnum moss were only recorded at the paired random locations. There was no significant difference between random points and those with turtles for wetland type (1m and 5m scale), the number of visible cut stumps, and the number of trees counted (basal area factor of 5 or 10) at Site 1. Analysis of variables at Site 2 was limited to dominant vegetation (1m and 5m scale) due to the homogeneity of habitat variables across the site. All wetland types at Site 2 were emergent except for 1 location at the 5m scale, no visible cut stumps were observed, and only one paired location had a tree large enough to be counted as ‘in’ using the Cruz All basal area factor 5.

Table 3.6: Results from Chi-square analysis comparing known turtle location dominant vegetation and paired random location dominant vegetation at the 1m scale for each site. Not applicable (NA) results are due to habitat homogeneity across the site, preventing the ability to compare habitat variables. All habitat variables had the same number of points (n = 30) collected for the known and paired random points.

	Site 1					Site 2				
	Known		Random			Known		Random		
Variable	Mean	Std. Dev.	Mean	Std. Dev.	P	Mean	Std. Dev.	Mean	Std. Dev.	P
Cut Stumps	0.23	0.57	0.60	1.40	0.998	0.00	0.00	0.00	0.00	NA
Cruz All 5	2.60	3.11	2.37	2.16	0.417	0.00	0.00	0.03	0.18	NA
Cruz All 10	1.43	1.92	1.23	1.63	0.716	0.00	0.00	0.00	0.00	NA
1m Dominant Vegetation					0.003					0.713
5m Dominant Vegetation					0.632					0.106
1m Wetland Type					0.947					NA
5m Wetland Type					0.677					NA

3.3.2 Objective 2

Throughout the three study periods, the overall number of turtles that had at least 20 observations per study period and the number of relocations obtained varied. At Site 1, study period 2 had the most individuals (24) and period 3 had the least (14), but period 3 had double the relocations as period 1 (Table 3.7). At Site 2 study period 1 had the greatest number of individuals (n=18) and period 3 had the least (11), but the number of relocations in period 3 was greater than two times as many as period 1 and three times as many as period 2 (Table 3.7).

Table 3.7: Total number of individual turtles that home ranges were created for and the number of relocations collected throughout the three study periods (1= 1994–1997, 2= 2005–2009, and 3= 2019–2022).

Site	Study Period	Individuals	Male	Female	Relocations
1	1	17	6	11	548
	2	24	10	14	785
	3	14	6	8	1168
2	1	18	7	11	534
	2	15	3	12	445
	3	11	6	5	1324

In the mixed effects models, individual turtle ID was used as the random effect since some turtles were found throughout multiple years. All models were directly compared. At Site 1 the 95% MCP home range size did not vary significantly between study periods (1= 0.42 ha, 2= 0.33 ha, 3= 0.49 ha; Table 3.8, Appendix D). Model comparison with likelihood ratio testing revealed no statistical significance for the effect of sex, study period, or the interactions of the variables. Analysis of the models' AIC values revealed that the mixed effect model with sex only was the top-performing model (Table 3.9). Site 2 95% MCP home range size averaged 0.62 ha in study period 1, 0.38 ha in study period 2, and 0.47 ha in study period 3 (Table 3.8, Appendix D). Likelihood ratio testing of mixed effects models revealed that the model that includes the addition of sex and study is significantly different than the mixed effects model that only includes sex ($p = 0.033$). There was no significant difference among any other models using likelihood ratio testing. Opposite of Site 1, AIC analysis of the candidate models revealed that the mixed effects model that includes study only has the lowest AIC value at Site 2 (Table 3.9).

Table 3.8: 95% MCP home range average, minimum, maximum, and sample size (n) for each study period by sex and site.

Site	Sex	Study Period	n	Average (ha)	Minimum (ha)	Maximum (ha)
1	Female	1	11	0.55	0.12	1.45
		2	14	0.28	0.03	1.12
		3	8	0.49	0.04	1.16
	Male	1	6	0.19	0.05	0.54
		2	10	0.41	0.07	1.69
		3	6	0.49	0.01	0.94
2	Female	1	11	0.61	0.13	2.35
		2	12	0.44	0.12	1.13
		3	5	0.50	0.21	0.75
	Male	1	7	0.63	0.23	1.36
		2	3	0.15	0.06	0.21
		3	6	0.45	0.19	1.03

Table 3.9: Generalized linear models (glm) and mixed effects models (lmer) to explain 95% MCP home range size (area) by sex and study period (study). Turtle ID was used as the random effect in the mixed effects models. Akaike information criterion (AIC), delta AIC, relative likelihood (L), and model weight (ω_i) values reported for each site independently.

Candidate Models	AIC	Δ AIC	L	ω_i	AIC	Δ AIC	L	ω_i
lmer(area ~ sex + (1|id))	60.10	0.00	1.00	0.38	50.92	4.33	0.11	0.06
lmer(area ~ study + (1|id))	61.57	1.46	0.48	0.18	46.59	0.00	1.00	0.54
glm(area ~ sex)	62.06	1.96	0.38	0.14	52.15	5.56	0.06	0.03
glm(area ~ study)	62.86	2.76	0.25	0.10	51.38	4.79	0.09	0.05
lmer(area ~ sex + study + (1|id))	63.56	3.46	0.18	0.07	48.12	1.52	0.47	0.25
lmer(area ~ sex * study + (1|id))	63.83	3.73	0.16	0.06	51.67	5.08	0.08	0.04
glm(area ~ sex + study)	64.68	4.57	0.10	0.04	52.97	6.38	0.04	0.02
glm(area ~ sex * study)	64.91	4.81	0.09	0.03	56.02	9.42	0.01	0.00

The average KDE home range size at Site 1 was 0.68 ha in study period 1, 0.65 ha in study period 2, and 0.48 ha in study period 3 (Table 3.10, Appendix D). Site 1 likelihood ratio tests of candidate models did not reveal significant differences in the models' abilities to fit the data when testing the models against the interaction term mixed effects model, the addition of sex and study mixed effects model, nor the interaction glm. AIC analysis revealed that the mixed effects model with only the study period as the explanatory variable best fit the data (

Table 3.11).

At Site 2 the average KDE home range size was 0.86 ha in study period 1, 0.69 in study period 2, and 0.53 in study period 3 (Table 3.10, Appendix D). Site 2 likelihood ratio testing of the candidate models revealed that the full mixed effect model is significantly better at fitting the data than the glm with study only ($p<0.001$), both glm models with sex and study (addition $p<0.001$ and interaction <0.001) in the study term, and the mixed effects model with sex only ($p<0.001$). AIC analysis of Site 2 models revealed that the mixed effects model with the only study period, not including sex, as explanatory variables had the lowest AIC value (

Table 3.11).

Table 3.10: 95% KDE home range average, minimum, and maximum per study, sex, and site.

Site	Sex	Study Period	Average (ha)	Minimum (ha)	Maximum (ha)
1	Female	1	0.79	0.35	1.45
		2	0.64	0.30	1.59
		3	0.52	0.17	1.42
	Male	1	0.47	0.24	0.76

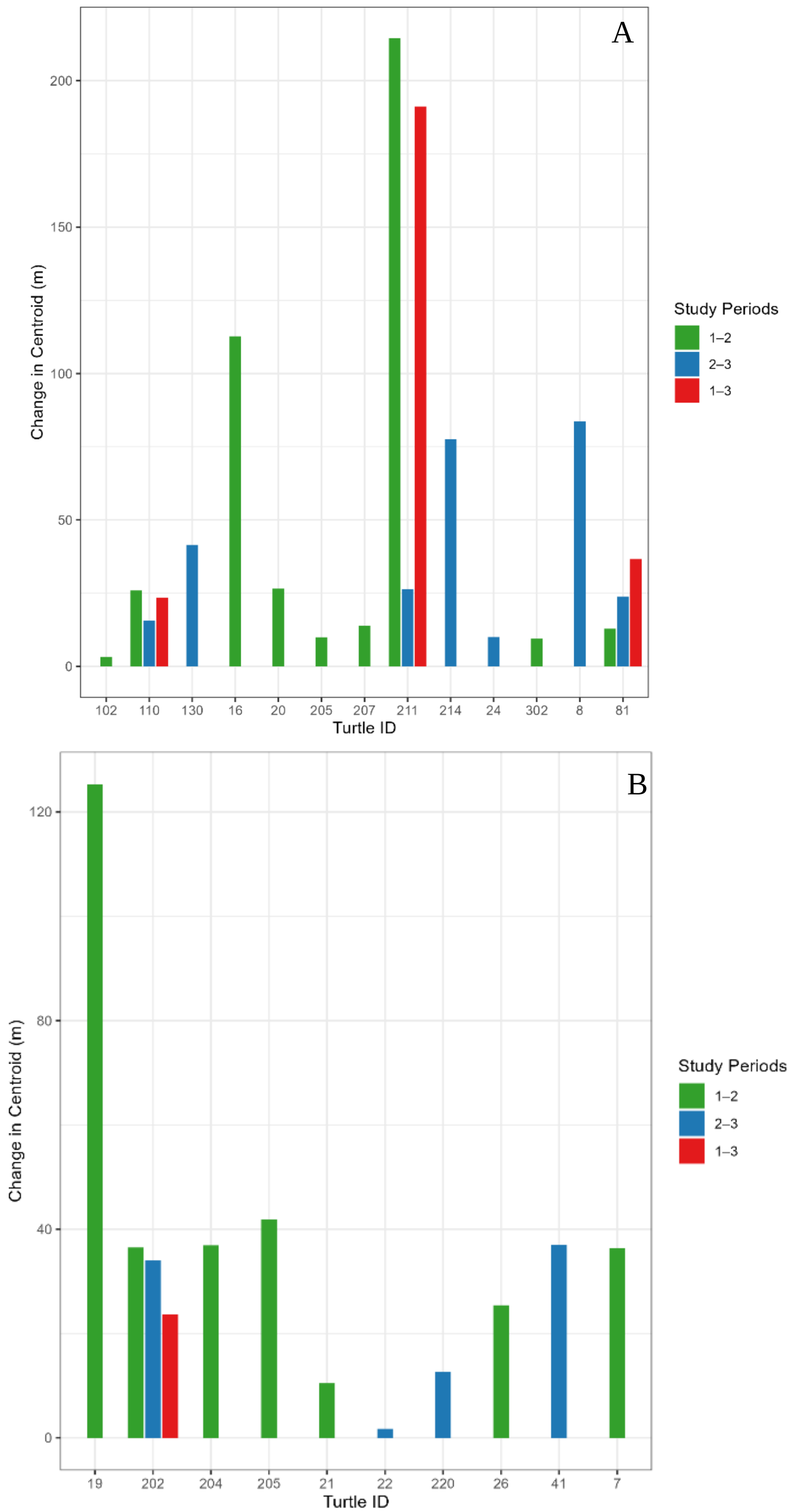
		2	0.65	0.35	1.43
		3	0.43	0.09	0.69
2	Female	1	0.81	0.45	1.29
		2	0.73	0.39	1.33
		3	0.56	0.34	0.69
	Male	1	0.94	0.55	1.14
		2	0.52	0.37	0.63
		3	0.51	0.37	0.72

Table 3.11: Mixed effects models (lmer) with turtle ID as the random effect and generalized linear models (glm) to test for significance in 95% KDE home range across three study periods. Akaike information criterion (AIC), delta AIC, relative likelihood (*L*), and model weight (ω_i) values reported for each site.

	Site 1				Site 2			
Candidate Models	AIC	Δ AIC	*L*	ω_i	AIC	Δ AIC	*L*	ω_i
lmer(area ~ study + (1|id))	33.14	0.00	1.00	0.36	-5.73	0.00	1.00	0.53
lmer(area ~ sex + study + (1|id))	34.28	1.14	0.56	0.20	-3.74	1.99	0.37	0.20
lmer(area ~ sex + (1|id))	35.32	2.18	0.34	0.12	11.37	17.10	0.00	0.00
lmer(area ~ sex * study + (1|id))	35.87	2.73	0.26	0.09	-1.61	4.12	0.13	0.07
glm(area ~ sex)	36.18	3.04	0.22	0.08	9.47	15.20	0.00	0.00
glm(area ~ study)	36.81	3.67	0.16	0.06	-2.71	3.02	0.22	0.12
glm(area ~ sex + study)	36.97	3.83	0.15	0.05	-0.73	5.00	0.08	0.04
glm(area ~ sex * study)	38.24	5.10	0.08	0.03	-0.80	4.93	0.08	0.05

Shift in 95% MCP home range centroids were calculated for 23 individual turtles across both sites, 13 at Site 1 and 10 at Site 2 (Figure 3.3). The average centroid shift at Site 1 from study period 1 to study period 2 was 47.7 m. Between study period 2 and study period 3, the average shift was 39.8 m, and the change in home range centroids between study period 1 and study period 3 was 83.8m. At Site 2 the average centroid shift between study period 1 and study period 2 was 44.7 m, the shift between study period 2 and study period 3 was 21.3 m, and between study period 1 and study period 3 there was only 1 turtle that had enough points to create home ranges in each year and it had a centroid change of 23.7 m.

Figure 3.3: 95% minimum convex polygon (MCP) home range centroid shifts of individual turtles at Site 1 (A) and Site 2 (B). Centroid shifts are measured between two study periods.



3.4 Discussion

3.4.1 Differences in Bog Turtle home range sizes among sites

Home range size estimates reported in this study fall within the range of yearly home range sizes per turtle reported in other studies (Table 3.12) (Pittman and Dorcas 2009, Sirois 2011, Byer et al. 2017, Roos and Maret 2018, McCoy et al. 2021). Many variables may influence the variation in home range sizes reported including the size of the site available, the method by which the home ranges are calculated, and the number of relocations that are included within the datasets (Bekoff and Mech 1984, Seaman et al. 1999, Springer 2003, Nilsen et al. 2008). Roos and Maret (2018) studied 5 sites that ranged from 2.1 ha to 12.3 ha in Pennsylvania while Pittmann and Dorcas (2009) looked at one site that was 0.6 ha in size. While we cannot control this variation in site size, we can limit our comparison to only studies that use the same method of estimation and a similar number of location points.

Table 3.12: Average Bog Turtle home range sizes reported in the literature by sex for 95% Minimum Convex Polygon and 95% Kernel Density Estimate analysis type.

Author	Analysis	Sex	Mean (SE) ha	Site & Year
Roos & Maret 2019	95% MCP	Female	1.01 (± 1.63)	2017–2018
McCoy et al. 2021		Female	0.70 (± 0.13)	2015
Pittman and Dorcas 2009		Female	0.08 (± 0.01)	2007
Sirois 2011		Female	0.25 (± 0.02)	Site 1 2009
Sirois 2011		Female	0.42 (± 0.03)	Site 2 2009
Sirois 2011		Female	0.54 (± 0.04)	Site 1 1997
Sirois 2011		Female	0.54 (± 0.07)	Site 2 1997
Roos & Maret 2018	95% MCP	Male	0.31 (+±0.25)	2017–2018
McCoy et al. 2021		Male	0.37 (± 0.26)	2015
Pittman and Dorcas 2009		Male	0.16 (± 0.01)	2007
Sirois 2011		Male	0.41 (± 0.05)	Site 1 2009
Sirois 2011		Male	0.28 (± 0.04)	Site 2 2009
Sirois 2011		Male	0.19 (± 0.03)	Site 1 1997
Sirois 2011		Male	0.52 (± 0.10)	Site 2 1997
Roos & Maret 2018	95% KDE	Female	1.40 (± 1.82)	2017–2018
Sirois 2011		Female	0.29 (± 0.01)	Site 1 2009
Sirois 2011		Female	0.36 (± 0.01)	Site 2 2009
Byer et al. 2017		Female	0.57 (± 0.69)	BA030 2013
Byer et al. 2017		Female	0.62 (± 0.56)	BA030 2014
Byer et al. 2017		Female	0.30 (±0.14)	HA411 2013
Byer et al. 2017		Female	0.29 (± 0.19)	HA411 2014
Roos & Maret 2018	95% KDE	Male	0.51 (± 0.33)	2017–2018
Sirois 2011		Male	0.32 (± 0.01)	Site 1 2009
Sirois 2011		Male	0.34 (± 0.02)	Site 2 2009

MCP estimation for home range size has been critiqued for potential overestimation and unpredictable bias in estimates (Lawson and Rodgers 1997, Litzgus and Mousseau 2004, Grgurovic and Sievert 2005, Börger et al. 2006, Nilsen et al. 2008). Due to the historical prominence of MCP estimation, it is still a valid choice in home range estimation, particularly when comparing values to older studies (Litzgus and Mousseau 2004, Nilsen et al. 2008). Research has documented, though, that to reduce bias in the estimation of home range size KDE is necessary as unused portions of the area are not included in the calculation (Seaman and Powell 1996, Börger et al. 2006). Many studies, especially those that seek to draw comparisons with studies completed decades ago, rely on the use of KDE to provide unbiased home range estimates in conjunction with MCP for comparison (Sirois 2011, Smith and Cherry 2016, Roos and Maret 2018, McCoy et al. 2021).

Independent of the calculation method, the number of locations used to calculate home range size influences the overall variation in home range size (Börger et al. 2006, Nilsen et al. 2008). Once a threshold is reached in the number of locations the home range size stabilizes and does not vary upon additional locations as the animal's territory has been established (Börger et al. 2006, Nilsen et al. 2008, Morollón et al. 2022). The minimum number of relocations selected for this study (20) is similar to that of other studies of Bog Turtles, ranging from 15–35 points per turtle (Carter et al. 1999, Whitlock 2002, Pittman and Dorcas 2009, Sirois 2011, Byer et al. 2017, McCoy et al. 2021). This number is slightly less than the minimum number of locations suggested by some studies (25–30) to obtain accurate home range size estimates (Bekoff and Mech 1984, Seaman et al. 1999). However, the minimum number of locations varies greatly on the biology of the species in question, the frequency of collection, and the availability of habitat (Bekoff and Mech 1984, Börger et al. 2006, Nilsen et al. 2008, Morollón et al. 2022). Bog Turtles rarely make large movements from the wetlands in which they reside (Pittman and Dorcas 2009, Smith and Cherry 2016), do not leave the wetlands for nesting (Zappalorti et al. 2015, Byer et al. 2018), and populations are commonly disconnected resulting in minimal emigration and immigration (Rosenbaum et al. 2007, Roos and Maret 2018).

With this understanding of the species' natural history, I do not expect that increasing the minimum number of relocations would influence the estimates of home range size.

Home range size in the current study period was found to be best explained by including both year and sex within the models. Within the literature studies have reported significance in home range size by sex with both males and females exhibiting larger home range sizes depending upon the population (Lovich et al. 1992, Whitlock 2002, Pittman and Dorcas 2009). Other studies have found no difference in home range size by sex (Carter et al. 1999, Morrow et al. 2001*b*, McCoy et al. 2021). These contradictory results are potentially due to low sample sizes (Bekoff and Mech 1984, Seaman and Powell 1996, Smith and Cherry 2016), large variability between individual behavior (Grgurovic and Sievert 2005, Börger et al. 2006), large variation in site characteristics (Börger et al. 2006, Smith and Cherry 2016), and short duration of studies (Pittman and Dorcas 2009, Smith and Cherry 2016). As previously discussed, the size of the wetlands inhabited by Bog Turtles varies, and within this variation, the extent of suitable habitat also differs. Variability in Bog Turtle home range size, attributed to fluctuations in suitable habitat, has been demonstrated (Sirois et al. 2014, McCoy et al. 2021), as has the difference in the rate of movement in wetlands of varying sizes (Smith and Cherry 2016). Home range sizes reported from data collected in just one or two years may not accurately represent the typical state of the population, as unusual climatic conditions could occur during those study years (Smith and Cherry 2016). The comparison of the three study periods' home range sizes addresses two of these shortcomings by providing a long-term view of some turtles while also capturing home ranges of different turtles throughout the years, increasing the overall sample size.

3.4.2 Changes in Bog Turtle habitat use with habitat management

Temporal variation was found to be significant in all home range size estimates at Site 2, in both the current study period and the comparison of all three study periods. This variation is largely due to the fluctuations in habitat quality and hydrology throughout the site. As Sirois et al. (2014) described the average home range size at the site drastically decreased between the first and second study periods, following the degradation of habitat due to beavers moving into the site in the early 2000s. This constriction of available habitat shifted the focus of habitat management at the site to mitigating the risk of flooding and restoring habitat that had become inundated with invasive vegetation. Specifically, habitat management focused on maintaining the core area of habitat as it contains essential hibernacula and is the only documented nesting area at the site. As the risk of flooding has been controlled through the active removal of beaver dams, beavers, and the vegetation that supplies the dams, the focus of habitat management has shifted to restoring habitat quality along the periphery of the core use area. The results of this study reflect these efforts in the temporal significance of home range size throughout the three study periods.

The significance of home range size during the current study period is in part attributed to the hydrology at Site 2. The site is heavily influenced by flood and drought periods, with the turtles having to shift use areas in response to these events. Bog Turtles have been documented either increasing their large movements (≥80 m) or remaining inactive in pockets of saturated substrate in response to drought conditions (Feaga 2010).

However, during study period 3 the individual turtles that were on radio changed throughout the four years, so the variation cannot be solely attributed to climatic variations.

At Site 1 temporal variation was not found to significantly influence the home range size of the Bog Turtles. A lack of temporal variation in home range size has been demonstrated in other studies of Bog Turtles (Carter et al. 1999), although this study was only able to evaluate the relationship over two years. The long-term analysis completed in this study indicates that the extent of suitable habitat available to the population has remained similar throughout nearly three decades. This has been possible through continued habitat management focusing on maintaining the suitability of habitat within the known core area and expanding suitable habitat into new areas.

The results of these two sites indicate that by tailoring habitat management efforts to the individual needs of two sites the suitability of habitat can be maintained and improved over time. Many studies have been able to demonstrate how a decrease in habitat quality results in a significant increase in home range size due to turtles having to travel further to access quality habitat (Morrow et al. 2001*b*, Sirois et al. 2014, Byer et al. 2017, Roos and Maret 2018), but none have been able to demonstrate the stabilization of home range size in response to continued habitat management. By analyzing data first collected in 1994 I was able to identify that home range size has not significantly changed at a site where the habitat quality has been maintained over two and a half decades.

Through the analysis of the home range centroid shift of turtles across three decades, I discovered that the timing of the largest shift was dependent upon the site. At

Site 1 the largest shift occurred between the first study period to the third study period. However, there were only 3 turtles whose home range centroids could be tested from the first to the third study period. One of the three turtles had a shift in centroid location of 191.2 m, the second highest shift among any study periods. This female turtle was part of radio telemetry studies during all three study periods and exhibited a large home range shift between study periods 1 and 2, the largest home range shift recorded (Appendix E). This home range shift was likely a result of improved habitat within the core area of the wetland as the turtle shifted from the western portion of the fen to the eastern portion where a prescribed fire occurred and woody vegetation was thinned. There were no centroid changes larger than those exhibited by female 211 within the site.

At Site 2 the largest average home range shift was observed between study periods 1 and 2. This shift corresponds with the rapid loss of suitable habitat that the turtles experienced in the early 2000s due to flooding in the site resulting from beavers. Female 19 recorded the largest home range centroid shift (125.3 m) from the northern portion of the wetland south to what is currently considered the core use area (Appendix F). In the first study period, the turtles were known to use the northern edge of the wetland, however, after the beavers flooded the northeastern edge of the wetland the turtles apparently sought refuge southwest. The turtle with the second largest home range shift (41.8 m) at Site 2, also between study periods 1 and 2, also demonstrated this southwestern shift. One turtle that was located during all three study periods, female 202, was found to have moved south by 36.5 m from study 1 to 2, likely escaping flooding (Appendix G). Now that the flooding risk and density of invasive species have been mitigated, the turtle has shifted its home range centroid northwest by 34.0 m. This current centroid location is only 23.7 m from the centroid in study period 1.

The results of this study indicate that through restoration of degraded habitat and preservation of quality habitat, isolated populations of Bog Turtles can exist. While this study is the longest analysis of Bog Turtle populations to date it still is only able to capture a fragment of a Bog Turtle's lifespan. The genus *Glyptemys* has existed for 17 million years and fossil records of Bog Turtles have been found dating back to the Pleistocene (Ernst and Lovich 2009, Spinks et al. 2016). Before anthropogenic landscape management, it is assumed that Bog Turtles moved from a habitat degraded by natural ecological succession to a habitat created through burning by indigenous peoples, beaver activity, and natural wildfires (Rosenbaum et al. 2007). Research has found that current populations of Bog Turtles are both spatially and genetically isolated, due to the fragmentation of the landscape (Rosenbaum et al. 2007, Dresser et al. 2018). This increases the risk of extinction due to environmental disturbances, low genetic diversity, and inbreeding depression (Rosenbaum et al. 2007, Pittman et al. 2011, Dresser et al. 2018). Nevertheless, these small, isolated populations encompass the remaining genetic diversity for this highly fragmented species. Therefore, the preservation of these populations through the management of the habitat in which they reside is of utmost importance.

3.4.3 Characteristics of Bog Turtle habitat

This study revealed differences in micro-habitat between locations where turtles were present and absent during the active season, confirming findings from other studies that microhabitat influences the location of Bog Turtles within wetlands (Chase et al.

1989, Carter et al. 1999, Pittman and Dorcas 2009, Stratmann et al. 2020). Specifically, I found that the dominant vegetation at the 1m scale consisted of more *Carex* spp., *Alnus* spp., and skunk cabbage (*Symplocarpus foetidus*) at the known points. The paired random points had higher percentages of tamarack (*Larix laricina*), blueberry (*Vaccinium corymbosum*), and red maple (*Acer rubrum*) than was found at the known locations.

Wetland type was not different between Bog Turtle occurrence, but that is likely a result of the dominance by high-quality emergent wetland habitat within both sites in this study. Bog Turtles have been shown to prefer emergent wetland types through the selection of short vegetation, such as sedges and rushes, over vegetation that covers more of the canopy (Chase et al. 1989, Morrow et al. 2001*a*, Tesauro and Ehrenfeld 2007, USFWS 2022). Similarly, the lack of significance in the basal area of standing trees estimated with a Cruz All at both sites is likely a result of the dominance of emergent wetland.

Measuring habitat variables during different seasons may yield different results for all variables. During the fall transition into hibernation, Bog Turtles have shown a reliance on woody vegetation to create pockets of saturated silt for hibernacula (Feaga and Haas 2015). This would impact the significance of dominant vegetation most heavily at Site 1 where turtles are commonly found to use the edge of red maple swamp for hibernation. At Site 2 the availability of woody vegetation is lower resulting in turtles relying on root systems of smaller shrubs. Evaluation of the importance of woody vegetation cover through basal area needs to be refined in future studies. Since Bog Turtles occupy open canopy habitat throughout a large portion of the year the applicability of a Cruz All to determine basal area of standing trees is minimal. This was evident in the results of this study, specifically at Site 2. However, the relationship of Bog Turtles to the basal area is important to understand as habitat management often involves thinning of woody vegetation. Most importantly, future research into the habitat characteristics of Bog Turtle known occupancy points and the paired random points should be completed in conjunction. Completing both sets of data collection at the same time will remove any bias resulting from the seasonality in which data was collected, strengthening the inference of the relationship.

The findings of this study demonstrate that variations in habitat quality, stemming from natural succession, disturbance, and habitat enhancement through management efforts influence the spatial distribution of Bog Turtles within occupied wetlands. Nearly three decades of data show that Bog Turtles will use the core habitat of the wetlands when habitat quality is maintained as well as move into new areas of the wetland not previously known to be used. These results support the need for continued management of quality habitat within Bog Turtle wetlands while also highlighting the necessity for long-term studies of populations to understand the fluctuations in distribution.

CHAPTER 4

CONCLUSION

4.1 Population dynamics and habitat use by Bog Turtles in MA

In this thesis, I provide evidence that long-term analysis of Bog Turtle populations is critical in understanding fluctuations in population demographics and spatial distribution. By comparing two sites with contrasting histories of habitat degradation and management, we can see how the availability of quality habitat influences the populations. Specifically, at Site 1 continued management of suitable habitat was reflected in the lack of temporal fluctuations in population abundance, survival, and home range size estimates. Site 2 demonstrates how the degradation of habitat impacts the population through reduced abundance, survival, and home range size estimates. Since study period 2 site has provided a stable habitat for the Bog Turtles. However, to evaluate whether this positive response of population metrics directly stems from habitat management efforts, more sites are needed for comparison.

The signature of Bog Turtle wetlands is open-canopy vegetation and groundwater-driven hydrology, but within this framework, the unique history and geographic features of occupied wetlands results in varied habitat. Studies have demonstrated how this variation in wetland features impacts occupancy, home range size, abundance, and survival (Sirois et al. 2014, Byer et al. 2017, Stratmann et al. 2020), but few have analyzed how long-term fluctuations in habitat influence these metrics. While this thesis was limited to two populations, the diversity of habitat types, from harsh fen to red maple swamp, at each site allows the findings to apply to other populations of Bog Turtles.

The abundance and home range size estimates reported in this thesis fall within the range of those reported in other studies, indicating that the populations are comparable to other sites (Chase et al. 1989, Shoemaker et al. 2013, Sirois et al. 2014, USFWS 2022). A similar study that compared population abundance from 1997 and 2020 in Virginia attributed temporal variation in population estimates between years to decreased habitat suitability (Holden 2021). Estimates of home range size reported in this thesis align with values observed in other studies (Pittman and Dorcas 2009, Sirois 2011, Byer et al. 2017, Roos and Maret 2018, McCoy et al. 2021). Altered home range size and decreased abundance estimates in response to habitat degradation have been demonstrated across multiple sites (Morrow et al. 2001*b*, Sirois et al. 2014, Stratmann et al. 2020). However, estimates of both abundance and home range size are reflective of the characteristics of the individual sites more than the species.

4.2 Future research needs

Direct and indirect impacts of development, road mortality, altered hydrology, ecological succession of nesting habitat, and influx of invasive vegetation have been recognized as the greatest threats to the northern populations of Bog Turtles (Erb 2019). The results of this study provide evidence that through continued habitat management managers can mitigate some of these threats and provide suitable habitat to sustain and grow populations of Bog Turtles. However, more long-term studies of population monitoring and investigations into fluctuations in abundance and survival are needed to support these findings. While the temporal scope of this thesis, 1994–2022, is the longest timespan published for the species it still is only able to capture a portion of a Bog Turtle's life span. Additionally, the limitation of this study to two sites reduces the interpretation and application of the results. More sites across the species range are needed for accurate comparison. These additional sites should encompass both managed and unmanaged habitats so that the responses of the populations can be compared.

Continued monitoring of the two sites in Massachusetts is critical to capturing generational shifts in population demographics and habitat use. The discovery of new areas being used by turtles at Site 1 suggests that the extent of habitat used by the turtles is expanding. Further investigation is needed on the age distribution of turtles within these new areas to assess whether this is a shift in use area by older turtles or the expansion of use area by younger turtles. During the 2021 and 2022 seasons, juvenile turtles were observed in one of these new areas, providing strong evidence of nesting activity. Finding new nesting habitat would enhance the understating of nesting habitat requirements and potentially alleviate low recruitment rates commonly exhibited by Bog Turtle populations (Zappalorti et al. 2015, 2017, Stratmann et al. 2020).

Research within fragile ecosystems such as wetlands does lead to large levels of disturbance on the site, particularly to the vegetation and silt. As researchers move through the sites, place traps, and look for turtles the vegetation gets damaged. After years of extensive presence in the sites, this damage is easily visible and extends beyond the visible vegetation down into the substrate. The repeated weight of humans on fragile roots causes the structure of the silt to break down and eventually paths in the sites are bare. Intensive studies of the sites are set on a 10-year interval to allow the sites time to recover from the heavy presence of humans and capture new generations of Bog Turtles. This timeframe is appropriate to capture fluctuations in populations but may need to be altered if a disturbance that has the potential to be detrimental to a population occurs. In that instance, close monitoring of the population, while minimizing the impact on the habitat, needs to take precedence as a reduction in adult survival may destabilize a population (Knoerr et al. 2021) regardless of the quality of the habitat.

Subsequent studies should incorporate turtles not previously included in radio telemetry studies completed at the sites. Diversity in the individual turtles tracked would provide the opportunity to capture shifts in the population’s core habitat and discover new areas that were not previously known to be used by the population. The turtles that were tracked throughout the three study periods should not be excluded from future studies as they provide valuable information through comparison. By having turtles that were part of all three study periods I was able to directly compare fluctuations in habitat use. The analysis of home range centroid shifts addressed the question of how the spatial distribution of Bog Turtles within the fens has changed over time; however, the study was not designed to investigate a direct response of Bog Turtles to areas of improved habitat. I recommend that future studies complete pre- and post-treatment monitoring of areas designated for habitat management. These data will allow managers to know if the turtles were already using the area or if they shifted home ranges in response to the availability of new habitats.

The cryptic nature of the Bog Turtle has led many to question whether there are more extant populations than what is currently known. Research has attempted to address this question with habitat distribution models; however, the limited number of known populations often limits the predictive power of these models (Barron II 2021). During the 2020 and 2021 seasons, I completed visual surveys and trapping at eleven other wetlands in Berkshire County, MA that were identified as potentially suitable Bog Turtle habitat within 20 km of the known populations. While these surveys did not produce any Bog Turtles nor signs of the species, they cannot be considered conclusive due to the low probability of detection of the species (Barron II 2021). A limitation in finding new sites is the amount of time that is required to complete traditional survey methods. This issue can be mitigated by implementing the camera trapping methodology developed by Michael Knoerr (2021). Additionally, eDNA may be a useful technique for determining the presence of Bog Turtles with further refinement of methods for use in wetlands (Erb 2019). These passive methods of sampling wetlands for Bog Turtle occupancy would greatly enhance the ability of managers to find new populations.

4.3 Management implications

The results of this thesis have demonstrated the need for continued long-term habitat management tailored to the needs of individual fens for the survival of Bog Turtle populations. Through continued studies on the habitat used by the Bog Turtles within MA sites managers have been able to identify where to focus management efforts. At Site 1 managers have continued to treat and remove invasive vegetation and thin native woody vegetation to reduce ecological succession, both within the wetland and along the perimeter. These efforts have resulted in the extent of suitable habitat being stable, as indicated by the minimal shifts in home range centroids across nearly three decades. At Site 2, habitat management was shaped by the observed loss of habitat use by Bog Turtles in the northeastern portion of the wetland after beavers moved into the site. Management has focused on mitigating the influence of beavers on hydrology and the resulting influx of non-native invasive vegetation. Through years of continued work to restore hydrology through beaver dam removal and installation of flow control devices the wetland has not experienced a potentially detrimental flooding event in recent years. The methods of vegetation control at Site 2 have included treatment of invasive vegetation, mechanical thinning of woody vegetation, use of grazing, and most recently prescribed fire to reduce the density of tall vegetation. The implementation of these efforts will continue to provide quality habitat for the populations of Bog Turtles as long as they continue to be completed with caution. Heavy disturbance management, such as removal of woody vegetation, use of herbicide, and any heavy machinery operation near or in the wetland, is completed while Bog Turtles are in hibernation to minimize any direct impacts. Radio telemetry is used before conducting these management activities to confirm whether the turtles on radio are in hibernacula.

Prioritizing the protection of Bog Turtle wetlands and the surrounding watershed results in critical refugia for many species that rely on fen habitat. Many flora and fauna that benefit from the protection of these lands are threatened including spotted turtles (*Clemmys guttata*), slender cottongrass (*Eriphorum gracile*), and a variety of native bees. The USFWS is currently reviewing the status of spotted turtles in determination for listing under the Endangered Species Act (ESA). Spotted turtle habitat overlaps with Bog Turtles and the species are commonly found in the same wetlands (Ernst 1970, Arndt 1977). Some species of native bumblebees have experienced a 96% decline in relative abundance (Cameron et al. 2011). The protection and management of these open-canopy fens provide critical habitat for a variety of imperiled species, including the Bog Turtle.

APPENDICES

APPENDIX A

Number of relocations for each Bog Turtle used for construction of home ranges during the 2019–2022 study period at each site.

Site	Turtle ID	2019	2020	2021	2022	Total Years
1	8	24	0	0	0	1
	24	25	27	47	51	4
	50	22	0	0	0	1
	55	0	0	0	40	1
	56	24	27	43	50	4
	81	24	26	45	50	4
	94	0	0	46	51	2
	110	0	0	30	0	1
	130	0	0	46	50	2
	131	0	0	43	0	1
	135	20	29	45	50	4
	139	0	0	0	34	1
	152	22	27	45	51	4
	157	21	27	0	0	2
	214	0	0	45	51	2
2	22	24	26	48	52	4
	41	25	27	48	52	4
	46	0	20	0	0	1
	87	25	26	47	49	4
	145	0	0	43	51	2
	162	0	0	43	51	2
	165	24	27	47	52	4
	202	0	28	48	52	3
	211	0	25	46	51	3
	220	22	24	47	52	4
	241	22	0	0	50	2

APPENDIX B

Yearly 95% MCP home range size reported in hectares for each turtle that had at least 20 relocations within each year.

Site	Sex	ID	2019	2020	2021	2022	Average
1	Female	50	0.07				0.07
		55				0.05	0.05
		81	0.61	0.34	0.73	0.95	0.66
		94			0.65	1.21	0.93
		110			0.10		0.10
		131			0.89		0.89
		139				2.02	2.02
		157	0.72	0.52			0.62
		214			0.13	0.17	0.15
	Male	8	0.01				0.01
		24	0.08	0.12	0.20	0.21	0.16
		56	0.37	1.92	1.84	1.66	1.45
		130			0.28	0.20	0.24
		135	0.09	0.78	0.27	0.22	0.34
		152	0.62	0.13	1.77	1.37	0.97
2	Female	46		0.33			0.33
		87	0.11	0.14	0.34	0.16	0.19
		162			0.15	0.18	0.16
		202		0.20	0.39	0.26	0.28
		220	0.40	0.42	0.69	0.43	0.48
	Male	22	0.14	0.17	0.21	0.24	0.19
		41	0.09	0.18	0.23	0.36	0.22
		145			0.44	0.13	0.28
		165	0.20	0.12	0.09	0.16	0.14
		211		0.53	0.47	0.57	0.52
		241	0.04			0.24	0.14

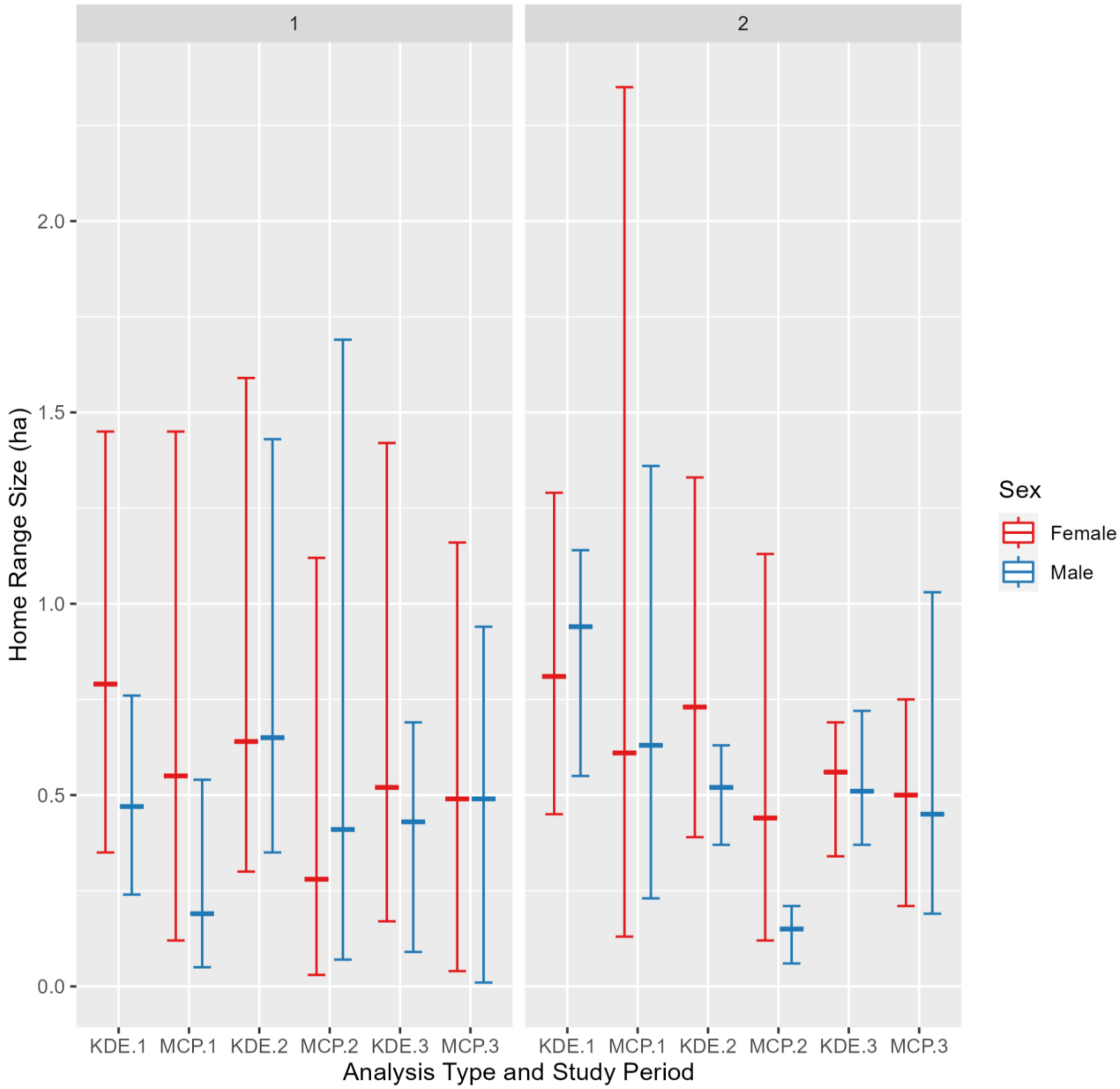
APPENDIX C

Yearly 95% Kernel density estimations of individual turtles throughout the study years. Each turtle had at least 20 relocations within the study year for home range construction.

Site	Sex	ID	2019	2020	2021	2022	Average
1	Female	50	1.57				1.57
		55				0.57	0.57
		81	2.95	2.58	1.62	1.75	2.23
		94			0.74	1.74	1.24
		110			0.34		0.34
		131			0.90		0.90
		139				1.37	1.37
		157	3.34	4.09			3.72
		214			0.45	0.75	0.60
	Male	8	1.16				1.16
		24	1.86	1.82	0.51	0.79	1.24
		56	2.35	5.08	1.00	1.97	2.60
		130			0.64	0.78	0.71
		135	1.40	3.76	0.55	0.78	1.62
		152	2.73	2.10	0.67	1.62	1.78
2	Female	46		0.66			0.66
		87	0.37	0.39	0.57	0.57	0.47
		162			0.38	0.42	0.40
		202		0.53	0.65	0.80	0.66
		220	0.47	0.46	0.75	2.32	1.00
	Male	22	0.37	0.46	0.46	0.62	0.48
		41	0.34	0.47	0.54	0.89	0.56
		145			0.69	0.40	0.54
		165	0.56	0.37	0.29	0.39	0.40
		211		0.75	0.65	1.52	0.97
		241	0.29			0.67	0.48

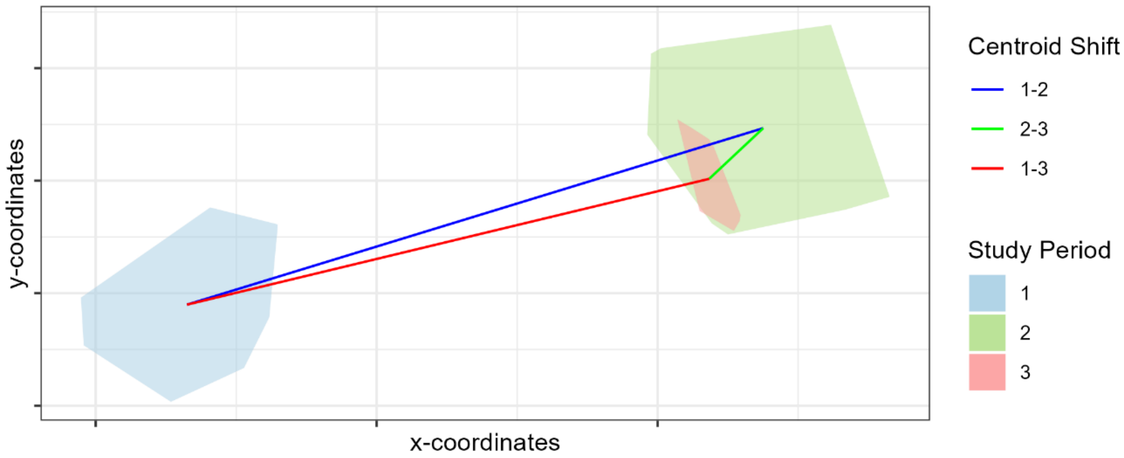
APPENDIX D

Box plot of home range average, minimum, and maximum by site, study period, analysis type, and sex. Sites are separated with Site 1 on the left and Site 2 on the right of the figure. Minimum convex polygon (MCP) and kernel density estimate (KDE) analysis types are indicated on the x-axis with the corresponding study period number.



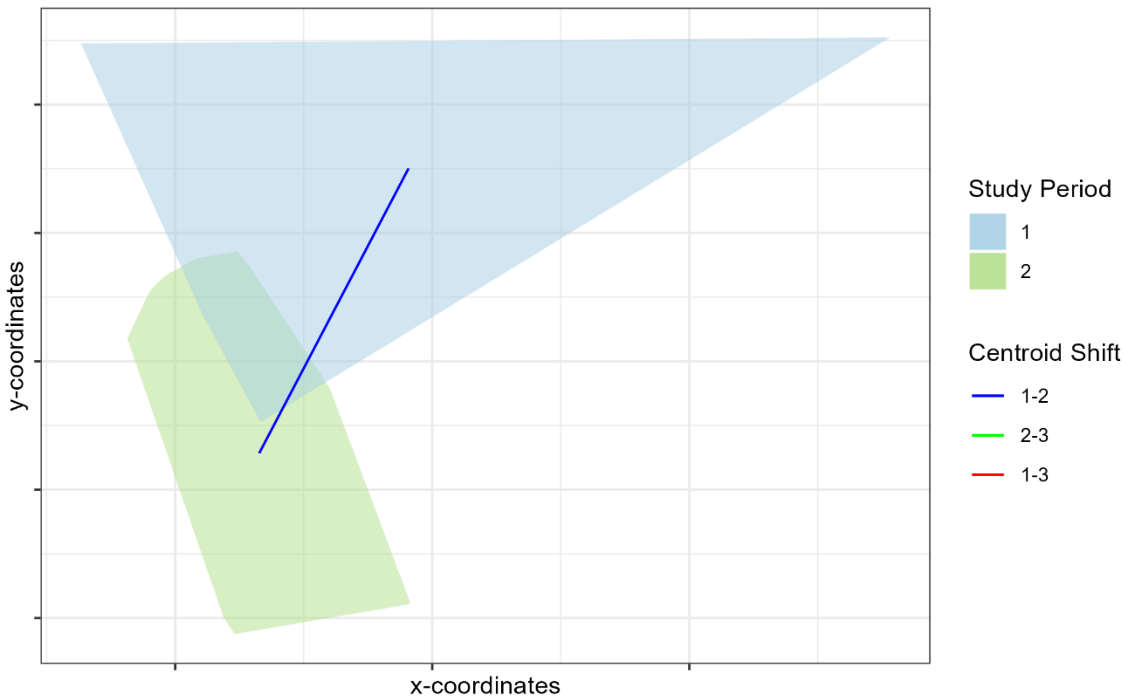
APPENDIX E

Home range centroid shift of female turtle ID 211 at Site 1 throughout three study periods. This turtle was tracked in each of the three study periods. Home range centroid shifted 191.2 m between the first and second study periods.



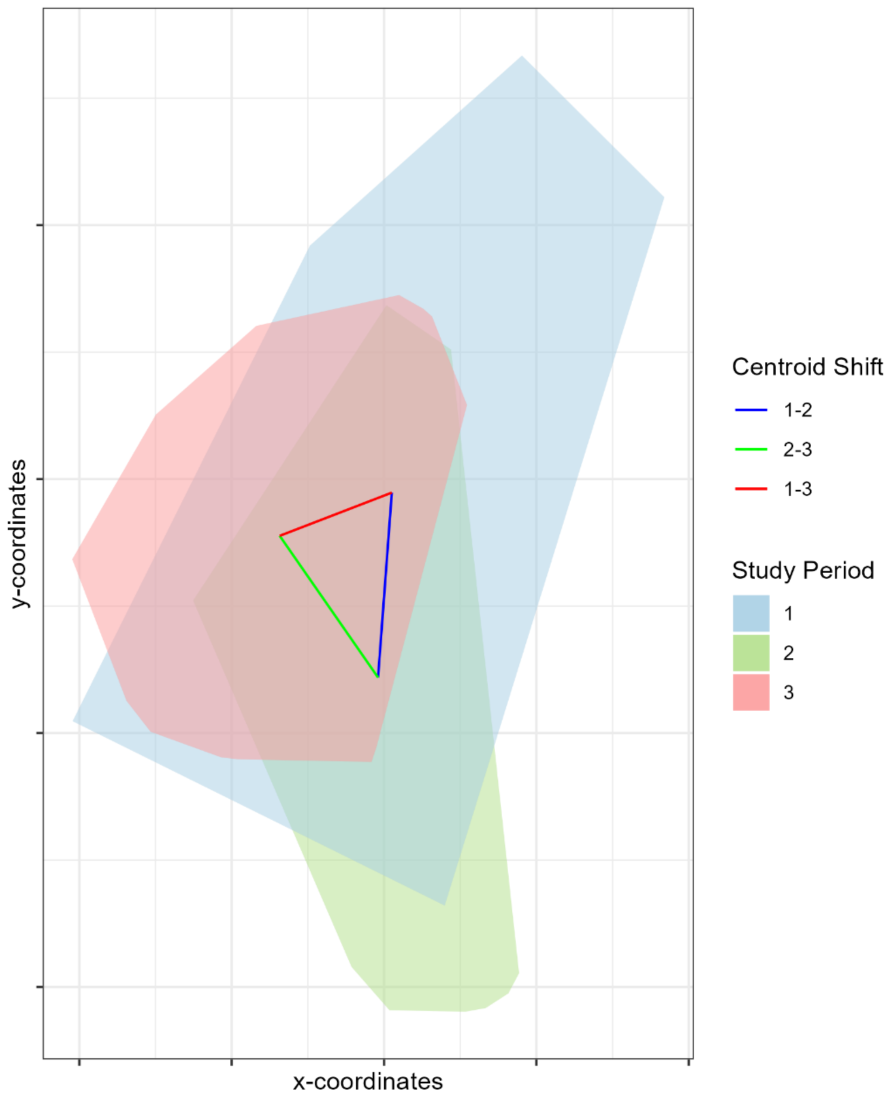
APPENDIX F

Home range centroid shift of female turtle ID 19 at Site 2 from the first to second study period.



APPENDIX G

Home range centroid shift of female turtle ID 202 at Site 2 throughout the three study periods.



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