
STATUS OF A *GLYPTEMYS INSCULPTA* (WOOD TURTLE) POPULATION IN NORTHEASTERN MINNESOTA

MADALINE M. COCHRANE^{1,8}, DONALD J. BROWN^{2,3}, MARK D. NELSON⁴, RICHARD R. BUECH⁵,
MIKE SCHRAGE⁶, DAN RYAN⁷, AND RON A. MOEN¹

¹Natural Resource Research Institute, Biology Department, University of Minnesota–Duluth,
5013 Miller Trunk Highway, Duluth, Minnesota 55811, USA

²West Virginia University, School of Natural Resources, PO Box 6125, 322 Percival Hall,
Morgantown, West Virginia 26506, USA

³U.S. Forest Service, Northern Research Station, Parsons, West Virginia 26287, USA

⁴U.S. Forest Service, Northern Research Station, St. Paul, Minnesota 55108, USA

⁵U.S. Forest Service (Retired), Forestry Sciences Lab, Grand Rapids, Minnesota 55744, USA

⁶Fond du Lac Resource Management Division, Cloquet, Minnesota 55720, USA

⁷U.S. Forest Service, Laurentian Ranger District, Aurora, Minnesota 55705, USA

⁸Corresponding author, email: cochr081@umn.edu

Abstract.—Wood Turtles (*Glyptemys insculpta*) have experienced population declines across their North American range and are state-listed as threatened in Minnesota, USA. To improve our understanding of the current conservation status of one population in northeastern Minnesota, we: (1) performed a snapshot comparison of population structure and relative abundance in 1990 and 2015 using survey data at 12 sites; (2) estimated the population growth rate (λ) from 1997–2014 based on population monitoring data at six sites; (3) estimated change in abundance between 2016 and 2017 based on population monitoring data at eight sites; and (4) performed a population reconstruction to estimate the minimum *Glyptemys insculpta* alive each year and geometric mean of λ from 1990 to 2017. The snapshot comparison indicated that relative abundance, adult sex ratio, and juvenile:adult ratio did not significantly differ between years. Captures/hour was higher in 1990 than in 2015. Mean λ from six sites monitored from 1997–2014 was 1.016, indicating the population was stable over that period. However, abundance estimates from surveys at eight sites in 2016 and 2017 indicated a substantial decrease from 247 to 112 individuals. The population reconstruction estimated a mean λ of 1.007 and 0.970 from 1990–2005 and 2006–2017, respectively. In northeastern Minnesota, *Glyptemys insculpta* exists in a forested landscape with predominantly public ownership and little development pressure, likely avoiding many anthropogenic stressors. However, we obtained equivocal results for our population status assessment. Continued monitoring is necessary to understand the trajectory of this *Glyptemys insculpta* population.

Key Words.—conservation; monitoring; relative abundance; snapshot; survey

INTRODUCTION

Glyptemys insculpta (Wood Turtle; Fig. 1) is a semi-aquatic freshwater turtle endemic to northeastern North America. In many parts of its range, *G. insculpta* have experienced population declines (Garber and Burger 1995; Daigle and Jutras 2005; Willoughby et al. 2013). It is listed as a threatened species in the state of Minnesota, USA (Moriarty and Hall 2014), considered endangered globally (International Union for the Conservation of Nature [IUCN]. 2016. The IUCN Red List of Threatened Species. Available from <http://www.iucnredlist.org/details/4965/0> [Accessed 12 November 2017]), and is currently under review for listing under the Endangered Species Act in the United States (U.S. Fish and Wildlife Service [USFWS]. 2016. Species Profile for Wood Turtle (*Glyptemys insculpta*). Available from [\[ecos.fws.gov/ecp0/profile/speciesProfile?spcode=C06A\]\(https://ecos.fws.gov/ecp0/profile/speciesProfile?spcode=C06A\) \[Accessed 12 November 2017\]\).](https://</p></div><div data-bbox=)

Glyptemys insculpta in Minnesota represent the westernmost population in the range of the species. The distribution of *G. insculpta* in northeastern Minnesota is discontinuous across midsize rivers of two watersheds that flow through forested regions (Moriarty and Hall 2014). The population is likely isolated from *G. insculpta* populations to the south and east because of the disjunct distribution of sandy glacial outwash, which underlies optimal riverine habitat for this species in the northern Great Lakes Region (Buech et al. 1997). In addition, *G. insculpta* in northern Minnesota occur in the Lake Superior watershed, whereas southern populations are in the Mississippi watershed. Isolated populations and those at distributional limits are typically more vulnerable to extirpation than connected populations



FIGURE 1. Photograph of an adult Wood Turtle, *Glyptemys insculpta*. (Photographed by Madaline Cochrane).

and those in core portions of the distribution (Henle et al. 2004; Cushman 2006; Yackulic et al. 2011). Despite its overall threatened status in Minnesota and likely added vulnerability to extirpation due to its isolation from other Minnesota and Wisconsin populations, the status of the *G. insculpta* population in northeastern Minnesota is currently unknown.

As a species with low fecundity and delayed reproductive maturity, even minimal additive mortality of reproductive adults each year can precipitate extirpation in small and isolated populations (Compton 1999). It is currently believed that terrestrial habitat loss from land-use conversion and human recreation, habitat degradation from mesopredator population increase and fire suppression, and direct human impacts from road mortality are negatively impacting many *G. insculpta* populations (Harding and Bloomer 1979; Garber and Burger 1995; Buech et al. 1997; Jones et al. 2015). While not previously recorded in northeastern Minnesota (Maya Hamady, pers. comm.), collection for the commercial pet trade could also contribute to *G. insculpta* declines (Levell 2000). Threats found in other parts of the range from altered stream flow and insufficient sandbar availability during the nesting season are not currently present in the northern watersheds of Minnesota (Lenhart et al. 2013). However, projected climate change in the region includes an increase in extreme events, such as increased storm frequencies and more pronounced flood and drought intensities (Wisconsin Department of Natural Resources 2016), which could negatively impact adult survival, nest success, and habitat quality.

State agencies in the Upper Midwest of the U.S. (i.e., Minnesota, Wisconsin, Michigan, and Iowa) are currently engaged in research and management actions to identify threats to *G. insculpta* populations, increase *G. insculpta* recruitment and survivorship, and enhance *G. insculpta* habitat quality. As part of this larger conservation initiative, we sought to determine

the status of the species in northeastern Minnesota. We used population monitoring data collected between 1990 and 2017 to perform four abundance change analyses: (1) a snapshot comparison of relative abundance and population structure in 1990 and 2015 using survey data at 12 sites (hereafter snapshot comparison); (2) an estimate of the population growth rate (λ) from 1997–2014 based on low survey intensity monitoring data at six sites (hereafter historical population trend); (3) an estimation of the change in abundance between 2016 and 2017 based on high intensity monitoring data at eight sites (hereafter recent population change); and (4) a population reconstruction to estimate the minimum number of *G. insculpta* alive each year from 1990 to 2017 (hereafter population reconstruction). We also quantified broad-scale habitat changes relevant to *G. insculpta* that occurred in the study region over the past two to three decades, including forest size class and cover, developed land, and wetland habitat cover (hereafter habitat change).

MATERIALS AND METHODS

Study site.—We located population survey sites for *G. insculpta* along a 70 km stretch of river and tributaries in a section of northeastern Minnesota, USA (specific locations withheld in compliance with state of Minnesota data practices law). Mean human density is 1.7 people/km² (ArcGIS. 2012. USA Population Density. Available from <https://www.arcgis.com/home/item.html?id=302d4e6025ef41fa8d3525b7fc31963a> [Accessed 15 June 2017]). The elevation of the survey sites ranges from approximately 450–520 m. Mean temperature in May (the primary month in which annual monitoring is conducted) is 6.4° C, and annual monthly average precipitation in May is 12.7 cm (National Centers for Environmental Information. 2017. Climate at a Glance: U.S. Time Series, Precipitation. U.S. National Oceanic and Atmospheric Administration. Available from <http://www.ncdc.noaa.gov/cag/> [Accessed 19 June 2017]). The study area includes one of two *G. insculpta* populations in northeastern Minnesota (Moriarty and Hall 2014). It is currently unknown if there is connectivity between these populations (Gaea Crozier, pers. comm.). Mean distance between the two populations is approximately 35 km.

The river and tributaries are located within the Laurentian Mixed Forest ecological province (Minnesota Department of Natural Resources. 1999. Ecological classification system. Available from <http://www.dnr.state.mn.us/ecs/index.html> [Accessed 12 December 2016]). More than 90% of the surrounding land within the watershed is forested, with the remainder in non-forest and aquatic habitat classes. About 75% of the area is in public ownership. Mesic forest types,

which comprise 80% of the area, are dominated by aspen (*Populus* spp.), Balsam Fir (*Abies balsamea*), and Paper Birch (*Betula papyrifera*). Although pine forest types (*Pinus* spp.) are less common in the surrounding landscape, they are present in sandy soils adjacent to some nest sites at river cutbanks. Black Spruce (*Picea mariana*), Balsam Fir, Northern White Cedar (*Thuja occidentalis*), and Tamarack (*Larix laricina*) comprise over 90% of hydric forest types in the surrounding area. Non-forest vegetation consists of lowland alder (*Alnus* spp.) and grass/forb openings. Oxbow lakes and other non-flowing water features also occur in the study area (Brown et al. 2016).

Snapshot comparison.—In 1990, the *G. insculpta* population was surveyed at 12 sites in May and early June by two of the authors (Mark Nelson and Richard Buech). The sites were selected based on representative riparian habitat for the region, a range of stream sizes occupied by *G. insculpta* (including seven sites on the main river and five sites on smaller tributaries), and accessibility (i.e., public access or supportive private land owners; Buech et al. 1990). We replicated population surveys at the 12 sites in May of 2015. In 1990, seven of the sites were surveyed once, and the remaining five sites were surveyed twice. In 2015, we completed a single survey at each of the 12 sites. For the snapshot comparison data set, we compared the 1990 survey replicate that most closely matched the air temperature during the 2015 survey, to control for the influence of air temperature on availability to be detected (Brown et al. 2017).

One of the original researchers (Mark Nelson) delineated surveyed boundaries from 1990 using capture records, aerial imagery with marked capture locations, and field notes from the original surveys to standardize search area between years. This allowed us to standardize area surveyed and search protocol. In both years, observers completed a single-observer visual encounter survey of each site. Surveyors walked approximately 15 m apart from one another traversing all potential *G. insculpta* habitat within the site. The river distance of each site varied from 0.63–3.37 km (mean = 1.47 km). The survey area at each site included the shallow edges of the river up to approximately 100 m inland, with both sides of the river surveyed. We could not survey the interior of the river due to the natural turbidity of the water.

We ensured the same area was surveyed in both survey periods by loading survey boundaries into handheld GPS units for 2015 surveys. Two to four observers surveyed each side of the river and we recorded total survey time (survey time included time spent processing turtles). We did not attempt to standardize search rate. Observers in both survey periods were novices with ≤ 1 y previous *G. insculpta* field experience. We attempted

to survey sites under similar air temperature and time of year as the original surveys. In both years, each turtle found was measured and individually marked using carapace notches (Cagle 1939). Data recorded in both years included sex, plastron annuli count, and location (in 1990, pin point capture locations on an aerial photograph flown in July, 1981; 1:15,800 scale; in 2015, Etrex 30x GPS units, Garmin, Kansas City, Kansas, USA). For this study, we classified individuals as juveniles when straightline carapace length (SCL) was ≤ 170 mm (Harding and Bloomer 1979).

To assess changes in population size and structure between 1990 and 2015, we calculated relative abundance (i.e., number of captures), relative abundance standardized by survey effort (i.e., number of captures/person-hour; hereafter standardized relative abundance), adult sex ratio, and adult:juvenile ratio at each site for each survey year. We used paired randomization tests with 10,000 iterations to determine if these population metrics differed between years. When sample sizes are small such as in our study ($n = 12$ sites), randomization tests are an appropriate alternative to *t*-tests because the statistical distribution is derived from the randomized data, rather than assuming the data follow an underlying parametric distribution (Sokal and Rohlf 1995). Specifically, we paired relative abundance, standardized relative abundance, adult sex ratio, and adult:juvenile ratio from 1990 and 2015 from each site. In each iteration, the 1990 and 2015 values for each site were randomized, and the overall difference between years was computed. We tested the null hypothesis that there was no change in population size or structure between 1990 and 2015. The *P*-values represent the proportion of trials resulting in a mean difference between sampling years as great or greater than the one obtained in our study (Sokal and Rohlf 1995). Thus, a small *P*-value indicates our observed 1990 and 2015 data were unusually different compared to the randomized data. We considered differences to be significant at $P \leq 0.05$. For each metric tested, we only included sites with data in both 1990 and 2015, which ranged from eight sites (adult sex-ratio) to 12 sites (relative abundance).

Historical population trend.—From 1997–2014, we surveyed for *G. insculpta* at nesting sites and pre-nesting staging areas within six of the sites surveyed in 1990. Similar to the snapshot comparison protocol, we completed a single-observer visual encounter survey of each site. The same lead observers completed these surveys annually. Surveys occurred from 28 April to 17 July (primarily from late May to early June). We performed a total of 179 surveys across 18 survey years. Sites varied in size but surveyed area was consistent across years. We did not record search effort. We removed 12 surveys from 2013–2014 that we conducted

with the assistance of a dog. We recorded sex, age, mark number, and location of detected individuals, and marked new individuals using carapace notches. Over the 18-year period, the number of sites surveyed per year ranged from 1–6 (mean = 3.1). During years when sites were surveyed, number of survey replications per site ranged from 1–11 (mean = 3.4).

To estimate λ from 1997–2014, we used an open population N -mixture model (Dail and Madsen 2011). N -mixture models use both spatial and temporal replication of count data to jointly estimate abundance and detection probability, and thus they account for observed numbers being a product of both ecological and observational processes (Royle 2004b; Kéry and Royle 2016). We used a binomial distribution for the observation (i.e., detection) process, and a Poisson distribution for the state (i.e., abundance) process. We specified the population as open between sampling years, and closed between replications within sampling years. Because our interest was solely in estimating λ , we used the simplest population dynamics structure:

$$N_{i,t} = N_{i,t-1} \times \lambda$$

Where the estimated abundance (N) at time t is based on N at time $t-1$ and λ . This model does not separately estimate apparent survival and recruitment. We did not include environmental covariates for state variables (i.e., initial abundance or λ). During model development, we tested the influence of two variables on detection probability. First, we created a model where detection probability was a function of day of year, to account for the potential influence of year-to-year variation in survey days on the probability of documenting *G. insculpta*. Second, we created a model where detection probability was a function of observer experience (i.e., number of years of survey experience), based on previous research that found observer experience was a strong predictor of *G. insculpta* detection probability (Jones et al. 2015). We performed a model selection analysis with the two candidate detection probability models as well as a null model (i.e., no covariates) using Akaike's Information Criterion corrected for small sample size (AICc). The model selection indicated that neither of the detection probability models performed better than the null model (AICc weight = 1), and thus we used the null model for the final analysis.

We estimated λ using the six survey sites and 18-y observation period. We did not estimate λ separately for each site because of the large numbers of missing observation years at the site-level. We assessed model goodness-of-fit using a 1,000-replication parametric bootstrap of the Pearson chi-square statistic (Kéry and Royle 2016), which indicated the model fit was adequate ($c\text{-hat} < 1$, $P = 0.840$). For this analysis, we used the

software packages unmarked (version 0.11-0; Fiske and Chandler 2011) and AICcmodavg (version 2.0-4) in program R, version 3.2.4 (R Core Team 2016). We specified the open population model using the function `pcountOpen` with `dynamics = trend`.

Recent population change.—We completed 112 population surveys across eight sites in 2016 and 2017, following the survey protocol of Brown et al. (2017). This included eight and six survey replicates at each site in 2016 and 2017, respectively. All surveys occurred between 30 April and 5 June. Each site consisted of a 380–560 m stretch of river ($\bar{X} = 486$ m) and adjacent riparian and upland habitat. We completed visual encounter surveys by foot and each site was surveyed by walking four transects spaced 15 m apart on each side of the river. Surveyors used a GPS unit with pre-loaded transects to ensure the same area was surveyed each time. Transects were surveyed by a single-observer in 2016, while in 2017 two-observers surveyed each transect independently (i.e., a double-observer survey approach). The survey team also differed in 2016 and 2017. For each *G. insculpta* detected, we recorded its mark (if previously captured and marked), location, and time of detection, and for new within-year captures, we obtained standard measurements and photographs, sex, and marked the individual with carapace notches. We also recorded the length of the survey, the temperature at the beginning and at the end of the survey, and the surveyors present.

To estimate abundance in 2016 and 2017, we used multinomial N -mixture models that included a removal sampling observation process (Royle 2004a). In a previous study, we found this model framework performed well using the 2016 survey data and using model simulations (Brown et al. 2017). For this analysis, we treated the two transect passes during each survey in 2017 as a single pass by combining the unique individuals captured. For the latent abundance distribution, we included a categorical covariate that represented the eight sites, thus allowing the abundance intercept to differ among sites, and we included mean air temperature during the survey as a covariate for detection probability (Brown et al. 2017). For both years, the goodness-of-fit assessment indicated some overdispersion ($c\text{-hat} = 1.52$ and 1.42 in 2016 and 2017, respectively). We accounted for this overdispersion in our site-level abundance estimates by inflating the 95% confidence intervals based on the $c\text{-hat}$ values (Kéry and Royle 2016).

Population reconstruction.—We developed an estimate of the minimum *G. insculpta* population from 1990 to 2017 using all capture records from research and monitoring activity in the study area (Buech et

al. 1997; Brown et al. 2016, 2017). We used annuli count data to estimate ages of individual turtles ≤ 20 y old (Ernst and Lovich 2009; Parren 2013). Adult *G. insculpta* with > 15 plastron annual growth rings in addition to $> 50\%$ plastron wear (smooth growth rings) were assumed to be 20+ y old (a highly conservative estimate, with individuals known to reach at least 55 y old in this watershed [Brown et al. 2015]). If no age was recorded, we estimated the individual to be 10 y old (also a conservative estimate, as mean age of all captures was 21 y). No hatchlings (≤ 1 y) were included in the analysis due to their low survivorship (Ernst and Lovich 2009; Jones 2009).

We included three calculations that estimated minimum number of turtles alive each year, each with different assumptions and biases. First, we summed the total number of unique individuals captured each year (hereafter Caught). This estimate has no assumptions about age of individuals or post-capture survival, but is heavily biased by survey effort, which was very high in 1990–1993 and 2015–2017 and comparatively low in the intervening years. Second, we added turtles that were alive, but not caught, during a survey year (hereafter Alive). To accomplish this, we used the capture history and estimated age at capture for each individual encountered over the 28-y monitoring period. This estimate has no assumptions about post-capture survival, but it assumes the estimated age of individuals during the first detection year is accurate in order to estimate their life status in the years prior to first detection. Third, we added turtles that were assumed to be alive in years following their last detection (hereafter Assumed Alive). For this model, we assumed turtles live 5 y after their last recapture event, which represented the mean time between capture events for individuals in our data set. The 5 y additional survival estimate is likely conservative, as average adult survival for *G. insculpta* is typically very high (e.g., 0.97–1.0 in Maine [Compton 1999]). We assumed immigration and emigration were negligible (Buech 1997). We provided graphical output displaying the results of the three calculations. We also determined the geometric mean of λ from 1990 to 2016 using the combined population reconstruction calculations.

Habitat change.—To determine if habitat changes relevant to *G. insculpta* have occurred in the study region over the past two to three decades, we quantified changes in forest size class and cover, developed land, and wetland habitat cover within the study area watershed (about 2,000 km²). We estimated changes in forest size class using U.S. Forest Service Forest Inventory and Analysis (FIA) 5-y and 20-y forest stand age data representing the period 1977–2013 (Miles et al. 2016). We calculated mean forest stand age by

weighting each age class by the proportion that class occupied out of the total area of the watershed. We estimated changes in forest, wetland, habitat cover, and developed land using the land cover data of the Coastal Change Analysis Program (C-CAP) of the National Oceanic Atmospheric and Administration from 1996 and 2010 (Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, and Office for Coastal Management. 2013. NOAA's Coastal Change Analysis Program [C-CAP] 1996 to 2010 Regional Land Cover Change Data - Coastal United States. Available from <https://coast.noaa.gov/dataviewer/#/imagery/search/where:ID=3688> [Accessed 3 February 2017]).

RESULTS

Snapshot comparison.—We captured 44 and 50 *G. insculpta* during population surveys in 1990 and 2015, respectively (Table 1). Relative abundance ($P = 0.585$) did not differ between years. Standardized relative abundance decreased ($P = 0.119$) from 0.50 captures/h (SD = 0.457) in 1990 to 0.32 captures/h (SD = 0.264) in 2015. The adult sex ratio ($P = 0.222$) and adult:juvenile ratio ($P > 0.999$) did not differ between years. In 1990, the overall adult ratio was 1.1 adults:1 juvenile, whereas in 2015 it was 1.4 adults:1 juvenile. In 1990, the adult sex ratio was 1.8 female:1 male, whereas in 2015 it was 1.4 female:1 male. The mean SCL for all turtles was 187.5 mm (± 61.9 SD) in 1990 and 188.6 mm (± 35.6 SD) in 2015. The minimum SCL was 76 mm and 66 mm, and the maximum SCL was 233 mm and 229 mm, in 1990 and 2015, respectively. We encountered 13 recently deceased (estimated < 1 y since death) and 20 *G. insculpta* shell remains in 2015. Only two deceased *G. insculpta* were noted in 1990.

Historical population trend.—We recorded 462 captures of 346 unique individuals at the six monitoring sites from 1997–2014 (excluding captures from surveys removed prior to analyses). The percentage of adult females captured per year ranged from 60–94%, with an overall sex ratio of 7.7 females:1 male from 1997–2014. The percentage of adults captured per year ranged from 57 to 100%, with an overall adult:juvenile sex ratio of 8.3 adults:1 juvenile. Estimated detection probability was 0.023. Estimated total abundance fluctuated between 627 and 804 individuals, but confidence intervals were wide (Fig. 2). Estimated λ over the 18-y period was 1.016.

Recent population change.—We recorded 313 captures of 174 unique individuals at the eight monitoring sites in 2016. Estimated abundance was 247, and ranged from 5–77 among sites. We made 136 captures of 98

TABLE 1. Summary of *Glyptemys insculpta* (Wood Turtle) captures, captures/hour, percentage adult (adult/total captures), and percentage female (females/total adult captures) at 12 population survey sites in northeastern Minnesota, USA, in 1990 and 2015. For the first summary statistic, we only included sites with data in both 1990 and 2015, which ranged from eight sites (adult sex-ratio) to 12 sites (relative abundance). For the second summary statistic, we included global values or global mean value for all terms. The abbreviation NA = not available.

Site	Captures		Captures/Hour		% Adult		% Female	
	1990	2015	1990	2015	1990	2015	1990	2015
Site 1	0	1	0.00	0.17	NA	0	NA	0
Site 2	2	2	0.33	0.17	0	100	NA	100
Site 3	5	8	0.63	0.31	100	75	40	83
Site 4	10	4	0.50	0.12	100	75	50	100
Site 5	2	9	0.33	0.82	100	67	50	67
Site 6	6	9	1.50	0.82	83	89	40	63
Site 7	3	2	0.43	0.11	67	50	50	0
Site 8	4	5	0.44	0.42	75	100	33	60
Site 9	3	4	0.75	0.23	100	75	33	33
Site 10	9	4	1.13	0.44	100	100	100	100
Site 11	0	0	0.00	0.00	NA	NA	NA	NA
Site 12	0	2	0.00	0.27	NA	50	NA	100
Mean (SD) - sites with replicate captures	3.7 (± 3.3)	4.2 (± 3.1)	0.5 (± 0.5)	0.3 (± 0.3)	81 (± 33)	81 (± 17)	50 (± 22)	63 (± 34)
Total / Mean - all data	44	50	0.56	0.33	89	72	56	72

unique individuals at the eight monitoring sites in 2017. Estimated abundance was 112 (a 54% decrease from 2016), and ranged from 4–26 among sites. Estimated abundance in 2017 was slightly higher than in 2016 at one site, slightly lower at four sites, and substantially lower at three sites (i.e., 58–81% decrease; Fig. 3). We encountered 20 dead *G. insculpta* during all fieldwork in 2016, including 18 during abundance surveys. We encountered 30 dead *G. insculpta* during all fieldwork in 2017, including 28 during abundance surveys. We

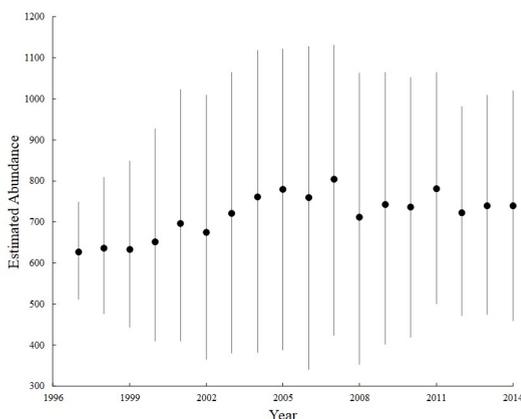


FIGURE 2. Total estimated abundance of *Glyptemys insculpta* (Wood Turtle) from 1997–2014 at six population monitoring sites in northeastern Minnesota, USA, based on visual encounter surveys at nesting sites and pre-nesting staging areas. Black circles represent total estimated annual abundance and gray bars show the 95% confidence intervals. Estimated mean annual population growth rate (λ) was 1.016.

had found 13 of these individuals alive in 2016. If we assume that in 2016 all 30 individuals were alive and total abundance was 247, then a maximum estimate of annual survival was 0.87.

Population reconstruction.—We made 3,753 captures of 1,064 unique individuals from 1990–2017. Mean number of capture events/turtle was 1.7 (SD = 1.1), with 38% of all turtles recaptured at least once. Mean number of years between recapture events was 5.7 (SD = 7.5). The mean estimated age of all turtles captured was 21.3 y (SD = 9.2). We found an overall sex ratio of 2.4 female:1 male and an overall adult:juvenile ratio of 4.6 adult:1 juvenile. Estimated mean population size from 1990 to 2017 was 668 individuals (SD = 61, range = 498–783), and mean λ was 0.992. The overall population reconstruction indicated a potential population decline since 2006 (Fig. 4). Mean λ was 1.007 and 0.970 from 1990–2005 and 2006–2016, respectively.

Habitat change.—The average age of forest stands in the watershed increased slightly from 1977–2013 (slope of trendline = 3.6 and 3.8 for 5-y and 20-y age class stand calculations, respectively). The mean age of forest stands (based on 5-y age classes) was 40.8 y and 55.4 y in 1977 and 2009–2013, respectively. Between 1996 and 2010, there was a 4.8% net increase in developed land, but even in 2010 only 0.6% of the watershed was developed. Overall, 14.1% of land in

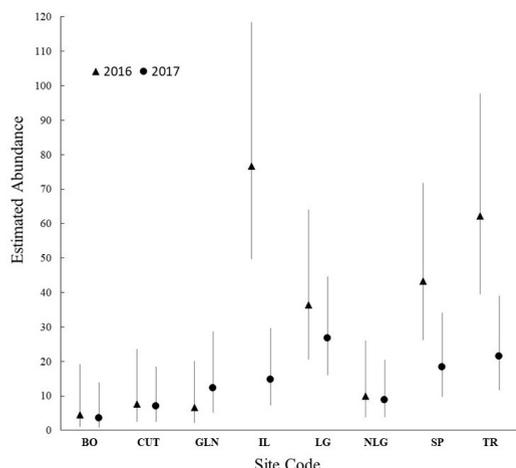


FIGURE 3. Estimated abundance of *Glyptemys insculpta* (Wood Turtle) in 2016 and 2017 at eight population monitoring sites in northeastern Minnesota, USA, based on visual encounter surveys using a robust survey design. Black triangles and circles represent estimated abundance in 2016 and 2017, respectively, and gray bars show the 95% confidence intervals.

the watershed changed cover classes between 1996 and 2010. Net forest cover decreased by 1.7%, and net wetland cover decreased by 0.2%.

DISCUSSION

The four population assessments we made did not indicate an increasing *G. insculpta* population. We found equivocal results about whether the population was stable or declining, and therefore the current status and trajectory of this *G. insculpta* population in

northeastern Minnesota is unclear. The snapshot and historical population trend analyses found no evidence of a population decline from 1990 to 2015. However, we found lower captures/h in 2015 in comparison to 1990. The population reconstruction suggests a possible declining trend since 2006, and population surveys in 2016 and 2017 indicate a recent large decrease in abundance. Population surveys in future years are needed to determine if a substantial proportion of the 2016 population was alive but not detected in 2017. This could be caused either by dispersal out of the study area or by a substantial reduction in terrestrial activity.

The large number of dead *G. insculpta* encountered, and the annual increase in mortality detections, is cause for concern. *G. insculpta* exhibit delayed reproductive maturity (about 15 y), long life-spans, and low annual reproductive success, and thus we would expect that annual adult survivorship above 95% is required for long-term population persistence (Compton 1999). Many of the factors that have been associated with *G. insculpta* declines in other regions (e.g., human development, heavy recreational use, poaching; reviewed by Jones et al. 2015) do not appear to be an issue in northeastern Minnesota; however, unless the population is much larger than we have estimated, recent mortality rates are unusually high.

It is possible that most of the observed adult mortality in our study area was the result of predation. American Badgers (*Taxidea taxus*) were the largest contributor to *G. insculpta* nest predation (Cochrane et al. 2015, 2017). We did not directly observe adult *G. insculpta* predation by *Taxidea taxus* or other mesopredators, but

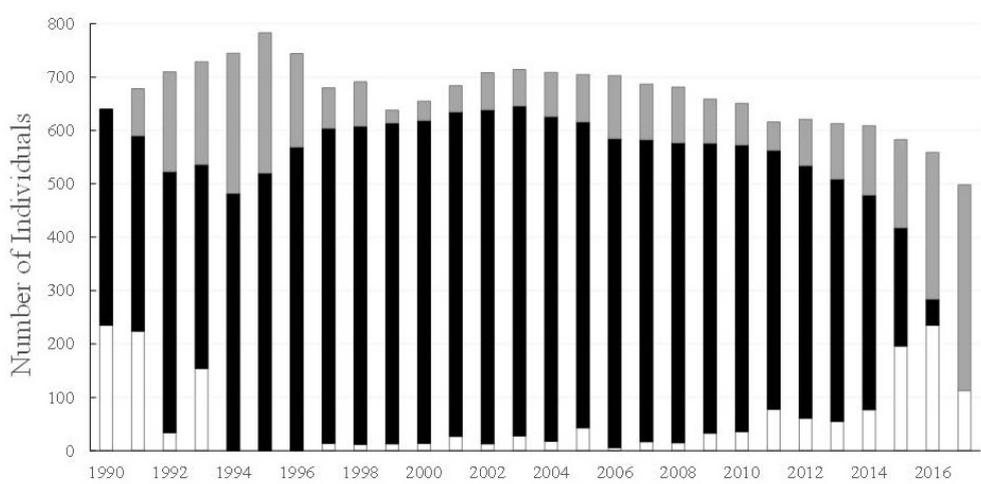


FIGURE 4. Estimated minimum number of *Glyptemys insculpta* (Wood Turtle) in a northeastern Minnesota, USA, population from 1990–2017 based on 3,753 detection events and 1,064 unique individuals. The Caught (open) bars represent unique individuals detected each year. Alive (black) bars include turtles that were detected in future years, but were not detected during a survey year. This estimate has no assumptions about post-detection survival, but it assumes the estimated age of individuals during the first detection year is accurate in order to estimate their life status in the years prior to first detection. For the Assumed Alive (gray) bars, we added turtles that were estimated to be alive after their final detection year. For this model, we assumed turtles live 5 y after their last recapture event, which represented the mean time between capture events for individuals in our data set.

many of the live individuals encountered had limb or shell deformities, and some of the dead individuals had either been predated or had their carcasses scavenged. Another potential threat to *G. insculpta* in the region is road mortality. In 2015 and 2016, we documented substantial use of roadsides as nesting sites by *G. insculpta*. However, we also found low vehicular mortality rates near key nesting areas (Cochrane et al. 2017). Determining causes and mitigating mortality of adults in this population will be an important component of long-term conservation, and additional research is needed.

Our area of inference is restricted to a 70-km stretch of habitat in one of two major river systems currently inhabited by *G. insculpta* in northeastern Minnesota. While our study area included representative riparian habitat and stream sizes occupied by *G. insculpta* in northeastern Minnesota, we cannot confidently extrapolate our results to other populations, as factors that influence population dynamics vary across the landscape. Some historic survey data exists for the other northeastern Minnesota river system (Buech et al. 1997), but low survey effort and few marked turtles restricted our ability to include these data in our status assessment.

We recognize that each of the data sets used for this population status assessment has limitations. Performing snapshot survey replications at 25-y intervals inherently limits inferential power due to a lack of long-term rigorous population monitoring. However, in the absence of long-term monitoring data, standardized snapshot comparison studies provide valuable quantitative assessments of the status of populations (Dodd et al. 2007; Brown et al. 2012; Foster et al. 2013). The sampling design for the historical population trend data limited the precision of our abundance estimates (i.e., low number of sites surveyed, low number of survey replications completed annually, and low detection probability). The survey methodology was also female-biased, and thus may not be fully representative of the overall population trend. The recent population change analysis used a more robust population survey design, but was limited to two years of survey data. While the population reconstruction incorporated a large amount of observation data, those data were not collected systematically over time or space, and the reconstruction required assumptions regarding age of individuals and persistence after detection. Further, Alive estimates from the 1990s were predictably larger than estimates from 2010s, as they benefitted from decades of recapture events. Using FIA data to make inferences about changes in forest stand age across a relatively small geographic area is also limited in its inferential power to low sample sizes. Despite the inherent data limitations, our status assessment provides useful

information regarding long-term population dynamics of *G. insculpta* in northeastern Minnesota, and serves as a baseline for comparing future survey results.

Moving forward, a sustained conservation commitment to ensure long-term persistence of *G. insculpta* populations in the Upper Midwest is needed, given the potential for rapid population declines (e.g., Garber and Burger 1995; Daigle and Jutras 2005; this study). We encourage continued investment in development of a collaborative, long-term monitoring program that is based on a robust survey design (Brown et al. 2017). Abundance monitoring programs increase the ability to determine causal relationships and implement appropriate response strategies when needed, and allow managers to link current conservation actions (e.g., nest protection, roadside barriers) to population trends, thus informing future management decisions.

Acknowledgments.—This study was funded by the Minnesota Department of Natural Resources through a Competitive State Wildlife Grant (#F14AP00028), with additional financial support provided to Madaline Cochrane through a University of Minnesota Integrated Biosciences Graduate Program summer fellowship. We thank Bruce Brecke and Lynelle Hanson for field assistance in 1990. We thank Katelin Goebel, Stefan Nelson, Lucas O’Neil, Nathan Ose and Ashlee Ricci for field assistance in 2015. Kelcy Huston, Cord Reno, Alek Holleran, Josh Kircher and Maria Berkeland assisted with surveys in 2016 and 2017. We thank local landholders who allowed us access to conduct research on their lands. Maya Hamady was instrumental in obtaining funding for *G. insculpta* research in northeastern Minnesota. Sampling and handling methods were approved by the University of Minnesota Institutional Animal Care and Use Committee (Protocol No. 1504-32514A), and permitted by the Minnesota Department of Natural Resources. Any use of trade, product, or firm names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

LITERATURE CITED

- Buech, R.R., L.G. Hanson, and M.D. Nelson. 1997. Identification of Wood Turtle nesting areas for protection and management. Pp. 383–391 *In* Proceedings: Conservation, Restoration, and Management of Tortoises and Turtles - an International Conference. Van Abbema, J. (Ed.). New York Turtle and Tortoise Society, New York, New York, USA.
- Brown, D.J., M.M. Cochrane, and R.A. Moen. 2017. Survey and analysis design for Wood Turtle population modeling. *Journal of Wildlife Management* 81:868–877.

- Brown, D.J., M.D. Nelson, D.J. Rugg, R.R. Buech, and D.M. Donner. 2016. Spatial and temporal habitat-use patterns of Wood Turtles at the western edge of their distribution. *Journal of Herpetology* 50:347–356.
- Brown, D.J., M. Schrage, D. Ryan, R.A. Moen, M.D. Nelson, and R.R. Buech. 2015. *Glyptemys insculpta* (Wood Turtle). Longevity in the wild. *Herpetological Review* 46:243–244.
- Brown, D.J., A.D. Schultz, J.R. Dixon, B.E. Dickerson, and M.R.J. Forstner. 2012. Decline of Red-Eared Sliders (*Trachemys scripta elegans*) and Texas Spiny Softshells (*Apalone spinifer emoryi*) in the Lower Rio Grande Valley of Texas. *Chelonian Conservation and Biology* 11:138–143.
- Cagle, F.R. 1939. A system of marking turtles for future identification. *Copeia* 1939:170–173.
- Cochrane, M.M., D.J. Brown, and R.A. Moen. 2017. Wood turtle nest monitoring and mortality surveys. Technical Report No. NRRI/TR-2017-06 Release 1.0. Natural Resource Research Institute, Duluth, Minnesota, USA. 40 p.
- Cochrane, M.M., R.A. Moen, and D.J. Brown. 2015. *Glyptemys insculpta* (Wood Turtle). Nest predation. *Herpetological Review* 46:618.
- Compton, B.W. 1999. Ecology and conservation of the Wood Turtle (*Clemmys insculpta*) in Maine. M.S. Thesis, University of Maine, Orono, Maine. 91 p.
- Cushman, S.A. 2006. Effects of habitat loss and fragmentation on amphibians: a review and prospectus. *Biological Conservation* 128:231–240.
- Daigle, C., and J. Jutras. 2005. Quantitative evidence of decline in a southern Québec Wood Turtle (*Glyptemys insculpta*) population. *Journal of Herpetology* 39:130–132.
- Dail, D., and L. Madsen. 2011. Models for estimating abundance from repeated counts of an open metapopulation. *Biometrics* 67:577–587.
- Dodd, C.K., W.J. Barichivich, S.A. Johnson, and J.S. Staiger. 2007. Changes in a northwestern Florida Gulf Coast herpetofaunal community over a 28-y period. *American Midland Naturalist* 158:29–48.
- Ernst, C.H., and J.E. Lovich. 2009. *Turtles of the United States and Canada*, 2nd Edition. Smithsonian Institution Press, Washington, D.C., USA.
- Fiske, I.J., and R.B. Chandler. 2011. Unmarked: an R package for fitting hierarchical models of wildlife occurrence and abundance. *Journal of Statistical Software* 43:1–23.
- Foster, R.L., A.M. McMillan, and K.J. Roblee. 2013. Population status of Hellbender Salamanders (*Cryptobranchus alleganiensis*) in the Allegheny River Drainage of New York State. *Journal of Herpetology* 43:579–588.
- Garber, S.D., and J. Burger. 1995. A 20-y study documenting the relationship between turtle decline and human recreation. *Ecological Applications* 5:1151–1162.
- Harding, J.H., and T.J. Bloomer. 1979. The Wood Turtle, *Clemmys insculpta*, a natural history. *Bulletin of the New York Herpetological Society* 15:10–26.
- Henle, K., K.F. Davies, M. Kleyer, C. Margules, and J. Settele. 2004. Predictors of species sensitivity to fragmentation. *Biodiversity and Conservation* 13:207–251.
- Jones, M.T. 2009. Spatial ecology, population structure, and conservation of the Wood Turtle, *Glyptemys insculpta*, in central New England. Ph.D. dissertation. University of Massachusetts, Amherst, Massachusetts, USA. 243 p.
- Jones, M.T., L.L. Willey, T.S.B. Akre, and P.R. Sievert. 2015. Status and Conservation of the Wood Turtle in the Northeastern United States. Report submitted to the Northeast Association of Fish and Wildlife Agencies for Regional Conservation Needs Grant 2011-02. Cabot, Vermont, USA. 271 p.
- Kéry, M., and J.A. Royle. 2016. *Applied Hierarchical Modeling in Ecology: Analysis of Distribution, Abundance and Species Richness in R and BUGS*. Academic Press, Boston, Massachusetts, USA.
- Lenhart, C.F., J.R. Naber, and J. L. Nieber. 2013. Impacts of hydrologic change on sandbar nesting availability for riverine turtles in Eastern Minnesota, USA. *Water* 5:1243–1261.
- Levell, P. 2000. Commercial exploitation of Blanding’s Turtle, *Emydoidea blandingii*, and the Wood Turtle, *Clemmys insculpta*, for the live animal trade. *Chelonian Conservation and Biology* 3:665–674.
- Miles, P.D., C.L. VanderSchaaf, C. Barnett, B.J. Butler, S.J. Crocker, D. Gormanson, C.M. Kurtz, T.W. Lister, W.H. McWilliams, R.S. Morin, et al. 2016. *Minnesota Forests 2013*. Resource Bulletin NRS-104. U.S. Department of Agriculture, Forest Service, Northern Research Station, Newtown Square, Pennsylvania, USA. 134 p.
- Moriarty, J.J., and C.D. Hall. 2014. *Amphibians and Reptiles in Minnesota*. University of Minnesota Press, Minneapolis, Minnesota, USA.
- Parren, S.G. 2013. A twenty-five year study of the Wood Turtle (*Glyptemys insculpta*) in Vermont: movements, behavior, injuries, and death. *Herpetological Conservation and Biology* 8:176–190.
- R Core Team. 2016. *R: A language and environment for statistical computing*. R Foundation for Statistical Computing, Vienna, Austria. <http://www.R-project.org>.
- Royle, J.A. 2004a. Generalized estimators of avian abundance from count survey data. *Animal Biodiversity and Conservation* 27:375–386.

Cochrane et al.—Status of a *Glyptemys insculpta* population in Minnesota.

- Royle, J.A. 2004b. N-mixture models for estimating population size from spatially replicated counts. *Biometrics* 60:108–115.
- Sokal, R.R., and F.J. Rohlf. 1995. *Biometry: The Principles and Practice of Statistics in Biological Research*. 3rd Edition. W.H. Freeman and Company, New York, New York, USA.
- Willoughby, J.R., M. Sundaram, T.L. Lewis, and B.J. Swanson. 2013. Population decline in a long-lived species: The Wood Turtle in Michigan. *Herpetologica* 69:186–198.
- Wisconsin Department of Natural Resources. 2016. Wisconsin Wood Turtle (*Glyptemys insculpta*) Status Assessment and Conservation Strategy. PUB NH-935 2016. Wisconsin Department of Natural Resources, Madison, Wisconsin, USA.
- Yackulic, C.B., E.W. Sanderson, and M. Uriarte. 2011. Anthropogenic and environmental drivers of modern range loss in large mammals. *Proceedings of the National Academy of Sciences* 108:4024–4029.



MADALINE M. COCHRANE completed her M.S. thesis in August 2017, where she studied the spatial and population ecology of *Glyptemys insculpta* at the University of Minnesota, Duluth, USA. She received her B.A. in Biology and Environmental Studies from Macalester College, St. Paul, Minnesota, USA. Her research interests include herpetological conservation, spatial ecology, and resource selection. (Photographed by Kelcy Huston).



DONALD J. BROWN is a Research Assistant Professor in a joint position with West Virginia University, Morgantown, USA, and the U.S. Forest Service, Northern Research Station, St. Paul, Minnesota. He received his B.Sc. in Fisheries & Wildlife from the University of Minnesota-Twin Cities, St. Paul, USA and his M.Sc. and Ph.D. from Texas State University, San Marcos, USA, in Wildlife Ecology and Aquatic Resources. Donald's lab conducts research that contributes to conservation of wildlife populations and communities under changing environmental and management conditions, with a focus on amphibians, reptiles, and songbirds. (Photographed by Andrew Badje).



MARK D. NELSON, Ph.D., is a Research Forester with the U.S. Forest Service, Northern Research Station, St. Paul, Minnesota. Mark conducts research to support the Forest Inventory and Analysis (FIA) program, which is responsible for the inventory of United States forest land across all ownership categories. Mark develops and assesses geospatial approaches for making FIA estimates more spatially explicit and relevant to a wider range of natural resources. His current research involves the use of forest inventory data, geospatial data, and satellite imagery for assessing and mapping forest land cover, land use, canopy disturbance, and wildlife habitat. (Photographed by James D. Garner).



RON A. MOEN is an Associate Professor in the Department of Biology and a Senior Research Associate with the Natural Resources Research Institute at the University of Minnesota Duluth, USA. He received his B.Sc. in Ecology from Cornell University, Ithaca, New York, USA, and his M.Sc. and Ph.D. from the University of Minnesota, St. Paul, USA, in Wildlife Conservation. Ron's lab generally conducts research on habitat use and population dynamics of mammals, with occasional work on other taxonomic groups. (Photographed by Brett Groehler).