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Demography and Population Status of the Flattened Musk Turtle, *Sternotherus depressus*, in the Black Warrior River Basin of Alabama

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ABSTRACT. – The demography of the flattened musk turtle, *Sternotherus depressus*, was compared among sites differing in degree of human impact and among this and previous demographic studies. Sites designated as impacted due to increased sedimentation, algae, and turbidity had lower rates of turtle capture, adult-biased size distributions, and decreased parasitism by the leech *Placobdella parasitica*. Sex ratio, shell damage, and relative body mass did not differ significantly among sites. Comparison of demographic studies since 1981 revealed a consistent decline in rate of turtle capture at five of the six sites investigated, a consistent decline in ratio of males to females, and a decline in frequency of large individuals. These results indicate that population declines have continued, even at sites considered relatively pristine, as a result of insufficient recruitment and/or illegal collection. We recommend the continued protection and monitoring of *S. depressus* populations and water quality within the Black Warrior River Basin of Alabama.

KEY WORDS. – Reptilia; Testudines; Kinosternidae; *Sternotherus depressus*; turtle, population status, demography, sex ratio, habitat, parasitism, conservation; Alabama; USA

The flattened musk turtle, *Sternotherus depressus* (Tinkle and Webb, 1955), is a relatively small turtle that occurs above the Fall Line in the Black Warrior River Basin in Alabama (Figs. 1 and 2). Individuals are rarely found outside their aquatic habitat and quickly seek refuge under rock crevices, logs, and debris when disturbed. Typical *S. depressus* habitat is permanent oligotrophic streams with abundant rock crevices for shelter (Mount, 1981). In addition to *S. depressus*, at least one species of salamander, three fish, one mollusk, and six caddisflies are endemic to this habitat and region. The Fall Line functions as a dispersal barrier for many species, creating a unique biogeographic unit upstream in the Black Warrior River Basin (Tinkle, 1959). Much of this area is underlain by the Warrior Coal Field (Tolson, 1985) and is impacted by active as well as abandoned and "unreclaimed" surface mines. In addition to impacts from mining, portions of the Black Warrior Basin are affected by municipal sewage discharge, agricultural runoff, and erosion from certain silvicultural practices (Harkins, 1980), all of which alter water chemistry and increase sedimentation (Knight and Newton, 1977; Dyer, 1982). These factors make this an area of conservation concern because of potential impacts these activities can exert on this unique fauna.

Various biologists expressed concerns in the mid-1960s that *S. depressus* populations were declining. A federal notice of review was published in 1977 by the U.S. Fish and Wildlife Service. This prompted a series of demographic studies by Mount (1981), Ernst et al. (1983), Drummond Coal Company (Anonymous, undated), Dodd et al. (1986), and Dodd (1988a), most of which indicated that declines in *S. depressus* population density and distribution were associated with habitat degradation. As a result of these studies, *S. depressus* was designated a threatened species (U.S. Fish and Wildlife Service, 1987). Subsequent studies revealed a propensity for low recruitment, skewed sex ratios and age structure, and diseased individuals at some sites (Dodd et al., 1988; Dodd, 1988a, 1989a; Ernst et al., 1989; Mount et al., 1991).

Mount (1981) suggested that the absence of certain food items necessary for juvenile *S. depressus* may be the limiting resource. In this paper we focus on population parameters which may be associated with that hypothesis; a companion paper evaluates diet and prey availability of *S. depressus*. Because the evidence indicates that some populations of *S. depressus* have declined, the objectives of this study are to (1) compare demography of *S. depressus* at sites impacted by human activity with those considered to be relatively unimpacted, and (2) compare our findings with those of previous studies at the same or nearby sites.

MATERIALS AND METHODS

Six sites were chosen, three of which were considered unimpacted, three classified as impacted (Fig. 2). The three unimpacted sites were: (1) Sipsey Fork (Winston County, T9S R8W S33 and 34), a third-order stream with a 306 km^2 drainage area that originates in the Sipsey Wilderness and flows through the Bankhead National Forest, and that has the highest known density of *S. depressus* (Dodd et al., 1988; Ernst et al., 1989); (2) Brushy Creek (Winston County, T9S R7W S23), a second-order stream with a 110 km^2 drainage area within the Bankhead National Forest, that has a "small but stable" population of *S. depressus* (Dodd et al., 1988); (3) Capsey Creek (Winston County, T9S R6W S18), a first-order stream with a 26 km^2 drainage area in the Bankhead National Forest. The three impacted sites were: (4) Blackwater Creek (Walker County, T13S R7W S15 and 22), a third-order stream with a 281 km^2 drainage area, lying

Figure 1. Adult *Sternotherus depressus* from Bankhead National Forest, Winston Co., Alabama. Photo by James C. Godwin.

outside the Bankhead National Forest and downstream from agricultural activities; (5) Clear Creek (Winston County, T12S R8W S3, 10 and 11), a third-order stream with a 183 km^2 drainage area north of Lewis Smith Lake, lying within the Bankhead National Forest but downstream from mining and agricultural activities, where *S. depressus* was described by Mount (1981) as "moderately common" but food-limited, and subsequent trapping efforts by Dodd et al. (1988) were unsuccessful; and (6) Lost Creek (Walker County, T13S R9W S28 and 29), a first-order stream draining 84 km^2, lying outside the Bankhead National Forest and immediately adjacent to an active mine, with a population skewed toward larger turtles (Dodd et al., 1986).

These six sites were chosen because they differed substantially in impacts associated with human activities. Forests predominate at the three unimpacted sites (Sipsey Fork, Brushy Creek, and Capsey Creek) that occur within the Bankhead National Forest, whereas croplands, pastures, mines, and/or residential areas have displaced much forest within the headwaters of the three impacted sites (Blackwater Creek, Clear Creek, and Lost Creek). Inferior water quality parameters (increased stream turbidity, silt accumulation, and algae) were found in association with the three sites considered to be impacted as compared to the three unimpacted sites. Water turbidity was highly variable and significantly greater at impacted sites ($H = 11.3$, $p < 0.01$). Substrate types differed significantly between unimpacted and impacted sites ($\chi^2=33.8$, $p = 0.002$). The difference resulted from an increased frequency of silt and algae at the impacted sites.

Trapping of *S. depressus* was conducted from 15 April to 30 August during both 1994 and 1995. Traps were placed near rocks or logs between 1800 to 2000 hrs each evening and removed between 0600 to 0800 hrs each morning. The number of hours a baited trap was set (trap-hours) was usually 12 hrs per day. Trap-hours were later converted to trap-days by dividing the total trap-hours by 12 for comparison among sites and with previous studies. Trapping effort was greatest at the Sipsey Fork site to maximize the volume of diet data at that optimal site, and lowest at the Capsey Creek site because trapping began there in 1995. Turtles were caught by hand or in funnel traps baited with sardines or chicken (Iverson, 1979). Each individual was weighed, maximum carapace length (CL in mm) measured and, if unmarked, marginal scutes notched for identification as described by Dodd et al. (1986). Sex was determined by tail length and vent location (Ernst and Barbour, 1989). Captured turtles were kept for 48 hrs at most and released at the place of capture.

Three variables — population density, sex ratio, and size distribution — were used to compare populations between impacted and unimpacted sites as well as among this study and previous ones (Mount, 1981, 1996; Ernst et al., 1983, 1989; Dodd et al., 1986, 1988; Dodd, 1988a, 1988b; Mount et al., 1991). Population density was estimated by the Jolly-Seber method (Begon, 1979) and Program CAPTURE (White et al., 1982). Turtle captures per trap-day for 1994 and 1995 were calculated for each site. Size distribution classes were based on CL and classified as < 40, 40–49.9, 50–59.9, 60–69.9, 70–79.9, 80–89.9, 90–99.9, and ≥ 100 mm. Turtles ≥ 70 mm CL were considered adults based on Close's (1982)

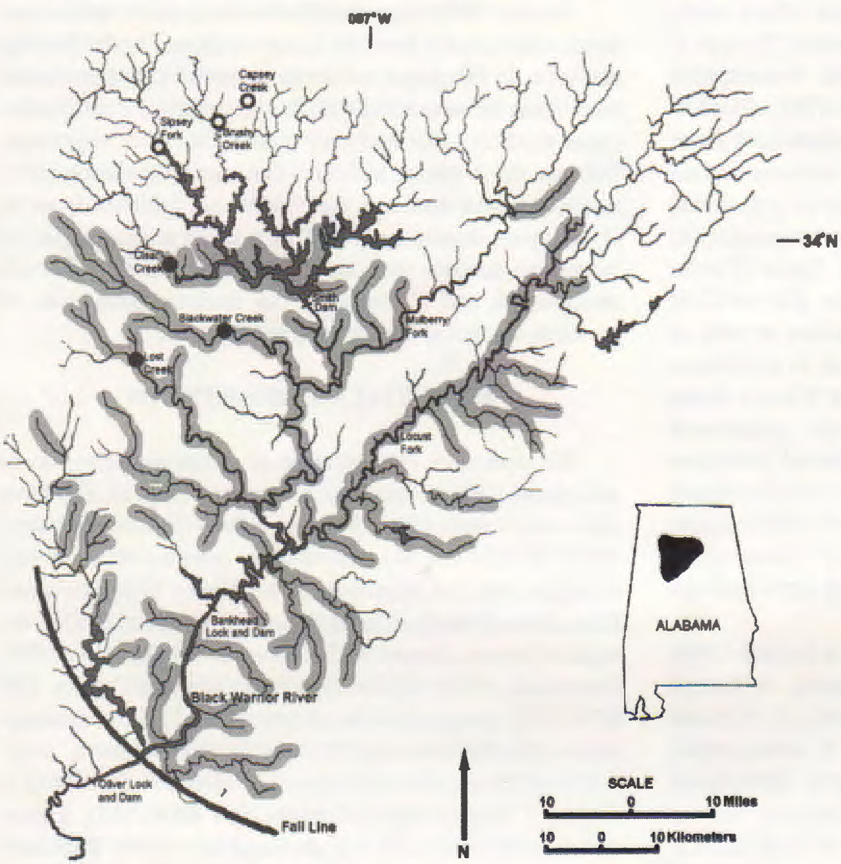

Figure 2. Map of the upper Black Warrior River Basin in Alabama showing surface waters impacted by mining through 1980 (shaded areas), the Fall Line (solid line), and location of study sites (impacted sites – solid dots; unimpacted sites – open circles). Modified from Harris, 1987.

study of *S. depressus* reproductive cycles that revealed that sexual maturity occurs at approximately 70 mm CL for both sexes. Smaller individuals (< 70 mm CL) were classified as juveniles.

Three variables—relative body mass, leech parasitism, and shell damage — were used to assess the physical condition of individuals at each site. Relative body mass was determined by comparing individual body mass to CL. A log-log plot of these variables was used to compare regression lines between unimpacted and impacted sites. To assess parasitism, we recorded the presence or absence of the leech, *Placobdella parasitica*, a common ectoparasite of freshwater turtles (Dodd, 1988b; Ernst and Barbour, 1989). Shell condition was assessed by recording if individuals had missing or eroded marginal scutes or shell cavities.

Significant differences in population estimates between this study and that of Dodd et al. (1988) were determined by non-overlapping confidence limits. To examine patterns of trapping success and relative body mass, we compared regression lines with an analysis of covariance (PROC GLM, SAS Institute, Inc., 1990). Data (CL, size ratio, sex ratio, leech parasitism, and shell damage) were evaluated with the CATMOD procedure in SAS (SAS Institute, Inc., 1990). A constant of one was added to all cells when zero counts were encountered. The level of significance for all analyses was set at $\alpha = 0.05$ and $\alpha = 0.10$ was used to define trends.

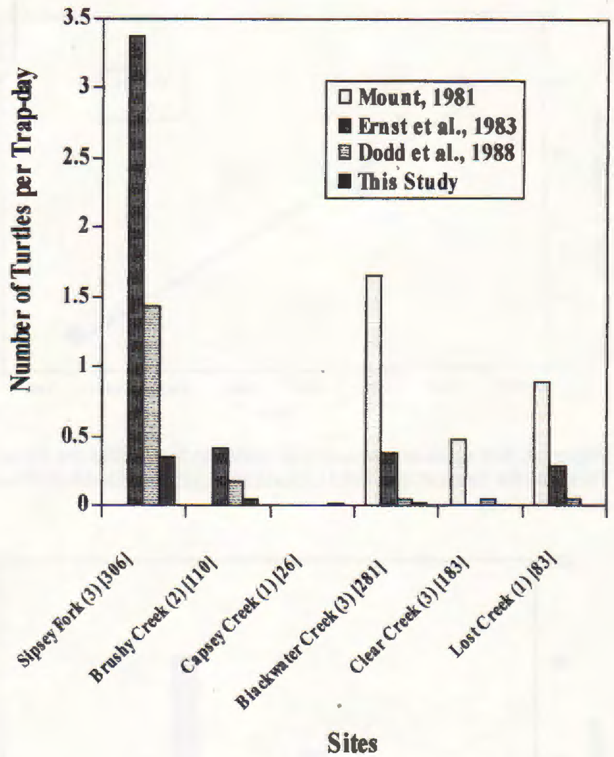

Figure 3. Histogram comparing number of *Sternotherus depressus* caught per trap-day from 1981 to 1995. Historical data are from Mount (1981), Ernst et al. (1983), and Dodd et al. (1988) for the same or nearby sites. Numbers in parentheses and brackets are stream order and km² of drainage area, respectively.

RESULTS

Two hundred and eighty-one *S. depressus* were captured during this study. Sixty-two individuals were recaptured, resulting in a total of 343 captures during 3035 trap-days (Table 1). Trapping effort ranged from 819 trap-days at the Sipsey Fork site to 180 trap-days at the Capsey Creek site. The collecting effort that was made during the summer of 1994 at Sipsey Fork provided the only adequate sample size for estimation of population size. Because hatchlings were caught and two known deaths occurred during this study, populations were considered to be open. Using the Jolly-Seber population model (Begon, 1979), population size was calculated to be 459 (95% C.L. = 132–786) for early July 1994 at this site, a figure that was not significantly different from that calculated for early July 1985 (644; 95% C.L. = 442–824) by Dodd et al. (1988). The collecting activity by Dodd et al. (1988) and Dodd (1988) was centered approximately 1 km upstream from our study site. Therefore, the upstream limits of our study site likely overlap the downstream limits of the earlier studies. Unlike the previous year, Dodd (1988a) considered the Sipsey Fork population to be closed in 1986 because neither mortality nor hatchlings were observed. Using Program CAPTURE (model M_t; White et al., 1982), Dodd (1988a) reported a population size of 306 (95% C.L. = 239–373). A test for closure failed ($p = 0.027$)

Table 1. Capture data for *Sternotherus depressus* during the summers of 1994 and 1995 at six sites in the Black Warrior River Basin of Alabama. New = number of new turtles captured each year. Recaptures = number of recaptured turtles previously caught in 1983 by Ernst et al. (1983), in 1985 by Dodd et al. (1988), or in 1994 during this study.

Location	New 1994	Recaptures From 1983	Recaptures From 1985	New 1995	Recaptures From 1994	Recaptures From 1983	Recaptures From 1985	Total No. Captures	Total Trap-Days	Turtles per Trap-Day
Unimpacted										
Sipsey Fork	115	26	1	67	5	11	1	274	819	0.3346
Brushy Creek	18	0	1	18	2	0	2	47	551	0.0853
Capsey Creek	0	0	0	3	0	0	0	3	180	0.0167
Subtotal	133	26	2	88	7	11	3	324	1550	0.2090
Impacted										
Blackwater Creek	2	0	1	11	0	0	1	15	535	0.0280
Clear Creek	2	0	0	1	0	0	0	3	530	0.0057
Lost Creek	1	0	0	0	0	0	0	1	420	0.0024
Subtotal	5	0	1	12	0	0	1	19	1485	0.0128
Overall Total	138	26	3	100	7	11	4	343	3035	0.1130

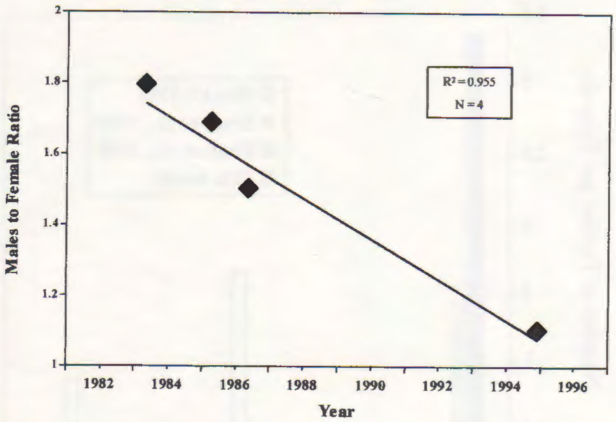

Figure 4. Sex ratio as frequency of males to females at the Sipsey Fork site for Ernst et al. (1983), Dodd et al. (1988), Dodd (1988a), and this study.

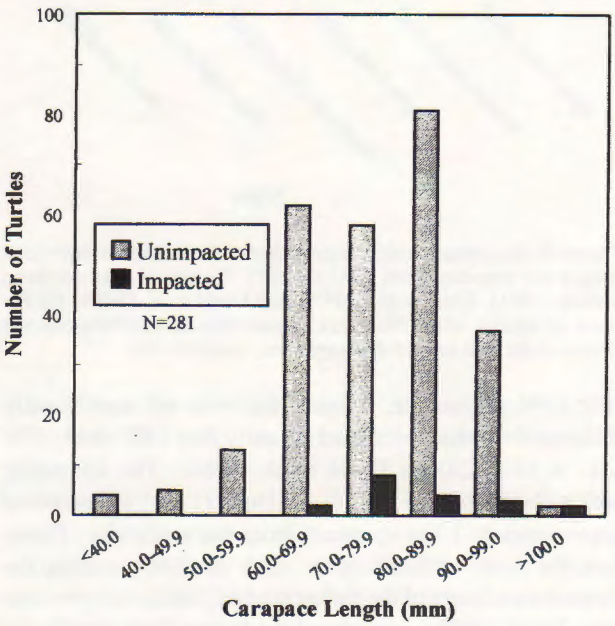

Figure 5. Size-class frequencies of *Sternotherus depressus* collected at unimpacted and impacted sites during the summers of 1994 and 1995 in northern Alabama.

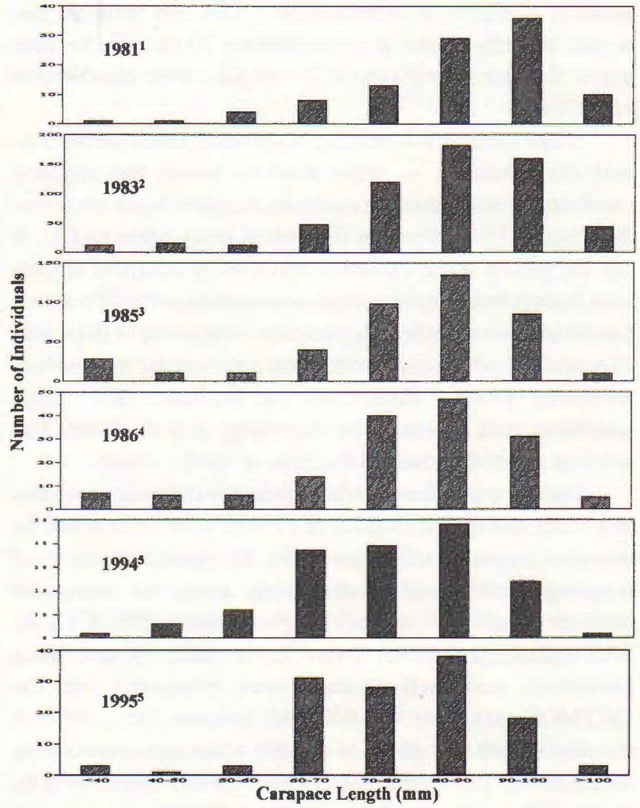

Figure 6. Comparison of carapace length distributions of *Sternotherus depressus* collected between 1981 and 1995, based on the studies of [1]Mount (1981) for 1981, [2]Ernst et al. (1983) for 1983, [3]Dodd et al. (1988) for 1985, [4]Dodd (1988a) for 1986, and [5]this study for 1994 and 1995.

for a similar estimation of population size for 1994. However, this failure should be interpreted cautiously because Program CAPTURE cannot differentiate behavior responses and/or time trends in capture probabilities from violations of assumptions associated with closure. Therefore, for comparative purposes, we used Program CAPTURE and found that model M_h, which assumes variation in individual trapping probability, was the most appropriate model for our data. Population size was estimated to be 347 (95% C.L. = 295–399) for 1994, a figure that did not differ from that reported by Dodd (1988a).

Turtle captures per trap-day ranged from 0.3346 at Sipsey Fork to 0.0024 at Lost Creek and differed significantly between impacted and unimpacted sites (Table 1; $F = 5.16$; $p = 0.044$). A significant reduction in rate of turtle capture over time (Mount, 1981; Ernst et al., 1983; Dodd et al., 1988) was observed at the Sipsey Fork, Brushy Creek, Blackwater Creek, Clear Creek, and Lost Creek sites (Fig. 3, $F = 6.58, p = 0.026$). The rate of decline in turtle captures per trap-day was not significantly different between impacted and unimpacted sites ($F = 1.22, p > 0.25$). However, a significant difference in sex ratios was found at Sipsey Fork between this study and that of Dodd (1989a) ($p < 0.005$) and an apparent decrease in the relative number of males has occurred there since 1983, with the ratio now approaching 1:1 (Fig. 4).

Carapace length ranged from 31.9 to 106.3 mm. The distribution of CL differed significantly between unimpacted and impacted sites (Fig. 5; $\chi^2 = 11.88$; $p = 0.018$). The difference resulted from populations skewed toward larger

Table 2. Frequency of recaptured males (M) and females (F) at Sipsey Fork in each size class ≥ 70 mm CL for Dodd et al. (1988), Dodd (1988a), and this study. Only individuals with an interval of at least one year between capture and recapture were included in this contingency table.

	Carapace Length (mm)					
	70–79		80–89		90+	
	M	F	M	F	M	F
1985[a]	16	4	10	9	23	5
1986[b]	22	4	28	14	23	15
1994	4	1	11	6	6	5
1995	2	0	8	3	3	4
Total	44	9	57	32	55	29

[a]Data from Dodd et al. (1988)
[b]Data from Dodd (1988a)

individuals at the impacted sites. The CL distribution for this study differed significantly from all previous studies ($\chi^2 = 115.00$; $p < 0.0001$) as a result of a shift toward smaller individuals over time (Fig. 6). The frequency with which each sex was recaptured during this study did not differ significantly from that of Dodd et al. (1988) or Dodd (1988a) (Table 2; $\chi^2 = 5.46$, $p = 0.792$ and $\chi^2 = 0.92$, $p = 0.821$, respectively). However, males were recaptured more frequently than females ($\chi^2 = 5.67$, $p = 0.0173$).

Relative body mass did not differ significantly between unimpacted and impacted sites (Fig. 7, $F = 1.34$, $p > 0.25$). Thirty percent of the individuals caught during May in 1994 and 1995 (Table 3) were parasitized by the leech, *Placobdella parasitica*. Overall, frequency of leech parasitism for both years increased to more than 75% as the summer progressed. Frequency of leech parasitism did not differ among males, females, and juveniles (Table 3; $\chi^2 = 0.94$, $p = 0.63$). There were significantly more leeches on individuals from the unimpacted sites (Table 3; $\chi^2 = 17.6$, $p < 0.0001$). Overall, leech parasitism did not differ between this study and that of Dodd (1988b) (77.0 vs. 73.1%, respectively; $\chi^2 = 2.19$, $p = 0.14$). Ninety-four individuals had shell damage in the form of eroded or missing marginal scutes or pitting of the shell surface. Frequency of shell damage was not different between unimpacted and impacted sites (Table 3; $\chi^2 = 0.013$, $p = 0.91$) nor was frequency of shell damage from erosion of carapace marginals different from that of Dodd et al. (1988a) (33.5 vs. 36.1%, respectively; $\chi^2 = 0.576$, $p = 0.448$).

DISCUSSION

Impacted vs. Unimpacted Sites

The overall objective of this study was to provide a basis for eventual testing of Mount's hypothesis of food limitation as a major factor for *S. depressus* populations by examining demographic trends and physical condition of turtles at impacted and unimpacted sites. We predicted that population density would be highest at unimpacted sites in comparison to impacted sites. Such a comparison, however, was impossible at our sites due to inadequate sample sizes. However, current densities can be roughly estimated by examination of turtle capture rate. For this variable, turtle captures per trap-day were significantly less at the impacted sites than the unimpacted sites. This comparison should be interpreted cautiously because differential trapping success

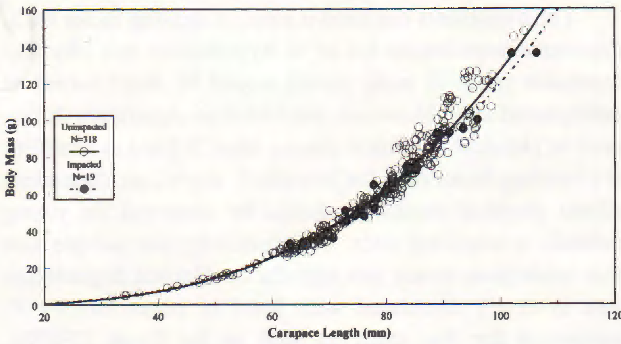

Figure 7. Relative body mass of *Sternotherus depressus* at three unimpacted (solid line) vs. three impacted (dashed line) sites in northern Alabama during the summers of 1994 and 1995.

could result from differences in habitat structure and turtle behavior as well as turtle abundance (Dodd et al., 1988).

In addition to decreased trapping success, impacted sites have unaltered sex ratios, altered size distributions, and a trend toward fewer juveniles. The overall sex ratio of 1:1 observed at both impacted and unimpacted sites should facilitate genetic variability (Fisher, 1930; Verner, 1965). However, unbalanced sex ratios have been reported for *S. depressus* and other congeners (Risley, 1933; Cagle, 1942; Gibbons, 1970; Bury, 1979; Dodd et al., 1988; Dodd 1988a, 1989a, 1989b; Ernst et al., 1989). Biased trapping techniques, sexual differences in movement patterns, environmental sex determination (ESD), and/or seasonal changes have been suggested as possible causes for observed sex ratio biases (Dodd, 1989a; Gibbons, 1990). The results from our study indicate no sex ratio bias associated with stream water quality. The frequency of large adults (≥ 100 mm CL) relative to all other size classes was significantly higher at impacted sites. We predicted that juvenile frequency would be highest at unimpacted sites. However, frequency of juveniles relative to all other size classes was not significantly different between unimpacted and impacted sites, although they tended to be higher at unimpacted sites, a trend consistent with this prediction ($p = 0.105$). Only 19 individuals were captured at the impacted sites and an adequate sample size may not be possible to detect a significant difference due to the rarity of *S. depressus* at these sites.

From the above demographic patterns we conclude that (1) adult males and females are affected equally by human impacts, and (2) carapace size distribution is associated with human impacts.

Table 3. Occurrence of shell damage and leech parasitism of *Sternotherus depressus* at impacted vs. unimpacted sites during the summers of 1994 and 1995 in northern Alabama. J = juveniles, M = males, and F = females.

	Damaged Turtles*		Turtles with Leeches**				Percent with Leeches**				
	n	%	J	M	F	Total	May	June	July	August	Total
Unimpacted	88	33.6	73	96	91	260	30.4	80.5	77.4	86.7	80.2
Impacted	6	31.6	1	3	0	4	—	10.0	40.0	50.0	21.1
Overall	94	33.5	74	99	91	264	30.4	71.6	74.7	85.5	77.0

* Includes only new captures, not recaptures.
** Includes all captures and recaptures.

The hypothesis that food is a major limiting factor for *S. depressus* populations led us to hypothesize that physical condition (relative body mass) would be more robust at unimpacted sites. However, we found no significant difference in physical condition among sites. If food availability is a limiting factor only for juvenile *S. depressus*, then a less robust physical condition should be observed for young animals at impacted sites. Unfortunately, our sample size was inadequate to test this hypothesis. Habitat degradation was inversely correlated with level of parasitism by *P. parasitica* for this study as well as for Dodd (1988b). Hilsenhoff's (1988) tolerance classification ranks macroinvertebrates from zero to five, with zero being the least tolerant. *Placobdella parasitica* is considered relatively intolerant (tolerance classification = 2) to organic wastes. Whether the decreased frequency of parasitism by *P. parasitica* at the impacted sites is due to habitat degradation or the scarcity of their host, *S. depressus*, could not be determined during this study. Whereas individuals at unimpacted sites did not appear to gain a benefit in more robust physical condition or reduced shell damage, they did incur a potential greater cost in higher leech parasitism. Individuals at impacted sites showed no evidence of being at a disadvantage in terms of physical condition, shell damage, or leech parasitism, as compared to those at unimpacted sites.

Long-Term Trends

Although we found no statistically significant differences in population estimates from those published by Dodd et al. (1988) and Dodd (1988a), a decline in rate of turtle capture was observed. These results appear to be contradictory because the rate at which turtles enter traps would be expected to be related to turtle density. We also found a consistent decline in the frequency of males over time, a trend similar to that of turtle capture rate, and a shift in carapace length distributions toward smaller individuals over time. Three factors that might produce these results are sampling bias, turtle behavior, and turtle abundance. Differing trapping techniques such as trap placement, trap size, and/or bait used could result in sampling bias. However, this factor is unlikely to result in a consistent decline in trapping success, especially of males, over time. Declining trends could be the result of trap avoidance by males over time. Closed population estimators allow for altered trappability; however, this population was considered open because mortality and reproduction were observed. Therefore, no estimator is available if the assumption of equal trappability is violated. If male turtles learn to avoid traps, then fewer males than females and fewer old (large) than young (small) individuals would be recaptured. This was not the case. More males than females were recaptured. Thus, males appeared to be "trap-happy" rather than "trap-shy." Additionally, all size classes (70 to 100+ mm) were recaptured with equal frequency.

If sampling bias or trap response by turtles cannot be attributed to these trends, then the consistent decline in trapping success may reflect a general decline in population size that was not confirmed by population estimators due to violations of assumptions. If such a decline in density is occurring, then sex ratio and body size trends indicate that mortality, emigration, or illegal collecting may be higher for males than females, and higher for adults than for juveniles. Additionally, these trends appear to have occurred consistently over the past 12 years. Dodd et al. (1988) found a high frequency of diseased individuals (23% on 11 September 1985), and 20 dead turtles at Sipsey Fork in 1985. Disease symptoms and associated observations were emaciation, shell lesions, shell discoloration, eroded shell marginals, necrotic tissue, swollen or closed eyes, pale faces, lack of leech parasitism, and altered basking behavior (Dodd, 1988a). The decreased male:female ratio that we observed might be due to sex-specific mortality from the disease found in 1985, although Dodd (1988a) found no significant difference between the frequency of diseased males and females. Disease may have impacted juveniles less strongly due to decreased exposure to disease via social interactions associated with sexual maturity. The smallest diseased individual found by Dodd (1988a) was a male with a CL of 75.0 mm. However, if sex- or age-specific mortality occurred during 1985, a sharp decline or "spike" rather than a consistent decline in frequency of males and adults since 1981 should occur. In 1986, Dodd (1988a) found fewer diseased individuals (12.4%) and less severe disease symptoms. We found no disease symptoms during this study but some individuals had healed shell cavities from previous lesions. Therefore, because no sex- or age-specific effect from disease was observed, the decline in trapping success and frequency of males, and increased frequency of juveniles, is more likely the result of increased male emigration, decreased male survivorship, and/or the removal of males by illegal collecting.

The Sipsey Fork site has the highest known densities of *S. depressus*, suggesting that conditions are more favorable for this species at that site than elsewhere. Additionally, patterns of sex ratios indicate that mating opportunities for males would be less advantageous elsewhere. Therefore, we found no evidence that males would benefit by emigrating either up- or downstream from the Sipsey Fork site.

Recreational activities (e.g., canoeing, swimming, and fishing) have increased over time due to improved access to the Bankhead National Forest. Such access could affect local non-commercial pet collecting and illegal pet trade. This factor could impact males more than females because males move over longer distances and therefore are encountered more frequently, and based on recapture frequencies, may be more readily trapped (Dodd et al., 1988).

CONCLUSIONS

Trapping success and size distributions indicate that there are fewer turtles, especially juveniles, at impacted sites in comparison to unimpacted sites. With low or zero recruitment at impacted sites, *S. depressus* will eventually be extirpated from these areas. The body condition of individuals remaining in impacted areas was not affected. However, reproduction needs to be examined. Population dynamics over the past 16 years are complex but they appear to indicate declines, especially of adult males, even at the relatively unimpacted sites. Therefore, *S. depressus* populations should continue to be protected from illegal collection and monitored for changes in demography and incidence of disease. The Black Warrior River Basin is inhabited by many unique species and special conservation efforts are necessary to provide adequate habitat for their continued existence.

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