

Sample manuscript submission for reference regarding the formatting guidelines for the family of natural history journals co-published by Eagle Hill Publishing:

Caribbean Naturalist

Northeastern Naturalist

Southeastern Naturalist

Urban Naturalist

Note: this sample was compiled by taking parts of several different manuscripts in order to illustrate a wider range of the significant formatting concerns than any single manuscript might contain.

A Range-wide Survey of the Endangered Black-capped Vireo in Texas

Tiffany M. McFarland^{1,*}, Heather A. Mathewson¹, Julie E. Groce², Michael L. Morrison³, and R. Neal Wilkins¹

¹Institute of Renewable Natural Resources, Texas A&M University, College Station, TX 77843. ²Institute of Renewable Natural Resources, Texas A&M University, San Antonio, TX 78215. ³Department of Wildlife and Fisheries Sciences, Texas A&M University, College Station, TX 77843. *Corresponding author - tiffany.mcfarland@agnet.tamu.edu.

Abstract - *Vireo atricapilla* (Black-capped Vireo) is an endangered migratory songbird with a breeding range that exists predominantly within Texas. Despite the species' listing under the Endangered Species Act in 1987, Black-capped Vireos were largely undocumented in much of the range. We sampled over 10,700 points in Texas, resulting in 2458 Black-capped Vireo detections. We examined the relationship between Black-capped Vireo occurrence and vegetation and broad-scale landscape variables, and we assessed if detections were clustered. Black-capped Vireo detections occurred often on a common soil type, but were found where slopes were higher in the western part of the range. We found evidence of clustering in six of our eight study areas but no evidence of habitat metrics driving that clustering. These data improve the current knowledge of Black-capped Vireo distribution and offer opportunities for improved guidance for conservation and management efforts.

Introduction

Coastal embayments and their tributaries provide important habitat for numerous economically and ecologically important fishes and invertebrates (Nixon 1982, Sogard and Able 1991, Szedlmayer and Able 1996, Yanez-Arancibia et al. 1994). Unlike river-dominated estuarine systems, coastal bays are generally well mixed with less dramatic environmental gradients (Kjerfve 1986, Kjerfve and Magill 1989, Mariani 2001). The physical and hydrodynamic features of coastal bays, as well as their interaction with tides and predominant patterns in wind speed and direction can therefore influence the structure of the associated faunal communities (Mariani 2001, Murphy and Secor 2006). Fishes that use coastal bay habitats can be classified into numerous groups that describe their level of residency in the system (Whitfield 1999, Yanez-Arancibia et al. 1994). Resident species remain in the coastal

bay for their entire life cycle. Transient species use coastal bays as opportunistic foraging grounds, spawning areas, juvenile nurseries, and migration corridors (Day et al. 1989, Moyle and Cech 1982). Some transient species are facultative in their use of coastal lagoons, while others commonly described as estuary-dependent use coastal bays as primary nursery habitats. The high productivity, abundant prey resources, suitable physicochemical conditions, and shallow nature of coastal bays provide fishes with favorable conditions for reproduction, growth, and refuge from predation (Nixon 1982, Yanez-Arancibia et al. 1994).

The coastal bays of Delaware, Maryland, and Virginia on the Delmarva Peninsula support numerous species of resident, estuary-dependent, and facultative transient fishes (Derickson and Price 1973; Love et al. 2009; Murphy and Secor 2006; Pacheco and Grant 1965; Richards and Castagna 1970; Schwartz 1961, 1964; Weston 1993). As in most temperate estuaries, the composition and abundance of the fish fauna of those in the families Sciaenidae and Clupeidae, utilize these bays as seasonal nursery grounds (Able and Fahay 2010, Wang and Kernehan 1979), occurring in greatest abundance during summer and early autumn (Cowan and Birdsong 1985, Derickson and Price 1973, Pacheco and Grant 1965, Weston 1993). The nekton assemblages in the Delmarva coastal bays have been found to differ spatially among individual embayments (Murphy and Secor 2006) with distributions within embayments influenced by variability of environmental factors such as salinity (Love et al. 2009).

The Delaware Coastal Bays are the most anthropogenically impacted on the Delmarva Peninsula (Chaillou et al. 1996, DIBEP 1995, Maxted et al. 1997, Price 1998, Valdes-Murtha 1997). Since 1950, the population of Sussex County, DE, the county bordering the bays, has increased 221% (Delaware Population Consortium 2010, US Census Bureau 1995). Shoreline alteration and nutrient enrichment have contributed to the disappearance of *Zostera marina* L. (Eelgrass) and *Ruppia maritima* L. (Widgeongrass) and an increase of dense drift macroalgal communities dominated by *Agardhiella tenera* J. Ag. (Red Weed) and *Ulva lactuca* L. (Sea Lettuce) (DIBEP 1995; Maxted et al. 1997; Price 1998; Timmons and Price 1996a, b; Tyler 2010; Weston 1993). Such changes to both physical and chemical environments often adversely affect the functional role of specific habitats for the faunal assemblages by altering food webs and species composition (Deegan 2002, Holland et al. 2004). These changes can result in a homogenization of species assemblages to those only tolerant of variable conditions (Maxted et al. 1997) as well as a loss of productive habitat for transient fauna that rely on such habitat as nursery grounds (Holland et al. 2004). A review of previous studies examining the shore-zone fish community in the Delaware Coastal Bays concluded that a shift in dominance

had occurred since the 1950s, with a greater abundance of species in the Family Cyprinodontidae, which are more tolerant of low dissolved oxygen (DO), and lower abundances of estuary-dependent species (Chaillou et al. 1996, Price 1998). This shift was attributed to primarily attributed to recent nutrient enrichment in these bays.

Although several studies have examined the fish and *Callinectes sapidus* Rathbun (Blue Crab) assemblage structure in the shore zone of the Delaware Coastal Bays (Clark 2002, Derickson and Price 1973, Pacheco and Grant 1965, Weston 1993), most did not link trends in assemblage structure to environmental variability, and all relied on measurements of relative abundance rather than quantitative measurements of density. This limits the usefulness of the data to identify habitat areas and/or conditions that promote productive, biodiverse faunal communities. Additionally, the most spatially and temporally comprehensive studies were conducted several decades ago, while more recent studies consist mostly of limited temporal sampling at specific locations. The need for more extensive current data is highlighted by the fact that there has been a continued shift in the shore fish assemblage towards hypoxia-tolerant resident species since those early studies were conducted (Price 1998).

The objective of the present study was to use quantitative sampling to examine spatial and temporal dynamics of the shore zone fish and the Blue Crab assemblage in tidal creeks of the Delaware Coastal Bays and attribute those dynamics to habitat-specific variations in temperature, salinity, and DO. Specifically, we compared differences in densities of fishes and Blue Crabs among sections of tidal creeks within different embayments across the spring/summer nursery period and winter of two years. This comparison provided insight into spatial and seasonal differences in the density of individuals and assemblage structure and the underlying physicochemical characteristics that drive those changes.

Field-Site Description

Located in southeastern Delaware, the Delaware Coastal Bays consist of three interconnected embayments—Rehoboth Bay, Indian River Bay, and Little Assawoman Bay—and their associated tributaries and canals (Fig. 1). These shallow (average depth is 1.2 m), polyhaline to mesohaline systems have a combined water surface area of 83 km² and are sheltered from substantial direct interaction with the Atlantic Ocean (DIBEP 1995, Martin et al. 1996, NOAA 1990). The northernmost bays are contiguous with the Atlantic Ocean at Indian River Inlet, while Little Assawoman Bay is connected with the ocean through Ocean City Inlet,

MD. All bays are fringed by salt or brackish, tidally influenced marshes (Tiner 2001, Weston 1993) and have little freshwater inflow (Martin et al. 1996, Weston 1993). Indian River and Rehoboth Bays have more extensive drift macroalgal communities than does Little Assawoman Bay (Price 1998; Timmons and Price 1996; Tyler 2005, 2010; Valdes-Murtha 1997). Mean water temperatures ranged from a low of 5 to 7 °C in January to a high of 20 to 23 °C in August (NOAA 2010).

Sampling was conducted in White Creek (a tributary of Indian River Bay), Miller Creek (a tributary of Little Assawoman Bay), and Assawoman Canal (a man-made waterway connecting the two tributaries) (Fig. 1). Two stations were sampled in both tidal creeks, and four stations were sampled in Assawoman Canal (Fig. 1). Both White Creek stations were adjacent to substantial shoreline development and contained dense drift macroalgae. In contrast, the Miller Creek stations were bordered by conservation lands (Assawoman Wildlife Area) with salt marsh and sediment bank shorelines and generally contained no macroalgae. Habitat characteristics of Assawoman Canal varied between northern and southern stations. Northern Assawoman Canal stations (A1 and A2) were bordered by steeply sloping, high-bank shorelines and contained scattered deposits of large woody debris. In contrast, the southern stations (A3 and A4) were bordered by forested wetlands and fringing salt marsh with wood/leaf-litter detritus. All stations in Assawoman Canal were bordered by moderate upland development, and two of the stations (A2 and A4) were adjacent to bridges that traversed the canal.

Methods

Pleurobema strodeanum was studied in Eightmile Creek, Walton County, FL. Mussels were collected in 2004 by handpicking using qualitative visual and tactile searches that covered 250 m downstream from a bridge crossing. Individuals were measured using digital calipers to the nearest 0.01 mm along three shell axes (length, width, and height). Mussels were tagged by removing a small portion of the periostracum on the shell and attaching a numbered Floy® shellfish tag (Floy Tag, Seattle, WA) using cyanoacrylate glue (Sickel et al. 1997). Voucher specimens were preserved in 70% ethanol and are curated in the Troy University collection. Seven years later, in 2011, twenty-eight *P. strodeanum*, of the 161 originally tagged, were recaptured and measured. Growth rate percentages (G%) were calculated as follows:

$$G\% = 100 * (M_f - M_i) / M_i,$$

where M_f and M_i are the final and initial measurements (length, width, height, and volume) in 2011 and 2004, respectively (Negishi and Kayaba 2009). Morphometric ratios were determined as follows, width to length (W:L), height to length (H:L), and height to width (H:W). Volume was calculated using the formula:

$$V = (\text{Length} \times \text{Width} \times \text{Height}) / C,$$

where the constant (C) was determined with the method used by Martins et al. (2011). Briefly, 18 *P. strodeanum* shells between 31.6 mm and 60.4 mm in length were sealed with Parafilm®. Mussels were immersed in water, and the volume was determined by displacement to the nearest 0.1 ml in a graduated cylinder. These numbers were plotted against the volume obtained by multiplying the length, width, and height. The regression line was calculated and the slope (C) was 2.51 ($R^2 = 0.975$, $P < 0.001$). The von Bertalanffy growth equation was used to determine the age of the individuals. The equation is expressed as:

$$L_t = L_\infty (1 - e^{-K(t-t_0)})$$

It is usually applied when age and body size are known; however, it can be inverted to find age as follows:

$$t = \ln[(L_\infty - L_t) / (L_\infty - L_0)] / -K,$$

where L_∞ (asymptotic length) is the theoretical maximum length at infinite age, L_t is the length of the organism at time t (age), L_0 is the length of the organism at time 0 since age is not known, and K is Brody's growth constant. Using mark and recapture growth data, L_∞ and K can be estimated using a linear regression of the Ford-Walford relationship (Ricker 1975):

$$L_\infty = [a / (1 - \beta)]$$

$$K = -\ln \beta,$$

where a is the y intercept and β is the slope (Anthony et al. 2001, Haag 2009). Assuming constant growth, the Brody's growth constant (K) was divided by seven to be used in the inverted von Bertalanffy growth equation since the estimated K was based on a seven-year period.

Data were plotted using Excel 2007 for Windows, and linear regressions between growth rate (%) and all the measurements taken were performed using SPSS[®] version 11.0. Data were tested for normality, and a *t*-test was applied to measure differences between smaller and larger individuals' growth-rate percentages, and differences between morphometric ratios in 2004 and 2011. If data were not normal, a Mann Whitney (U) or Wilcoxon (Z) test was used. A *t*-test was used for width, width:length, and height:length; Mann-Whitney (U) tests were used for length, height, and volume; and Wilcoxon (Z) test for height:width. In addition, ratios in 2004 and 2011 were plotted against length to determine if samples were biased by the size of the individuals.

Results

Survey data

In 2009, we surveyed for Vireos within 282 survey squares, on approximately 300 randomly distributed properties in 57 counties and 6 ecoregions in central and west Texas, and detected Vireos in 25 counties (Fig. 4). Roadside surveys were conducted in addition to property surveys in 3 counties. We recorded 460 Vireo detections at 11% ($n = 4056$) of the 5-min survey points. The subset of data used in the habitat comparisons included 2322 point-count locations (non-detection points) and 251 Vireo detections (Table 1).

In 2010, we surveyed 6207 survey points on approximately 100 properties within our 8 study areas; surveys occurred across 14 counties and 6 ecoregions. We detected Vireos within 100 m of 942 survey points (Table 2). The percent of survey points with detections within our 8 study areas generally decreased from 27% in the west to 9.4% in the east (Table 2).

Remote sensing and GIS

Two main ecosites, Low Stony Hill and Steep Rocky, made up large a percent (>10% each) of the area within our buffers in both years of study (Figs. 1, 5). Low Stony Hill is characterized by relatively gentle slopes (<7°) and shallow, well-drained, moderately permeable soils, whereas Steep Rocky is characterized by similar soils but steep slopes (>7°; NRCS 2010). From our 2009 survey data, a higher average proportion of the detection buffers was comprised of Low Stony Hill than all buffers combined (i.e., available) across the three ecoregions (Fig. 5). However, our 2010 survey results indicated that Low Stony Hill was represented at points with Vireo detections more than available at three of the study areas toward the eastern portion of the range (Kerr, Balcones, and Fort Hood) but was represented at

detections less than available for the most western study region, Devil's River. No significant difference was detected for the other study areas (Fig. 5).

For our 2009 survey data, Steep Rocky was present only at points in the Edwards Plateau ecoregion, and detection buffers had higher average proportions of Steep Rocky than all buffers combined (Fig. 5). For the 2010 survey locations, detection buffers had higher average proportion of Steep Rocky than all buffers combined for our three western study areas, whereas this ecosite occurred in few, if any, of the central and eastern study areas (Figs. 1, 5).

Several other ecosites differed significantly between Vireo detections and non-detections. The ecosite Draw, which is associated with perennial streams (NRCS 2010), was significantly higher by 58% (absolute percent) at detections than at non-detections in the Chihuahuan Desert ($t_{391} = 6.3$, $P < 0.001$). Only 12% of the total area surveyed in the Chihuahuan Desert ecoregion was categorized as Draw, but Draw made up almost 70% of the area within each detection buffer, on average.

For our 2010 data, Adobe, an upland ecosite characterized by shallow, gravelly, droughty soils and slopes ranging from 0 to 12° (NRCS 2010), showed significant differences at three of the eastern study areas, and detection buffers had higher average proportion Adobe than all buffers combined at Mason and Fort Hood (Fig. 6). Additionally, the proportion of Shallow ecosite, characterized by shallow soils with moderate slopes and low water-holding capacity, was significant but lower at detections than non-detections Kickapoo ($t_{1007} = 3.7$, $P < 0.001$; Table 1), composing only 6% on average of area within detection buffers and 13% on average of areas within non-detection buffers. Proportion of the ecosite Clay Loam, characterized by flat slopes and fertile soil with high water-holding capacity, was significantly different between detections and non-detections at Balcones ($t_{607} = 2.9$, $P = 0.003$), but Clay Loam was not present in any detection buffers and composed only 7% on average of area within non-detection buffers.

For the 2010 surveys, Vireo detections were associated with significantly steeper slopes at Devil's River, Kickapoo, Devil's Sinkhole, Mason, and Fort Hood, while Vireos were associated with less steep slopes at Balcones.

For our 2009 surveys, mean profile curvature was significantly different between Vireo detections and non-detections in the Edwards Plateau, where slopes at detection locations were slightly more concave (Table 3). For our 2010 data, profile curvature was significantly different only at Devil's River, where slopes were again more concave at detection locations (Table 4). Planimetric curvature was significantly different between detection and non-

detection points only in the Cross Timbers ecoregion, where detections were on water-collecting (horizontally concave) slopes (Table 3).

Vegetation measurements

Although the 2010 survey data suggested several statistically significant differences between Vireo detections and non-detections within three study regions, the differences may not represent biological differences (Table 4). For example, the differences in vegetation height-at-top and height-at-bottom never differed by more than 0.5 m within any study area, and the distance to vegetation from the survey point differed between detections and non-detections by approximately 1m at Kerr (Table 4).

Spatial distribution

The results of the Getis-ord General G test indicated that the Vireo detection points were clustered (Bonferroni adjusted $P < 0.006$; Miller 1981) at all study areas except Devil's River, Devil's Sinkhole, and Taylor County (Table 5). We further determined for Devil's River and Devil's Sinkhole that the clustering of both detections as well as non-detections were not the cause of the insignificant P -values, as the Moran's I test indicated no clustering was occurring (Devil's River z -value = 0.479, $P = 0.532$; Devil's Sinkhole z -value = 0.270, $P = 0.787$) However, a Moran's I test for Taylor County results showed a clustering of both detections and non-detections (z -value = 3.115, $P = 0.002$).

Comparing these results to our remote-sensing and vegetation t -tests and associated box plots, we found the metrics showed extensive overlap between detections and non-detections. Slope was the only metric with some difference between detections and non-detections at Balcones, but the average difference was only about 2° (Fig. 7). Sample sizes at Mason and Taylor were too small to draw any conclusions.

Discussion

Although we documented new records of species in 10 of 11 counties surveyed, it is unlikely that these represent range extensions. Rather, our captures likely fill gaps in data. The rate of captures in the present study was higher than those reported from West Virginia by Brack et al. (2005; 1.4 bats per net-night, diversity = 4.0) and Menzel et. al. (2002; 0.76 bats per net-night, diversity not reported). Compared to previous studies in other eastern states, our results yielded lower capture success but higher diversity. In northern Pennsylvania, Brack (2009) reported a capture rate of 2.9 bats per net-night and diversity of 3.2. The capture rate

and diversity from a similar study in Ohio was 2.4 bats per net-night and 2.9, respectively (Brack and Duffey 2006). Below, we report species-specific trends in distribution and reproductive status. Species are presented according to federal or state protection status, as classified in the Biota of Virginia database.

Non-game, protected species

Big Brown Bat. The Big Brown Bat is considered a habitat generalist (Agosta 2002) and is one of the most widespread and common species in North America. Big Brown Bats were the most frequently encountered species (28% of all individuals), with an overall capture rate of 0.55 bats per net-night (range = 0.28 to 1.01). Big Brown Bats comprised 16%, 18%, and 35% of total captures in past surveys in the Appalachians of Virginia (Hobson 1998), West Virginia (Menzel et al. 2002), and North Carolina (O’Keefe 2009), respectively. Sampling bias may partially explain the higher capture rate of adult female than adult males. Net sites over riparian areas, often thought to be foraging hot spots for adult females (Barclay 1989, Grindal et al. 1999), were sampled more frequently than other habitats. In addition, lactating females make more drinking passes than non-reproductive females (Adams and Hayes 2008).

Evidence of reproduction was documented in all counties surveyed, indicating the presumed presence of maternity colonies. While maternity colonies may contain up to several hundred adult females, adult males usually are not typically associated with maternity colonies and appear to be more dispersed (Whitaker and Hamilton 1998). Parturition occurred as early as 2 June and volant young first appeared as early as 16 June. In a building in southern Michigan, lactating Big Brown Bats were reported as early as 3 June and volant young by 23 June (Kurta 2010).

Northern Bat. The Northern Bat is a forest-obligate species that may be most abundant in the forest-dominated landscape of the Appalachian Mountains, and our capture rates are consistent with other studies from the region (Brack 2009, Brack et al. 2005, Castleberry et al. 2007, Menzel et al. 2002). Northern Bats were the second most abundant species captured (21% of all captures; 50% of all *Myotis* spp.) during our surveys, with an average capture rate of 0.41 bats per net-night (0.14 to 0.50). We obtained 5 new county records for this species (Buchanan, Pulaski, Shenandoah, Smyth, and Wise). Although we captured more adult females than adult males, we attribute these results to sampling bias (see above). While females usually form maternity colonies in trees, harboring between 30–50 bats (Carter and Feldhamer 2005, Foster and Kurta 1999, Timpone et al. 2010), males are typically more

dispersed and roost either singly or in small groups in trees, caves, mines, or manmade structures (Agosta et al. 2005, Broders and Forbes 2004, Perry and Thill 2007). Extending our sampling period, which began when maternity colonies had already been established and ended when maternity colonies were beginning to disband, may have yielded a more even adult sex ratio.

Pregnant Northern Bats were documented between 17 May and 19 June, and lactating bats were documented from 17 June to 22 July. The earliest date of capture of a juvenile Northern bat was 20 July. Similarly, Handley (1991) found this species can comprise 35% or more of *Myotis* species in late summer.

Red Bat. Red Bats are common summer residents throughout much of the eastern United States, including the Appalachian Mountains (Hutchinson and Lacki 1999, Menzel et al. 2002, O’Keefe 2009). Red Bats are seasonal migrants (Whitaker and Hamilton 1998), and although there are reports of individuals overwintering in Ohio and West Virginia (Linzey 1998), it is unknown whether or not they occur year-round in Virginia. Average capture rate for Red Bats was 0.34 bats per net-night (0.05 to 0.94) and 5 new county records were obtained (Buchanan, Dickenson, Pulaski, Russell, and Tazewell). Male Red Bats comprised over 80% of adult captures. The paucity of reproductive females captured during our surveys is consistent with other studies conducted in the Appalachian Mountains of North Carolina (O’Keefe 2009) and West Virginia (Brack 2009, Brack et al. 2005, Menzel et al. 2002). Males accounted for 93% and 95% of all Red Bat captures in Virginia (Hobson 1998) and West Virginia (Castleberry et al. 2007), respectively. Ford et al. (2002) suggest that female Red Bats are more abundant in the Coastal Plain than the Appalachian Highlands, due to warmer summer temperatures at lower elevations. In support of this idea, the majority of Red Bats captured by Johnson and Gates (2008) on Maryland’s Coastal Plain were reproductive females and juveniles.

Pregnant Red Bats were taken 15 May through 8 June, and lactating Red Bats were captured from 29 May through 11 July. The earliest capture date for a volant juvenile was 24 June. In Michigan’s Lower Peninsula, pregnant Red Bats were captured from 29 May to 28 June, lactating bats were reported from 15 June to 4 August, and earliest date of capture for juveniles was 2 July (Kurta 2010).

Little Brown Bat. The Little Brown Bat is one of the most widespread species in North America, but varies in local abundance. It was the fourth most commonly encountered species during our surveys (16% of all individuals), with an average capture rate of 0.30 bats per net-night (0.06 to 0.54). We caught this species in six counties, two of which (Buchanan and

Shenandoah) were new county records. The Little Brown Bat was the most frequently captured species reported by Castleberry et al. (2007) in West Virginia and Hobson (1998) in Virginia.

Pregnant females were captured between 6 June and 22 June, while lactating females were reported between 21 June and 18 July. Our earliest capture of a juvenile Little Brown Bat was 20 June. Kurta (2010) reported that Little Brown Bats from a maternity colony in a building in southern Michigan gave birth as early as 6 June and young were flying as early as 23 June.

Tri-colored Bat. The Tri-colored Bat is widespread throughout much of the eastern United States (Fujita and Kunz 1984) and is considered one of the most common bats in Virginia (VDGIF 1988). We captured this bat in 10 of the 11 survey counties, including two new county records (Wythe, Dickenson) at a rate of 0.21 bats per net-night (0.04 to 0.70). The male-biased sex ratio we observed seems to be typical of the Appalachian region. In a study of 11 mid-Atlantic National Parks in four physiographic provinces, 94% ($n = 16$) of adult female Tri-colored Bats were captured in the Coastal Plain and Piedmont, while only 6% ($n = 1$) were captured in the Blue Ridge and Ridge and Valley physiographic provinces (Johnson et al. 2008). In western Virginia, 88% ($n = 44$) of all Tri-colored Bats captured were males (Hobson 1998). Similarly, males comprised 94% all captures of Tri-colored Bats ($n = 17$) in West Virginia (Castleberry et al. 2007).

Pregnant females were captured between 22 May and 24 June, and earliest capture of a juvenile was 7 August. No data were available for lactating females. While comparative data on reproductive timing for this species was not available for the region of our study, Veilleux et al. (2004) reported pregnant females between 6 May and 23 June, lactating females between 29 June and 6 July, and volant young as early as 16 July in Indiana.

Silver-haired Bat. VDGIF's records of Silver-haired Bats are limited to a few scattered counties, although this species is listed as "likely" throughout the Commonwealth (Linzey 1998). We captured this species at an average rate of 0.01 bats per net-night (0 to 0.07). Captures in Buchanan and Tazewell counties were new county records. Low capture success but high likelihood of county presence may be related to their morphology and foraging habits. The Silver-haired Bat has lower wing-aspect ratio, higher wing-loading, and lower call frequencies than most species of *Myotis* (Aldridge and Rautenbach 1987), making it more suited for foraging in open habitats than in the heavily forested stream and road corridors we sampled. In the central Appalachians of West Virginia, Owen et al. (2004) documented higher rates of activity of Silver-haired Bats in relatively un-cluttered habitats such as clear-cuts.

All of the Silver-haired Bats we captured were adult males, which is consistent with Hobson's (1998) capture of 3 adult males and Cryan's (2003) study of gender segregation during the summer months. Kurta (2010) reported that males comprised 88% of adult Silver-haired Bats captured in Michigan over 32 years, and Kunz (1982) reported similar seasonal gender segregation throughout this species' eastern range. Apparently, females migrate to the northern United States and Canada to raise young, while males remain nearer to their winter range (Cryan 2003).

Hoary Bat. The Hoary Bat is listed as "likely" throughout the Commonwealth (Linzey 1988). However, each of the 6 counties where Hoary Bats were captured is a new county record. Similar to other studies conducted in the Appalachians, Hoary Bats comprised a small fraction of total captures (0.009 bats per net-night). Hoary Bats accounted for 0.005 percent of total captures in West Virginia (Castleberry et al. 2007) and 0.004 percent of total captures in Virginia (Hobson 1998).

Although Hoary Bats were infrequently captured, they are probably more widespread than our data indicate. Like Silver-haired Bats, Hoary Bats possess both morphological and acoustical traits suited for foraging in open habitats (Aldridge and Rautenbach 1987, Barclay 1985), and they frequently fly at heights in excess of 50 m (Barbour and Davis 1969). Using nets up to 20 m high, Brack (1985) found that 50% of Hoary Bat captures were at heights >8.3 m. Furthermore, in the central Appalachians of West Virginia, Owen et al. (2004) detected a greater number of echolocation calls of Hoary Bats in open- canopied habitats (e.g., clear-cuts) than in closed-canopied habitats. Because the majority of our sites were on streams or road corridors with closed canopies and nets were 5.2 to 7.8 m high, this species likely was not adequately sampled by our surveys.

Federally endangered species

Virginia Big-eared Bat. The capture of a single Virginia Big-eared Bat in Tazewell County suggests minimal species activity in summer. Dalton (1987) found this species hibernating in caves in 5 counties (Tazewell, Bland, Bath, Highland, Rockingham). R. Reynolds (VDGIF, Verona, VA, pers. comm.), W. Orndorff (Virginia Department of Conservation and Recreation, Christiansburg, VA, pers. comm.), and K. Francl (pers. observ.) confirmed the continued presence of this species outside caves during fall swarming in 2009 in Bland and Tazewell counties. The only known maternity colony for this species in Virginia is in Burke's Garden, Tazewell County (V. Brack, Jr., pers. observ.).

Gray Bat. In Virginia, previous records of the Gray Bat during summer indicate this species is restricted to just 4 caves in Lee and Scott counties (not surveyed in this project). These caves were thought to house only males (Handley 1991, Holsinger 1964). Our discovery of 4 adult males in Russell County support the idea that caves in Virginia mainly support bachelor colonies, and extends knowledge of distribution northeast to a third county. More recent surveys (August 2010) by R. Reynolds, W. Orndorff, and K. Francl (pers. observ.) also extend the range of the Gray Bat to 5 bachelor caves in Lee and Scott counties, and a single maternity colony near Bristol, VA.

Indiana Bat. Although Indiana Bats are considered more widespread in Virginia than Gray Bats, the species was thought to comprise just 1% of all bats in the Commonwealth (Handley 1991). The winter occurrence of Indiana Bats in 8 caves across 5 of our study counties (Lee, Wise, Tazewell, Bland, Bath), combined with our capture of a single male in Tazewell County, suggests that summer populations may be smaller than expected in the Cumberland Plateau and Ridge and Valley provinces. However, given that male Indiana Bats often remain close to their winter hibernacula during summer (Brack 1983, Whitaker and Brack 2002), our survey may have underestimated their abundance.

Federal species of concern

Eastern Small-footed Bat. The Eastern Small-footed Bat is listed as both a federal species of concern and a species of concern for the national forests (George Washington- Jefferson) that comprise over 7285 ha concentrated in the western portion of the state and encompassing many of our study sites (USDAFS 2004). We captured an average of 0.09 (0.01 to 0.30) Eastern Small-footed Bats per net-night and documented one new record for Buchanan County. Little is known concerning this species, but recent research indicates that Small-footed Bats may be dependent on rock outcrops for summer roosting habitat (Johnson and Gates 2008) and on surrounding forests for foraging habitat (Johnson et al. 2009). In some southwestern portions of the Commonwealth, Small-footed Bats comprise ca. 15% of *Myotis* inhabiting caves in late summer (Handley 1979, 1991).

Rafinesque's Big-Eared Bat. Rafinesque's Big-eared Bat was notably absent from our captures. The disjunct subspecies *C. rafinesquii macrotis* is listed as a state-endangered species, and is restricted to southeastern Virginia in the Coastal Plain physiographic region. However, *C. r. rafinesquii* have been captured in Pike County, KY (adjacent to Buchanan County, VA; K. Francl, pers. observ.), and Hancock County, TN (adjacent to Lee and Scott counties [not

surveyed]; Handley et al. 1979). Hence, this species is considered “likely” to be present in Lee, Scott, and Washington counties (BOVA database), which border 3 of the counties we surveyed (Wise, Russell, Smyth). Hobson (1998) did not capture any Rafinesque’s Big-eared Bats in western Virginia. However, this species is adept at eluding mist-nets, and the use of mist-nets alone is not an effective sampling protocol (Lance and Garrett 1997). We suggest additional efforts in Virginia are needed, including surveys of abandoned buildings, rock shelters, caves, and bridges throughout the study area.

Acknowledgments

We thank C. Shake, C. Diggins, and J. Stocking for assisting with field research. We thank J. Connors, K. and B. Davis, B. Desjardins, P. Doerr, J. Gerwin, M. Gibbs, R. Heiniger, D. Krementz, R. Lancia, D. McAuley, T. Simons, and C. Sorenson for providing technical assistance on various aspects of the project. Also, we are grateful to all the landowners who allowed access to their property. Funding for this project was provided by the US Fish and Wildlife Service Webless Migratory Game Bird Research Program and the Fisheries, Wildlife, and Conservation Biology Program at North Carolina State University.

Literature Cited

- Ball, B.S. 2001. Habitat use and movements of Eastern Hellbenders, *Cryptobranchus alleganiensis alleganiensis*: A radiotelemetric study. M.Sc. Thesis. Appalachian State University, Boone, NC. 101 pp.
- Briggler, J.T., J. Ettling, M. Wanner, C. Schuette, M. Duncan, and K. Goellner. 2007a. *Cryptobranchus alleganiensis* (Hellbender). chytrid fungus. Herpetological Review 38:174.
- Briggler, J., J. Utrup, C. Davidson, J. Humphries, J. Groves, T. Johnson, J. Ettling, M. Wanner, K. Traylor-Holzer, D. Reed, V. Lindgren, and O. Byers. 2007b. Hellbender population and habitat viability assessment: Final report. IUCN/SSC Conservation Breeding Specialist Group, Apple Valley, MN. 119 pp.
- Crane, A.L., and A. Mathis. 2010. Predator-recognition training: A conservation strategy to increase postrelease survival of Hellbenders in head-starting programs. Zoo Biology 29:1–12.
- Dundee, H.A., and D.S. Dundee. 1965. Observations on the systematics and ecology of *Cryptobranchus* from the Ozark Plateaus of Missouri and Arkansas. Copeia 1965:369–370.
- Freake, M.J., and E.D. Lindquist. 2008. Geographic pattern analysis of pesticide exposure

- in salamander populations in the Great Smoky Mountains National Park. *Herpetological Conservation and Biology* 3:231–238.
- Gall, B.G., and A. Mathis. 2010. Innate predator recognition and the problem of introduced trout. *Ethology* 116:47–58.
- Graham, S.P., E.C. Soehren, G.R. Cline, C.M. Schmidt, W.B. Sutton, J.R. Rayburn, S.H. Stiles, and J.A. Stiles. 2011. Conservation status of Hellbenders (*Cryptobranchus alleganiensis*) in Alabama, USA. *Herpetological Conservation and Biology* 6:242–249.
- Huang, C.C., Y. Xu, J.T. Briggler, M. McKee, P. Nam, and Y.W. Huang. 2010. Heavy metals, hematology, plasma chemistry, and parasites in adult Hellbenders (*Cryptobranchus alleganiensis*). *Environmental Toxicology and Chemistry* 29:1132–1137.
- Humphries, W.J., and T.K. Pauley. 2005. Life history of the Hellbender, *Cryptobranchus alleganiensis*, in a West Virginia stream. *The American Midland Naturalist Journal* 154:135–142.
- Mayasich, J., D. Grandmaison, and C. Phillips. 2003. Eastern Hellbender status assessment report. Illinois Natural History Survey Center for Biodiversity, Champaign, IL. 40 pp.
- Nickerson, M.A., and J.T. Briggler. 2007. Harvesting as a factor in population decline of a long-lived salamander; the Ozark Hellbender, *Cryptobranchus alleganiensis bishopi* Grobman. *Applied Herpetology* 4:207–216.
- Nickerson, M.A., and K.L. Krysko. 2003. Surveying for Hellbender salamanders, *Cryptobranchus alleganiensis* (Daudin): A review and critique. *Applied Herpetology* 1:37–44.
- Nickerson, M.A., and C.E. Mays. 1973. The Hellbenders: North American giant salamanders. Milwaukee Public Museum Publications in Biology and Geology, Milwaukee, WI. 106 pp.
- Nickerson, M.A., K.L. Krysko, and R.D. Owen. 2003. Habitat differences affecting age- class distributions of the Hellbender salamander, *Cryptobranchus alleganiensis*. *Southeastern Naturalist* 2:619–629.
- Nickerson, C.A., C.M. Ott, S.L. Castro, V.M. Garcia, T.C. Molina, J.T. Briggler, A.L. Pitt, J.J. Tavano, J.K. Byram, J. Barrila, and M.A. Nickerson. 2011. Evaluation of microorganisms cultured from injured and repressed tissue regeneration sites in endangered giant aquatic Ozark Hellbender Salamanders. *PLoS one* 6:e28906.
- North Carolina Wildlife Resources Commission (NCWRC). 2005. NC Wildlife action plan. Available online at http://www.ncwildlife.org/plan/documents/WAP_complete.pdf. Accessed 11 September 2011.

- Peterson, C.L., R.F. Wilkinson, Jr., M.S. Topping, and D.E. Metter. 1983. Age and growth of the Ozark Hellbender (*Cryptobranchus alleganiensis bishopi*). *Copeia* 1983:225–231.
- Petranka, J.W. 1998. Salamanders of the United States and Canada. Smithsonian, Washington, DC. 587 pp.
- Redmond, W.H., and A.F. Scott. 2011. Atlas of amphibians in Tennessee. Available online at <http://apbrwww5.apсу.edu/amatlas>. Accessed 19 September 2011.
- Sabatino, S.J., and E.J. Routman. 2009. Phylogeography and conservation genetics of the Hellbender salamander (*Cryptobranchus alleganiensis*). *Conservation Genetics* 10:1235–1246.
- Smith, B.G. 1907. The life history and habits of *Cryptobranchus alleganiensis*. *Biological Bulletin* 13:5–39.
- Taber, C.A., R.F. Wilkinson, Jr., and M.S. Topping. 1975. Age and growth of Hellbenders in the Niangua River, Missouri. *Copeia* 1975:633–639.
- United States Fish and Wildlife Service (USFWS). 2011. Endangered and threatened wildlife and plants; Endangered status for the Ozark Hellbender salamander. *Federal Register* 76:61956–61978. Available online at <http://federalregister.regstoday.com/ViewSummary.aspx?ds=20111006&ct=AGENCY&cv=DEPARTMENT%20OF%20THE%20INTERIOR>. Accessed 13 February 2012.
- Wheeler, B.A., E. Prosen, A. Mathis, and R.F. Wilkinson. 2003. Population declines of a long-lived salamander: A 20+ year study of Hellbenders, *Cryptobranchus alleganiensis*. *Biological Conservation* 109:151–156.

Table 1. Minimum AIC model and models within 2 AIC units for survival of Vesper Sparrow nests found on Fort McCoy Military Base, WI, May–July, 2000–2002.

| Model | Exposure | Number of | AIC | | Weight |
|---------------------------------|----------|------------|------------------|----------|--------|
| | days | parameters | AIC _c | deltaAIC | |
| Nesting stage | 921 | 2 | 306.56 | 0.00 | 0.18 |
| Nesting stage+ proportion forbs | 921 | 3 | 306.72 | 0.16 | 0.16 |

| | | | | | |
|--|-----|---|--------|------|------|
| Nesting stage + distance to woody edge | 921 | 3 | 306.79 | 0.23 | 0.16 |
| Nesting stage + proportion grass + vegetation height-density | 921 | 4 | 307.07 | 0.51 | 0.14 |
| Nesting stage + vegetation height-density | 921 | 3 | 307.20 | 0.64 | 0.13 |
| Nesting stage + proportion litter | 921 | 3 | 307.33 | 0.77 | 0.12 |
| Nesting stage + log(patch size) | 921 | 3 | 308.36 | 1.80 | 0.07 |
| Constant survival | 921 | 1 | 309.63 | 3.07 | 0.04 |

Table 2. Number of samples, median number of taxa, and diversity of benthic macroinvertebrate taxa, by pool area and distance class of 24 seasonal forest pools on the Quabbin Reservoir watershed, 1998–99.

| Area class | Distance class | | |
|-------------------------------|--------------------|------------------|------------|
| | < 200 m classes | All 200–999 m | ≥ 1000 m |
| <hr/> | | | |
| < 300 m² | | | |
| Samples | 23 67 | 25 | 19 |
| Taxa | 9 7 | 6 | 5 |
| Diversity | 1.741 1.202 | 1.169 | 0.973 |
| 300–999 m² | | | |
| Samples | 29 70 | 14 | 27 |
| Ttaxa | 6 7 | 8.5 | 8 |
| Diversity | 1.329 1.363 | 1.353 | 1.498 |
| ≥ 1000 m² | | | |
| Samples | 35 61 | 26 | no samples |
| Taxa | 7 8 | 9 | |
| Diversity | 1.492 1.593 | 1.772 | |

| | | | |
|-----------|-------|-------|-------|
| Samples | 87 | 65 | 46 |
| | 198 | | |
| Taxa | 7 | 7 | 7 |
| | 7 | | |
| Diversity | 1.460 | 1.333 | 1.280 |
| | 1.385 | | |

Table 3. Spearman rank correlations among median richness and diversity of benthic macroinvertebrates and area, isolation, and hydroperiod of 24 seasonal forest pools on the Quabbin Reservoir watershed, Massachusetts, 1998–99.

| | Maximum | Between-pool Hydroperiod surface area distance | index |
|-------------------------|---------|--|--------|
| Total community | | | |
| Richness | 0.387 | 0.001 | |
| Diversity | 0.543** | 0.344 | -0.061 |
| | 0.693** | | |
| Overwintering residents | | | |
| Richness | 0.267 | -0.152 | 0.258 |
| Diversity | 0.137 | -0.124 | 0.041 |
| Predators | | | |
| Richness | 0.054 | -0.049 | 0.487* |
| Diversity | 0.243 | -0.069 | 0.493* |

*indicates that r_s significant for $P \leq 0.05$ ($r_s \geq 0.406$).

**for $P \leq 0.01$ ($r_s \geq 0.521$)

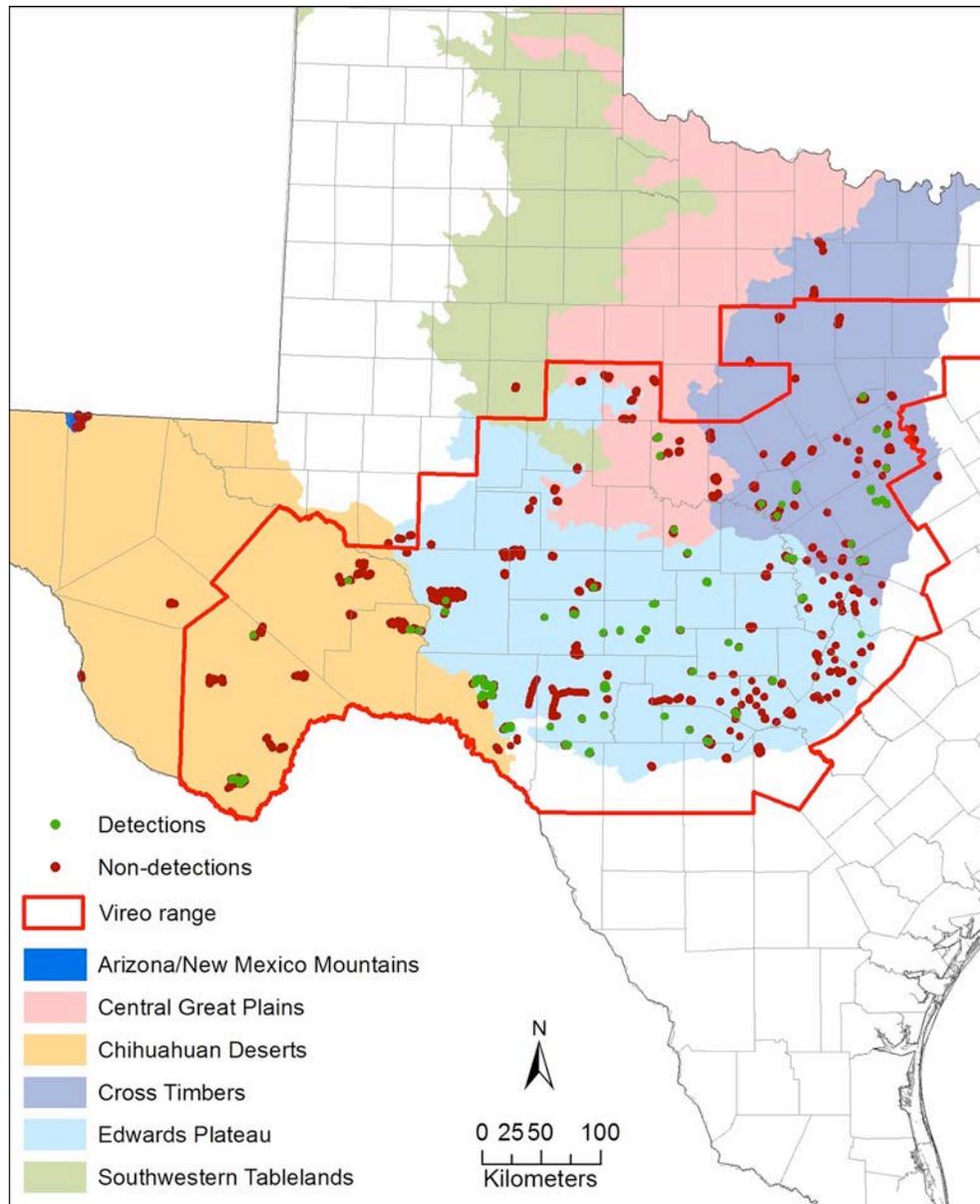


Figure 1. Results from Texas A&M 2009 Black-capped Vireo surveys. Sampling occurred in 57 counties in 8 different ecoregions across the range. Area outlined in red indicates the Vireo's breeding range in Texas as suggested for revision by the Population and Habitat Viability Assessment Report (USFWS 1996).

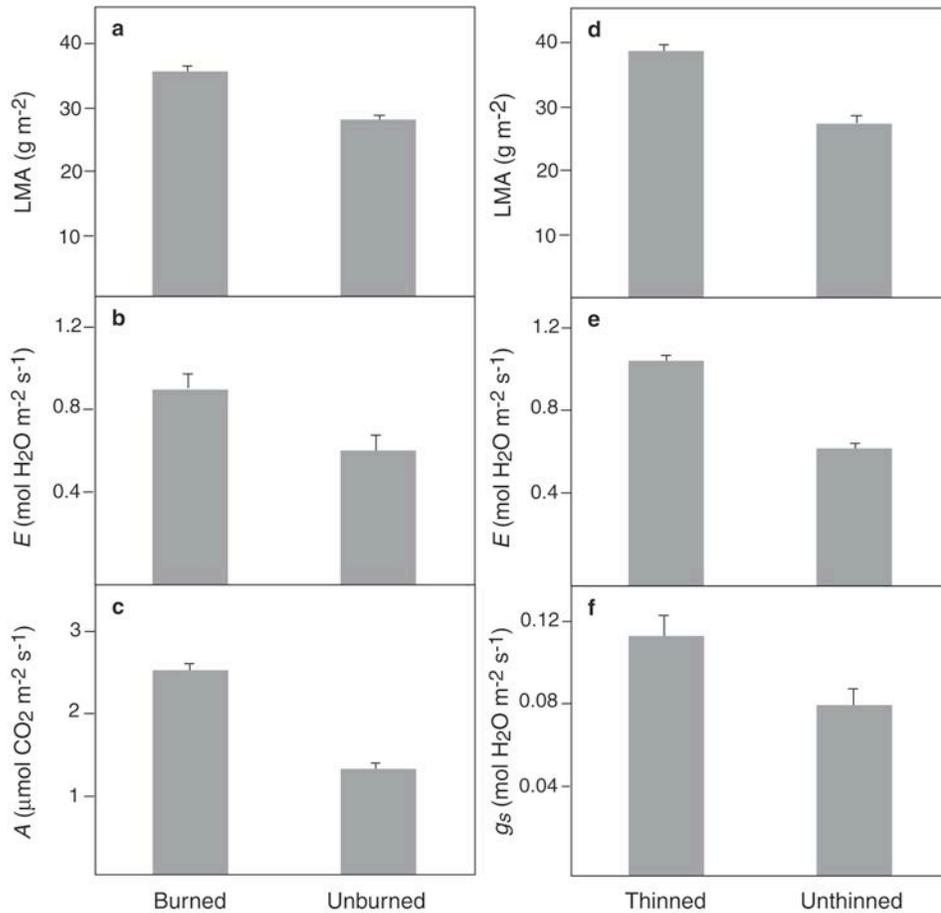


Figure 2. The significant main effect ($P \leq 0.05$) of burning on leaf mass per unit area (a; LMA in g m^{-2}), instantaneous leaf-level transpiration rate (b; E in $\text{mol H}_2\text{O m}^{-2} \text{s}^{-1}$), and instantaneous leaf-level photosynthetic rate (c; A in $\mu\text{mol CO}_2 \text{m}^{-2} \text{s}^{-1}$), and the significant main effect ($P \leq 0.05$) of thinning on LMA (d), E (e), and instantaneous stomatal conductance (f; g_s in $\text{mol H}_2\text{O m}^{-2} \text{s}^{-1}$) of transplanted *Scutellaria montana* individuals at the Tennessee Army National Guard Volunteer Training Site in Catoosa, GA. Values shown on means \pm 1 SE.

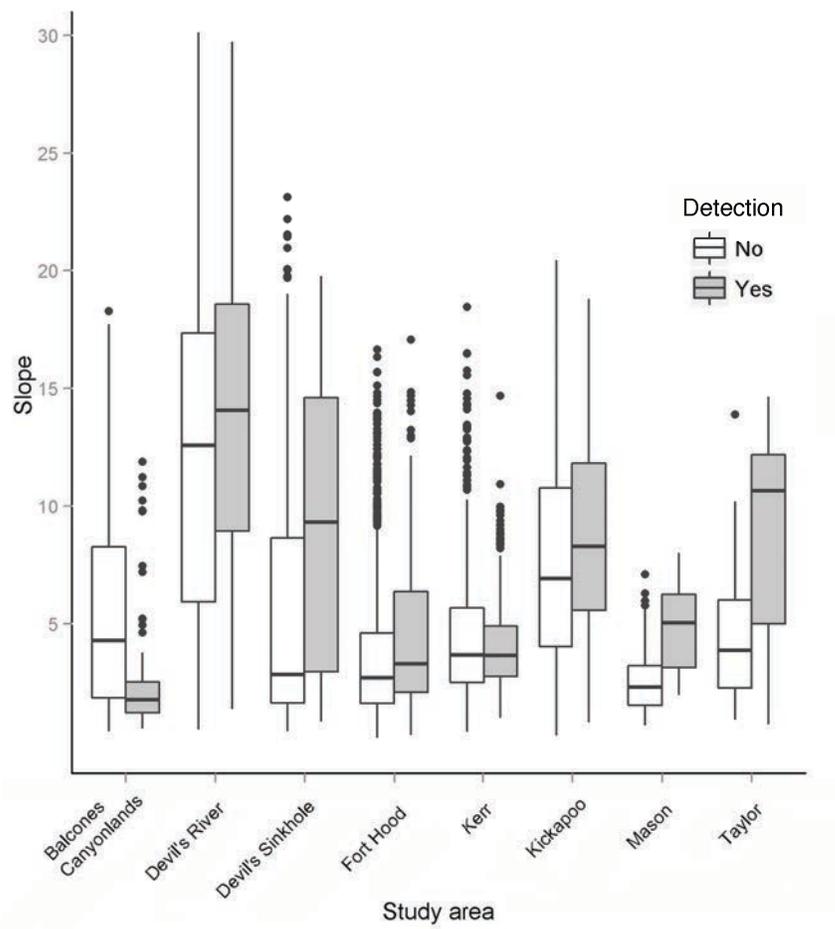


Figure 3. Box plots of average slope at each study site, comparing differences between detections and non-detections. Sample sizes are too small in Mason and Taylor to draw any conclusions from the differences. Average slope at Balcones differed with relatively little overlap, but the difference is only 2°, which is probably not biologically relevant. Boxplots for other metrics showed a similar inability to differentiate between detections and nondetections.

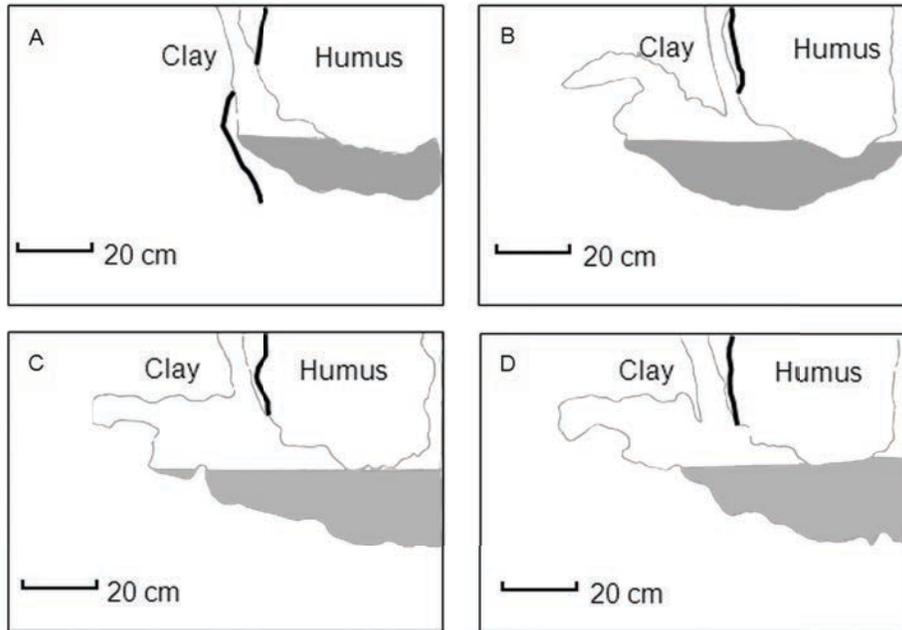


Figure 4. Four-day evolution of *C. harti* burrow in artificial burrowing chamber. Bold vertical line (straight prior to crayfish introduction) denotes the edge of the soil treatments (Clay = clayey loam, Humus = type locality soil) and shaded area is the underwater component of the burrow. Panels A, B, C, and D refer to days 1, 2, 3, and 4, respectively.

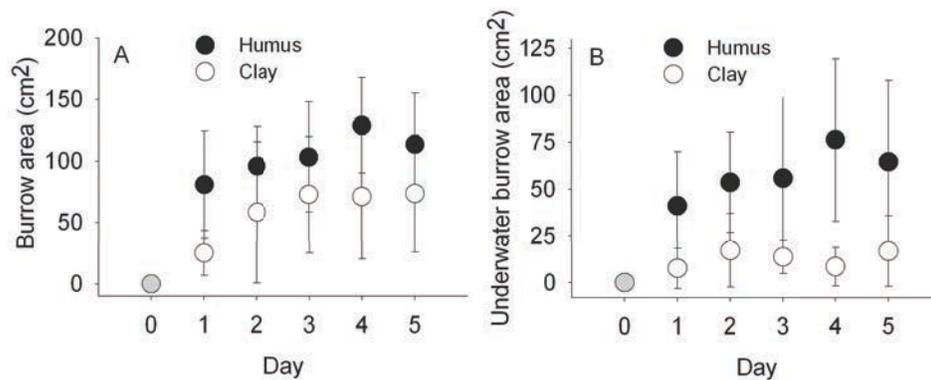


Figure 5. Total (A) and underwater (B) burrow area (cm²) for the clay and humus treatments over the 5-day trial. Points are daily means (± 1 SD) calculated from the 5 replicate ABCs. The gray point represents the beginning of the trial.



Figure 6. *Pholis gunnellus* (Rock Gunnel). Photograph © Evan Graff.