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Comparison of Small Mammal Communities in Ephemeral Wetlands and Wet Meadows during Drought

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COMPARISON OF SMALL MAMMAL COMMUNITIES IN EPHEMERAL WETLANDS AND WET MEADOWS DURING DROUGHT

being

A Thesis Presented to the Graduate Faculty
of the Fort Hays State University
in Partial Fulfillment of the Requirements for
the Degree of Master of Science

by

Brian M. Zinke
B.S., Kansas State University

Date__________________________ Approved__________________________________
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Chair, Graduate Council

This thesis for

The Master of Science Degree

By

Brian M. Zinke

has been approved

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PREFACE

This thesis is written in the style of the Journal of Mammalogy, to which a portion will be submitted for publication.

ABSTRACT

Ephemeral wetlands are characterized by a cyclical hydropattern, ranging from complete inundation to a total absence of surface water. This cycle between wet and dry phases is necessary for the flora of ephemeral wetlands to perpetuate. However, little research has been done to study the response of the non-avian fauna to these cycles, particularly during the dry phase. I live trapped small mammals by using Sherman live traps and conducted vegetation surveys monthly (May–August in 2012 and May–July in 2013) in the ephemeral wetlands and the surrounding wet meadows of the Cheyenne Bottoms basin in central Kansas. Drought occurred both years, leaving the wetlands dry; this allowed small mammal use of wetlands in the dry phase to be documented. Small mammal species richness in the 2 habitats differed by 1 the first year but was equal the second year, although species composition differed. In 2012, population estimates were higher in the wetlands than the wet meadows for *Peromyscus maniculatus*, *Sigmodon hispidus*, and *Mus musculus,* as well as in 2013 for *P. maniculatus*. Overall small mammal community estimates were higher in the wetlands than the wet meadows in both years. Small mammal survival rates varied by species and habitat. The survival rates of the overall small mammal communities were greater in the wetlands than the wet meadows in 2012, but were comparable between habitats in 2013. In both years, forbs had a higher aerial cover in the wetlands, while grasses had a higher aerial cover in the wet meadows. The height of the standing dead vegetation was taller in the wetlands than the wet meadows in 2012, but showed no difference between habitat types in 2013. These vegetational cover types, coupled with small mammal species interactions, influenced small mammal population estimates and survival rates in the 2 habitats. The vegetational cover types were also the likely reason for finding *M. musculus*, the additional species, in the wetlands. The process of wetland drawdown in a southern mixed-grass prairie ephemeral wetland greatly affected small mammal communities locally. With small mammals playing a pivotal role in many food webs, it is critical that managers understand the effects of processes, whether natural or man-induced, on small mammal communities.

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I especially thank the Kansas Department of Wildlife, Parks, and Tourism, principally Karl Grover and the staff at Cheyenne Bottoms, and The Nature Conservancy, particularly Dr. Robert L. Penner II, for letting me use their lands to conduct my research. The willingness of both entities to work with and around me was impressive, and this project would have never left the ground without their cooperation and support. I also thank Dr. Brett Sandercock from Kansas State University for his statistical guidance and willingness to help.

I thank Fort Hays State University, including the Graduate School, Sternberg Museum of Natural History, Kansas Wetlands Education Center, and the Department of Biological Sciences. The faculty has provided me with tremendous knowledge and opportunity, and has prepared me to be a professional biologist, in all senses of the term. To my fellow graduate students, we have celebrated success and struggled through hardship, only to come out the stronger. I thank you all, past and present, for the early mornings, the late nights, and everything in between. It has been an honor. I am

incredibly grateful to those who helped me in the field, including: Stasya Berber, Jessica Casey, Brian Gaston, Vickie Cikanek, Nina Luna, Brian Tanis, Taylor Rasmussen, and Samantha Pounds.

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INTRODUCTION

Wetlands are dynamic ecosystems that can exhibit extremes of the hydrologic cycle. Studies of wetlands typically occur during periods of inundation, but it is critical to understand that many wetlands are dry for at least part of the year (Zinn and Copeland 1982; Wissinger et al. 1999).

The dry phase consists of a period of drawdown, where surface water is either reduced or completely removed (Mitsch and Gosselink 2007). In prairie wetland systems, the flora and fauna are dependent on this cyclic nature between wet and dry phases to maintain diversity and productivity (Weller and Fredrickson 1974). These periodic drawdowns enhance the area for wildlife by allowing natural foods and cover to fill the dry pool bottoms (Meeks 1969).

Drawdown in prairie wetlands is typically a natural drawdown, in the form of drought. In contrast to natural drawdown, there is active, or man-induced, drawdown in which water levels are manipulated actively (Givens and Atkeson 1957; Kadlec 1962). Wetland managers often employ active drawdown to alter the natural hydrologic cycles of wetlands to an earlier successional stage to encourage the establishment of annual, seed-producing plant species, which are desirable for waterfowl (Kadlec 1962; Frederickson and Taylor 1982; Merendino et al. 1990; Mitsch and Gosselink 2007).

With the management strategy at many wetlands, focused on the avian fauna, little has been done to address the effects of drawdown, whether natural or active, on small mammal communities (Hoffman and Arbeton 1987).

The term ‘small mammal’ is a relative determination, but most would agree that small mammals are those weighing 5 kg or less (Merritt 2010). Because of their ecology and physiology, small mammals play a large but often unnoticed role in their respective ecosystems. Small mammals can alter plant species diversity and the amount of open ground in an area (Sieg 1987). Herbivorous small mammals can consume 60 to 80% of the total annual primary production in some cases, although large-scale impacts such as those typically are associated with population explosions and are uncommon (Sieg 1987). Small mammals can also increase plant stem height, abundance, diameter, and regrowth (Golley et al. 1975). Furthermore, plant community composition and species distribution can be altered by small mammal consumption and caching of seeds (Sieg 1987).

Small mammals also have a critical role in soil processes, many of which are not yet fully understood. For instance, the digging of burrows and construction of tunnels and runways brings buried soils to the surface, which mixes the soil horizons, sending organic matter below the A horizon and leached minerals back to the surface (Golley et al. 1975; Sieg 1987). This mixing of soil, coupled with the vegetative cuttings added to the litter layer by many small mammal species, increases decomposition rates as green plant material decomposes faster than brown plant material (Sieg 1987). The burrows and tunnels also facilitate increased water infiltration and aeration of the soil (Sieg 1987). Within these small mammal burrows, pH and other chemical properties of soil are altered via defecation and urination (Golley et al. 1975; Sieg 1987). Each of these aforementioned actions in turn affects the plant communities in those areas.

The most recognizable role small mammals play is in their predator/prey dynamics. Small mammals are primary consumers, as mentioned above, but are also secondary consumers feeding on large numbers of insects and insect larvae (Sieg 1987). They are also prey for many groups of predators, including canids, mustelids, snakes, and birds of prey, to name a few. Many previous studies have shown the influence of small mammal population cycles and their direct affect on the predator populations in those areas (Southern 1970; Golley et al. 1975; Sieg 1987).

Though small in stature, small mammals play pivotal roles in their ecosystems. Sieg (1987) states it best when she says, "Alteration of small mammal communities through control programs influence other components and ultimately the whole system."

In an effort to address the effects of drawdown on non-avian fauna, my study compared the small mammal communities of ephemeral wetlands and the surrounding wet meadows. Winfield et al. (1981) conducted a similar study in California's vernal pools and proposed the newly dried wetland pools would provide a potential food source and additional habitat. However, this has not been studied in the ephemeral wetlands of the Great Plains.

Thus, the purpose of my project was to 1) determine the small mammal species richness and composition in the ephemeral wetlands during the dry phase of the wetland cycle compared to nearby wet meadows; 2) compare the abundance and survival of small mammal populations and communities in the 2 habitat types; and 3) document the plant composition at both habitat types and determine its relation to the small mammal communities.

I hypothesized that 1) small mammal species richness would be higher in the wet meadows than the ephemeral wetlands; 2) small mammal species composition would be different between the ephemeral wetlands and wet meadows, with the species occurring in the ephemeral wetlands being a subset of those in the wet meadows; 3) abundances of small mammal species would be greater in the ephemeral wetlands than the wet meadows; 4) survival rates of small mammals would be greater in the wet meadows than the ephemeral wetlands; and 5) forbs would compose a greater percentage of the cover in the ephemeral wetlands compared to the wet meadows.

MATERIALS AND METHODS

Study sites – The project was conducted in the southern mixed-grass prairie region of the Great Plains in the Cheyenne Bottoms basin near Great Bend in Barton County, Kansas. The Cheyenne Bottoms basin is a 165 km^2 basin (Aschenbach and Kindscher 2006) that contains an approximately 16,600 ha freshwater wetland complex (KDWPT 2011) that is surrounded by agricultural and ranching operations, as well as the naturally occurring wet meadows. Wet meadows are grasslands that lack surface water for most of the year, but are characterized as having saturated soils near the surface (Mitsch and Gosselink 2007). The wetland complex within the Cheyenne Bottoms basin is the largest in the interior United States (Zimmerman 1990) and is recognized internationally as a major stopover point for migratory birds in the central flyway (WHSRN 2009).

Portions of the Cheyenne Bottoms basin are owned by the Kansas Department of Wildlife, Parks, and Tourism (KDWPT), The Nature Conservancy (TNC), and other private landowners. The Cheyenne Bottoms Wildlife Area (CBWA) is approximately 8,035 ha and is owned and managed by the KDWPT (KDWPT 2011). The Cheyenne Bottoms Preserve (CBP) is owned and managed by TNC and consists of another 3,237 ha (TNC 2013). Lands from both the KDWPT and TNC were incorporated in this project.

A dike system has been in place on the Cheyenne Bottoms Wildlife Area since the 1950s, creating multiple pools within the ephemeral wetlands, allowing state mangers to use active drawdown to create habitat for waterfowl and shorebirds (Griffith and Welker 1987). Additionally, canals were constructed to deliver water from the Arkansas River and Wet and Dry Walnut creeks to the Cheyenne Bottoms Wildlife Area, but that water source has since been compromised due to activities such as irrigation farming in the western portions of the state (Sadeghipour and McClain 1987). This lack of consistent water supply from local streams, coupled with the high evapotranspiration rates in the area, makes the Cheyenne Bottoms basin continually more dependent on local precipitation to fill the ephemeral wetlands (Sadeghipour and McClain 1987). Furthermore, the KDWPT can only manipulate water levels and employ active drawdown when there is water present. The Cheyenne Bottoms basin experienced drought conditions from November 2010 through September 2013 (U.S. Drought Monitor 2014), ultimately leaving the wetlands dry for much of that time (Fig. 1).

The locations for the 2 habitat types within the Cheyenne Bottoms basin were selected based on the vegetation, hydrology, and management practices of the specific locations. The ephemeral wetlands were within the dry and/or drying pools of the CBWA. The KDWPT used fire, disking, seeding, and herbicide application in the management of the CBWA and those activities were excluded in areas of this project, in coordination with the KDWPT. Wet meadows were along the periphery of the main pool system on the CBP. Management practices on the CBP included cattle grazing on a rotational basis and locations were selected based on areas that were not being grazed that particular year, in coordination with TNC.

Each habitat type contained 8 replicates and sampling was repeated for 2 consecutive years. Each replicate consisted of one 190 m transect composed of 20 trap stations placed at 10 m intervals. Transects were used instead of grids to better document small mammal species and abundance, as well as to reduce autocorrelation of vegetation data (Pearson and Ruggiero 2003). Placement of transects was at least 100 m away from edges (dikes, roads, wooded areas, fences) to remove edge effects. Along each transect I sampled small mammals and vegetational cover types.

Small mammal trapping – Small mammals were trapped by using aluminum, folding Sherman live traps (7.6 x 8.9 x 22.9 cm). Traps were baited with a mixture of creamy peanut butter and oats, wrapped in a square of wax weighing paper. The wax paper, with the bait mixture inside, was twisted into a "Hershey's kiss" shape and holes poked in it to release the scent. The "tail" of the bait was wedged into the top of the rear trap door, to ensure the trap was not set off accidently by the bait. Bait was replaced as needed throughout the trapping period. Each trap station consisted of 2 traps placed 1 m apart. Upon capture, I recorded data on: species identification, sex, mass, identification number, and if it was alive or dead. Captured individuals were tagged with a unique number by using Passive Integrated Transponder (PIT, Biomark, Boise, Idaho) tags. Some toes clips were used as well. After the first night of trapping, each captured animal was scanned to determine if it had been previously captured. On transects where toe clips were used, each captured individual was scanned as well as removed from the trap to check for a toe clip.

There was 1 trap session each month (May–August) at each habitat type. Each trap session consisted of 8 consecutive nights of trapping. Trap sessions were split into 2 4-night trapping periods, with 8 replicates being trapped the first 4 nights and the remaining 8 replicates the following 4 nights. The ephemeral wetland and wet meadow replicates were divided equally between the 2 trapping periods. Trapping was not conducted 3 nights before or after the full moon. Traps were checked in the morning beginning no earlier than 30 minutes before sunrise. Traps were closed after checking in the morning and reopened in the early evening to reduce heat-related fatalities. All tagging and handling procedures for small mammals followed protocol from the American Society of Mammalogists' Animal Care and Use Committee (Sikes et al. 2011) and was approved by the Fort Hays State University Institutional Animal Care and Use Committee protocol number 013-0009 (Appendix 1).

Vegetative sampling – A 1 m x 0.5 m modified Daubenmire frame was used to estimate aerial cover of plant species, bare ground, litter, and standing dead vegetation (Daubenmire 1959). Additionally, the height of the standing dead vegetation within the frame was measured. Vegetation was sampled on a transect 1 m to the left of and parallel to the main transect line to avoid trampling of the vegetation. I dropped the frame once per trap station (20/transect). Vegetation was measured once per month (May–August).

Statistical analysis

Statistical analyses were performed with the SPSS statistical package (Version 12.0, Chicago, Illinois, USA) with an alpha level of 0.05, unless noted otherwise. I used Program MARK (White and Burnham 1999) to create species and overall community models and to obtain survival and population estimates.

Small mammal data analysis – I reported the total numbers of individuals captured, trap nights, captures, and capture rates. Chi-square tests were used to determine whether the number of individuals captured differed between the 2 years and the 2 habitat types for each year.

I used the POPAN formulation (Schwarz and Arnason 1996) within Program MARK to obtain population size and survival rate estimates for each species of small mammal, as well as for the overall small mammal community in the 2 habitat types. The POPAN formulation assumes there is a super-population containing all of the animals that would ever be born into the population (Schwarz and Arnason 1996). From this super-population, the probability that an animal would enter the population at time *i* is estimated as b_i (Fig. 2) (Schwarz and Arnason 1996). The POPAN formulation was chosen over other formulations because it estimated abundance and allowed for losses on capture, whereas other formulations could not do both. Models to be tested were selected a priori (Table 1). Capture rates (*p*) were held constant to reduce the confounding of parameters, which would reduce the ability to accurately estimate survival rates and population size. Probabilities of entrance (*b*) were allowed to vary with time to allow population size estimates to vary over time. For each species and community, I used Program U-CARE (Choquet et al. 2009) to test the goodness-of-fit of the data to my most general model to determine that the model met the assumptions of the standard Jolly-Seber model. Models with a $\hat{c}$ value of 1 are said to fit the data perfectly (Burnham and Anderson 2010). Values of $\hat{c} > 4$ are said to have structural issues with the most general model (Burnham and Anderson 2010). Some of my models had very high $\hat{c}$ values (e.g. $\hat{c} > 8$) indicating strong lack of fit in the most general model. I ran the standard Cormack-Jolly-Seber models in program MARK for these models, which indicated age structure in the data. However, the POPAN formulation within Program MARK does not allow for age structure in the models. Thus, I ran the models as they were, noting the lack of fit due to age structure.

Akaike's Information Criterion (AIC) was used to select the "best" model, with corrections for small sample size (AIC_c) and overdispersion ($QAIC_c$) indicated by Burnham and Anderson (2010). Models were ranked by using the ΔAIC_c (the difference between the AIC_c score of a model and the AIC_c score of the "best" model (Burnham and Anderson 2010)). The "best" model is the one with the lowest (minimum) AIC_c score (Burnham and Anderson 2010). It is recommended to model average when the top 2 models have a $\Delta AIC_c < 2$, however I used model averaging when $\Delta AIC_c < 3$ to account for more of the model uncertainty (Burnham and Anderson 2010). Model averaging uses the Akaike weight (w_i) of each model to calculate a weighted average for the parameters selected (Burnham and Anderson 2010).

I assessed the alpha diversity of each habitat type by calculating species richness, evenness, and diversity. Species diversity was calculated by using the Shannon-Weiner index. For each year, I pooled the data from all 8 replicates within each habitat type to calculate these measures.

Vegetation data analysis – Cover of plant species was pooled into the following vegetational cover types: grasses, forbs, and sedges/rushes (Appendix 2). Due to violations of the assumptions of normality and equality of variances, I used nonparametric statistical tests to analyze the vegetation data. Mann-Whitney *U*-tests (Mann and Whitney 1947) were used to test for significant differences between the 2 habitat types for each vegetational cover type within each month. Friedman tests

(Friedman 1937, 1940) were used to test for significant trends over time within each vegetational cover type for both sites. Significant results from the Friedman tests were subjected to further post hoc analysis. I used Wilcoxon signed-rank tests (Wilcoxon 1945) to determine which specific time periods were significantly different.

Due to using multiple comparisons, a correction for significance level was needed to account for potential Type 1 errors. I chose to use the Benjamin and Yekutieli correction for multiple tests because it balances both Type 1 and Type 2 errors better than the Bonferroni correction (Narum 2006). A priori significance level was set at 0.05. For the Mann-Whitney *U*-tests for the 2012 data, there were 24 comparisons of cover and 4 comparisons of height that were adjusted to a significance level of $\alpha = 0.013$ and $\alpha = 0.024$, respectively. For the Mann-Whitney *U*-tests for the 2013 data, there were 18 comparisons of cover and 3 comparisons of height that were adjusted to a significance level of $\alpha = 0.014$ and $\alpha = 0.027$, respectively. For the Wilcoxon signed-rank tests for the 2012 data, there were 6 comparisons that were adjusted to a significance level of $\alpha = 0.020$. For the Wilcoxon signed-rank tests for the 2013 data, there were 3 comparisons that were adjusted to a significance level of $\alpha = 0.027$.

RESULTS

Small mammal population assessments

I captured 1,144 total individuals of 6 species in the 2012 and 2013 sampling periods (Table 2). In total, my trapping effort consisted of 17,111 trap nights with 2,529 captures (14.8% capture rate) (Table 3). Significantly more small mammals were captured overall in 2012 than 2013 ($\chi^2_1 = 425.88$, $p < 0.001$).

2012 sampling period – In 2012 I captured a total of 921 individuals of 6 species. My trapping effort was 9,696.5 trap nights with 1,901 total captures (19.6% capture rate). Significantly more small mammals were captured in the wetlands than the wet meadows ($\chi^2_1 = 592.97$, $p < 0.001$).

I was only able to run individual species models on 3 species due to small sample sizes of the other species captured. The top model for *Peromyscus maniculatus* (deer mouse) held survival constant over time but with a difference between the 2 habitat types ($w = 0.74$, $K = 12$, $-2*\text{LOG}(L) = 627.86$) (Table 4). The model with a time effect and difference between habitat types was thrown out due to poor (confounded) estimates. The top model for *Sigmodon hispidus* (hispid cotton rat) held survival constant over time ($w = 0.65$, $K = 12$, $-2*\text{LOG}(L) = 301.65$) (Table 5). The top model for *Mus musculus* (house mouse) held survival constant over time as well ($w = 0.65$, $K = 11$, $-2*\text{LOG}(L) = 159.19$) (Table 6). The 2 models that included differences in habitat type were excluded because no *M. musculus* were captured in the wet meadows.

Model averaging was used to obtain the survival rate for each of the species. Survival rates for *P. maniculatus* were over 2 times greater in the wetlands than the wet meadows (Table 7). The survival rates for *S. hispidus* were lower than *P. maniculatus*, and were nearly identical between the 2 habitat types across each month (Table 8). *Mus musculus* exhibited greater survival rates than *P. maniculatus* and *S. hispidus*, although confidence intervals were less precise (Table 9).

Model averaging was used to obtain the population estimates for each of the species. Population estimates of *P. maniculatus* and *S. hispidus* exhibited higher numbers in the wetlands than the wet meadows, even though there is some issue with the confidence intervals in the wet meadows (Tables 10 and 11, respectively). In the wetlands, *P. maniculatus* initially increased in numbers but then declined over time, whereas both *S. hispidus* and *M. musculus* (Table 12) increased continuously over time. In the wet meadows, *P. maniculatus* remained relatively constant, declining slightly over time while *S. hispidus* increased with time.

2013 sampling period – In 2013 I captured a total of 223 individuals of 6 species. My trapping effort was 7,414.5 trap nights with 628 total captures (8.5% capture rate). Significantly more small mammals were captured in the wetlands than the wet meadows ($\chi^2_1 = 74.63$, $p < 0.001$).

Due to the lower number of captures in 2013, I was only able to run individual species models on 1 species, *P. maniculatus*. The top model for *P. maniculatus* held survival constant over time with no habitat type effect ($w = 0.55$, $K = 9$, $-2*\mathrm{LOG}(L) = 262.41$) (Table 13).

The model averaged survival rates for *P. maniculatus* were slightly higher in the wet meadows than the wetlands at both time intervals (Table 14). Survival rates in the wet meadows more than doubled from the 2012 sampling period, while the wetlands were similar.

The model averaged population estimates of *P. maniculatus* were 2 to 3 times greater in the wetlands than the wet meadows (Table 15). Both habitat types exhibited the same trend, with *P. maniculatus* decreasing slightly over time. Compared to the 2012 sampling period, estimates in the wetlands were down nearly half, whereas estimates were only down slightly in the wet meadows.

Small mammal community assessments

Overall, in the 2012 sampling period I recorded the highest species richness, evenness, and diversity (Table 16). Both the wet meadows and wetlands exhibited greater species evenness and diversity in 2012 than in 2013.

2012 sampling period – Species richness, evenness, and diversity were greater in the wetland sites (S = 6, J' = 0.71, H' = 1.28). The species recorded in the wetlands were *P. maniculatus*, *S. hispidus*, *M. musculus*, *Reithrodontomys megalotis* (western harvest mouse), *Microtus ochrogaster* (prairie vole), and *Cryptotis parva* (least shrew).

The top model for the overall community data held survival constant over time with a habitat type effect ($w = 0.90$, $K = 12$, $-2*\text{LOG}(L) = 1221.26$) (Table 17). Models were not averaged because the second best model had a $\Delta\text{AIC}_c > 3$. Overall community survival rates in the wetlands were double of those in the wet meadows (Table 18). Overall community abundance estimates were more than 14 times greater in the wetlands than in the wet meadows across each time period (Table 19). Abundance estimates increased over time at both sites, with the exception of August in the wet meadows. The overall super-population estimate of the wetlands was more than 9.5 times greater than that for the wet meadows (Table 20).

2013 sampling period – Species richness was equal between the 2 habitat types (S = 5), but species evenness and diversity were greater in the wetlands (J' = 0.47, H' = 0.76). Although species richness was equal at each habitat type, the species composition differed. The species recorded in the wetlands were *P. maniculatus*, *S. hispidus*, *M. musculus*, *R. megalotis*, and *M. ochrogaster*. The difference in the wet meadows was that *M. musculus* was not captured, while *C. parva* was captured.

The top model for the overall community data held survival constant over time but had no habitat type effect ($w = 0.53$, $K = 9$, $-2*\text{LOG}(L) = 299.05$) (Table 21). Model averaging was used to obtain overall community survival rates and overall community abundance estimates. Overall community survival rates in the wet meadows were slightly higher at each time period, but were comparable to the wetlands (Table 22). Compared to the 2012 sampling period, the overall community survival rates in the wet meadows more than doubled while the wetlands remained similar. Overall community abundance estimates were 2 to 3 times greater in the wetlands than the wet meadows (Table 23). Comparatively, these abundance estimates were much lower in the wetlands and slightly higher in the wet meadows than in 2012. Over time, the overall community abundance estimates remained relatively stable in the wetlands and only slightly decreased in the wet meadows. The overall super-population estimate of the wetlands was more than 3 times greater than that for the wet meadows (Table 24). However, overall super-population estimates for both habitat types were much lower than in the 2012 sampling period.

Vegetational cover types

2012 sampling period – There were significant differences in the aerial cover of vegetational cover types between the 2 habitat types across each month (Table 25). The wet meadows had significantly higher cover of grasses each month (Fig. 3), whereas the wetlands had significantly higher cover of forbs and bare ground each month (Figs. 4 and 6). Sedges and rushes were significantly higher in the wetlands in May and June, but then declined to show no difference in July and August (Fig. 5). Litter cover showed no differences between the 2 habitat types until August, when the wet meadows were significantly higher (Fig. 7). The cover of standing dead vegetation was significantly higher in the wet meadows for May, June, and August, but showed no difference in July (Fig. 8). The height of the standing dead vegetation was significantly taller in the wetlands across each month (Table 26 and Fig. 9).

In the wetlands, there were significant differences of cover over time in grasses, forbs, sedges and rushes, bare ground, litter, and standing dead vegetation (Table 27). Grasses had significantly lower cover in July and August than in May and June (Table 28). Forbs had significantly higher cover in June, July, and August than in May (Table 28). The cover of sedges and rushes decreased significantly with each subsequent month (Table 28). Bare ground had significantly higher cover in May and August than in June and July (Table 28). The cover of litter was significantly lower in June, July, and August than in May (Table 28). Standing dead vegetation had significantly higher cover in May and July than in June and August (Table 28).

In the wet meadows, there were significant differences of cover over time in grasses, sedges and rushes, bare ground, litter, standing dead vegetation, and height of the standing dead vegetation (Table 29). The cover of grasses in June was significantly higher than May, July, and August (Table 30). Sedges and rushes had significantly higher cover in May and June than in July and August (Table 30). Bare ground had significantly lower cover in June than in July and August, but during May showed no differences from any of the other months (Table 30). The cover of litter was significantly higher in July and August than in May and June (Table 30). Standing dead vegetation decreased significantly with each subsequent month until August, which showed no difference from July (Table 30). Height of the standing dead vegetation was significantly shorter in May than in June, July, and August (Table 30).

2013 sampling period – There were significant differences in the aerial cover of vegetational cover types between the 2 habitat types across each month (Table 31). The wet meadows had significantly higher cover grasses and litter than the wetlands each month (Figs. 3 and 7), while the wetlands had a significantly higher cover of forbs (Fig. 4). The sedges and rushes were significantly higher in the wet meadows for May and June, but exhibited no difference from the wetlands in July (Fig. 5). Bare ground was significantly higher in the wetlands in May, but showed no difference between the habitat types in June and July (Fig. 6). The cover of standing dead vegetation showed no difference between the 2 habitat types in May, but was significantly higher in the wet meadows in June and July (Fig. 8). The height of the standing dead vegetation showed no significant difference between the 2 habitat types in any month (Table 32 and Fig. 9).

In the wetlands, there were significant differences of cover over time in grasses, forbs, litter, and standing dead vegetation (Table 33). Grasses had significantly higher cover in May than in June and July (Table 34). The cover of forbs was significantly different each month, with May having the lowest amount of cover and June the highest (Table 34). Litter and standing dead vegetation both had significantly higher cover in May than in June and July (Table 34).

In the wet meadows, there were significant differences of cover over time in grasses, sedges and rushes, litter, standing dead vegetation, and height of the standing dead vegetation (Table 35). The cover of grasses decreased significantly with each successive month (Table 36). Sedges and rushes had significantly lower cover in July than in May and June (Table 36). Although the Friedman test for litter cover indicated significant differences over time, the post hoc analysis indicated no significant differences between the 3 time periods (Table 36). The cover of standing dead vegetation increased significantly with each sequential month (Table 36). Height of the standing dead vegetation was significantly taller with each successive month (Table 36).

DISCUSSION

The species richness and composition in the 2 habitat types was not as I expected. I hypothesized the wet meadows would act as a species pool, and the species recorded in the drying wetlands would be a subset of the species that occurred in the wet meadows that had migrated to the newly developed habitat. However, species richness was greater in the wetlands due to the addition of *M. musculus*, which also affected the species composition (Table 2).

Significantly more individuals were captured in the wetlands, and this was represented by higher population estimates for *P. maniculatus*, *S. hispidus*, and *M. musculus* in the wetlands than the wet meadows in 2012 (Tables 10, 11, 12). Estimates for *P. maniculatus* also were higher in the wetlands in 2013 (Table 15). Consequently, the wetlands had higher community abundance estimates in both years as well (Tables 19 and 23). The difference in small mammal abundances can be attributed to the dominant vegetation types in the 2 habitats. In both years, the wetlands had significantly higher forb cover whereas the wet meadows had significantly higher grass cover each month (Tables 25 and 31). This supports previous studies on drawdown, which document annual forb species as being the first to grow in recently drawndown areas (Meeks 1969; Merendino et al. 1990). This is one of the reasons land managers employ active drawdown: to promote seed-producing plants that provide forage for waterfowl (Kadlec 1962; Frederickson and Taylor 1982; Merendino et al. 1990; Mitsch and Gosselink 2007). But small mammals are also major consumers of seeds (Sieg 1987), making recently drawndown areas ideal in terms of a food source. In both years, forb cover in the wetlands increased from May to June, then stabilized from June onward (Tables 28 and 34). Conversely, grasses in the wetlands declined from May to July and May to June in 2012 and 2013, respectively (Tables 28 and 34). The increase in forb cover coincided with the increase small mammal abundance in the wetlands in 2012 (Fig. 10). This correlation did not hold for the mammal abundance in the wetlands in 2013 (Fig. 11), but this was likely because 2 of the highly prolific species that were present in 2012, *S. hispidus* and *M. musculus*, had not recovered from their low winter numbers. Forb cover showed no significant trend over time in the wet meadows in either year (Tables 29 and 35), and neither did the small mammal abundances in the wet meadows (Figs. 12 and 13). Although other vegetational cover types showed significant differences between both habitat types and over time, none explained the difference in abundances as well as the grass and forb cover in the 2 areas.

Similar to Whitsitt and Tappe (2009), the peak population estimates for each species also varied with time. This was likely due to interactions among the different small mammals. Several of the species I captured were among those known to have negative effects on each other. For instance, *S. hispidus* is known to exclude *M. ochrogaster* when its abundances or densities become high (Frydendall 1969; Terman 1974). In 2012, *S. hispidus* abundances were large and might explain the lack of *M. ochrogaster* in my sampling areas. This does not explain, however, the lack of *M. ochrogaster* in 2013 when *S. hispidus* numbers were much lower. *Microtus* spp. exhibit pronounced population cycles ranging from incredibly high abundances to very few individuals in a given area and perhaps this study was conducted during one of the low ends of that population cycle (Merritt 2010).

Another species interaction noted in the literature is that of *M. musculus* and *P. maniculatus*. King (1957) noted that *M. musculus* and *P. maniculatus* might co-occur under favorable conditions, but that *M. musculus* would likely replace *P. maniculatus* under unfavorable conditions, such as high population density. This was likely one of the reasons why *P. maniculatus* abundance increased initially in the wetlands in 2012, but then declined as time went on and *M. musculus* abundances increased dramatically (Fig. 14).

A third interaction noted in the literature among the species I captured could be considered a neutral or perhaps even a mutualistic interaction. Catlett and Shellhammer (1962) proposed a co-species social hierarchy between *R. megalotis* and *M. musculus*. Their reasoning is these species seem to co-occur harmoniously with no antagonistic behavior between them. My data seemed to support this notion, as the number of individuals of *R. megalotis* captured was increasing in the wetlands in 2012 until the final sampling period in August, all while *M. musculus* numbers were rapidly increasing (Fig. 14). *Reithrodontomys megalotis* reproduces primarily between early spring and late autumn with reduced midsummer activity (Webster and Jones, Jr. 1982). This reduced midsummer activity could account for the decline in abundance in August, thus indicating no negative effects between *R. megalotis* and *M. musculus*. *Sigmodon hispidus* and *M. musculus* are known to be more aggressive species (Terman 1974; King 1957), whereas *P. maniculatus* and *R. megalotis* are more docile species (King 1957; Catlett and

Shellhammer 1962). These differences in behavior could explain why both *M. musculus* and *S. hispidus* (the aggressive species) were the 2 species that had rapid population increases while *P. maniculatus* and *R. megalotis* declined towards the end of the sampling period in 2012.

These species interactions might provide some basis for understanding the small mammal communities in 2012 but not 2013. This is because the 2 aggressive species whose populations increased rapidly in 2012, *M. musculus* and *S. hispidus*, were essentially non-existent in 2013 compared to 2012 (Table 2). Overall, community abundance estimates were down considerably in 2013 from 2012 (Tables 19 and 23). The drastic decline in abundances was likely due to winter mortality.

Winter and spring (December–April) temperatures were colder in the 2012–2013 period than in the 2011–2012 period (Figs. 15 and 16) (NOAA 2014). In the 2012–2013 period, the daily high temperature was equal to or below freezing 20 out of 151 days, whereas in the 2011–2012 period it was equal to or below freezing 12 out of 151 days (NOAA 2014). Additionally, in the 2012–2013 period, the daily low temperature was equal to or below freezing 114 out of 151 days, while in the 2011–2012 period it was equal to or below freezing only 87 out of 151 days (NOAA 2014). The 4 most abundant species in 2012 all declined in 2013 at both sites, except for *P. maniculatus* in the wet meadows. Notably, 2 of the most abundant species in 2012, *S. hispidus* and *M. musculus*, were nearly absent in 2013 (Table 2). *Sigmodon hispidus* is known to not overwinter well in Kansas (Frydendall 1969) because Kansas is at the northern edge of its distribution (Cameron and Spencer 1981). It is poorly adapted to cold weather as it constructs nests near the surface and stores little food, whereas the native microtine runway species, *M. ochrogaster*, nests well below the ground surface and stores considerable amounts of food for winter (Frydendall 1969). Similarly, *M. musculus* does not overwinter well unless winter temperatures are higher than average, particularly in February (Berry 1968; Frydendall 1969). *Reithrodontomys megalotis* has greater winter survival than *S. hispidus* and *M. musculus*, but *P. maniculatus* shows the greatest overwintering success of those 4 species in Kansas (Frydendall 1969). This possibly explained why *P. maniculatus* was the only species captured in 2013 in significant numbers. Other potential explanations for the decline in community and population estimates in 2013 from 2012 could be that the effects of the second year of drought were finally being felt or that there was increased predation over winter. Several bird of prey species exhibited higher than average overwintering population numbers in the winter of 2012–2013 at Cheyenne Bottoms (Robert Penner, pers. comm.). It is probable the decline in abundances of small mammals between years was due to a combination of all 3 factors.

Survival rates of each species varied greatly both between the sites and between the years. In 2012, the survival rate of *P. maniculatus* was higher in the wetlands than the wet meadows, but the survival rates in the wet meadows increased and slightly surpassed the wetlands in 2013 (Tables 7 and 14). In contrast, *S. hispidus* showed no difference in survival between the 2 habitat types in 2012 (Table 8). *Mus musculus*, while not captured in the wet meadows, had a higher survival rate than both *P. maniculatus* and *S. hispidus* in the wetlands (Table 9). The higher survival rate of *P.*

maniculatus in the wet meadows in 2013 could be due to the decreased abundance of *S. hispidus*. As mentioned earlier, *S. hispidus* is an aggressive species, and might have been antagonistic towards *P. maniculatus*.

The overall small mammal communities in each habitat type also showed differences in survival rates between sites and years. In 2012, the wetland community's survival rate was double that in the wet meadow community (Table 18). This difference could be attributed to the height of the standing dead vegetation in the wetlands. The initial drawdown left the emergent vegetation dry, ultimately leading it to die. This would have provided better cover from avian predators compared to the standing dead vegetation in the wet meadows (Table 26). This was further supported by the 2013 data, which showed no significant difference in the height of the standing vegetation between the 2 habitat types (Table 32) while survival rates showed no substantial difference between habitat types either (Table 22). However, this is likely only one of the factors affecting survival rates. In 2013, the wet meadow community's survival rate surpassed the wetland community's survival rate slightly. As mentioned above, the number of individuals of *S. hispidus* captured dropped substantially in the wet meadows from 2012 to 2013, while at the same time *P. maniculatus* numbers went up. The survival rate of *S. hispidus* was approximately 15% lower than that of *P. maniculatus* in 2012, so the switch from a *S. hispidus* dominated community to a *P. maniculatus* dominated community in 2013 would necessarily raise the overall community's survival rate. Interestingly, even though the wetlands had a more substantial food source in the forbs, this did not seem to affect the survival rates between the 2 habitat types in 2013.

The presence of the additional species of small mammal, *M. musculus*, in the wetlands but not the wet meadows was not expected. Kaufman and Kaufman (1990) noted *M. musculus* was not a major component of rodent communities in the Great Plains, which suggested a reason for not finding it in the wet meadows. However, it was a large component of the small mammal community in the wetlands. Populations of *M. musculus* can reach high densities in habitats characterized by protective cover and good food sources, such as crop fields (Houtcooper 1978; Kaufman and Kaufman 1990). Although neither of my habitat types were crop fields or near crop fields, the wetlands provided both a plentiful food source with the forbs as well as cover with the tall standing dead vegetation. This suggested the wetlands were basically mimicking a crop field setting by creating the habitat characteristics preferred by *M. musculus*, which permitted its habitation and sequential population explosion.

The similar study by Winfield et al. (1981) in California's vernal pools had different results than my study. They recorded little use of the newly dried pools by small mammals, whereas I documented unusually large numbers of small mammals using these habitats and reproducing in these areas. Although the process of drawdown is similar between these 2 wetland types, the effects of it on the small mammal community are apparently different.

Wetlands are complex systems that involve both aquatic and terrestrial ecological processes. As humans continue to manage increasingly more of earth's natural resources and systems, it is critical that we look at as many components of those systems as we can. As Winfield et al. (1981) and my studies have pointed out, similar processes in different, albeit comparable, types of wetlands can affect the fauna in substantially different ways. Small mammals are a group of species that are often neglected when management plans are developed. They play vital roles in ecosystems and when one process, such as drawdown, can produce such significant changes in abundances of small mammals, it is necessary that we take them into account as they are a critical nexus in the web of life.

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Table 1. – The 4 models selected a priori to run for each species and community of small mammal. Models showed probability of survival (ϕ), probability of capture or recapture (p), and probability of entrance (b). Probabilities were either held constant (.) or were allowed to vary with time (t) and habitat type (site).

Model
φ(.) p(.*site) b(t*site)
φ(t) p(.*site) b(t*site)
φ(.*site) p(.*site) b(t*site)
φ(t*site) p(.*site) b(t*site)

Table 2. – Total number of individual small mammals captured by species in each habitat type for both years. In 2012, trapping was conducted May–August but in 2013 only May–July were trapped due to flooding in August. For comparison, numbers in parentheses were the totals for May–July in 2012.

Species	Wetlands		Wet Meadows	
	2012	2013	2012	2013
Peromyscus maniculatus	321 (274)	135	31 (26)	38
Sigmodon hispidus	239 (169)	26	45 (31)	5
Mus musculus	209 (122)	3	0 (0)	0
Reithrodontomys megalotis	59 (52)	11	12 (10)	1
Microtus ochrogaster	1 (1)	1	2 (2)	2
Cryptotis parva	1 (1)	0	1 (1)	1
Total	830 (619)	176	91 (70)	47

Table 3. – Total captures of small mammals by species in each habitat type for both years. In 2012, trapping was conducted May–August but in 2013 only May–July were trapped due to flooding in August. For comparison, numbers in parentheses were the totals for May–July in 2012.

Species	Wetlands		Wet Meadows	
	2012	2013	2012	2013
Peromyscus maniculatus	1008 (740)	448	44 (36)	98
Sigmodon hispidus	351 (227)	37	84 (53)	8
Mus musculus	304 (158)	4	0 (0)	0
Reithrodontomys megalotis	91 (73)	28	13 (11)	1
Microtus ochrogaster	1 (1)	1	2 (2)	2
Cryptotis parva	1 (1)	0	2 (2)	1
Total	1756 (1200)	518	145 (104)	110

Table 4. – Set of models for the deer mouse (*Peromyscus maniculatus*) in 2012 showing Akaike's Information Criterion (AIC$_c$), the difference in AIC$_c$ of each model from the best model (ΔAIC$_c$), the AIC weight (*w*) of each model, which indicated relative support for that model, the number of parameters (*K*), and -2 times the logarithm of the likelihood. Models showed probability of survival (ϕ), probability of capture or recapture (p), and probability of entrance (b). Probabilities were either held constant (.) or were allowed to vary with time (t) and habitat type (site). The model fit was poor (ĉ = 7.95), but was likely due to age structure in the data, which cannot be accounted for in the POPAN formulation, thus overdispersion was not adjusted for. The top model suggested constant survival and a difference between the 2 habitat types. The model with survival as a function of time and site (t*site) was excluded due to confounding of parameters.

Model	AIC$_c$	ΔAIC$_c$	*w*	*K*	-2*LOG(*L*)
φ(.*site) p(.*site) b(t*site)	652.47	0.00	0.7403	12	627.86
φ(t) p(.*site) b(t*site)	655.42	2.95	0.1695	13	628.71
φ(.) p(.*site) b(t*site)	656.68	4.21	0.0903	11	634.16

Table 5. – Set of models for the hispid cotton rat (*Sigmodon hispidus*) in 2012 showing Akaike's Information Criterion (AIC_c), the difference in AIC_c of each model from the best model (ΔAIC_c), the AIC weight (*w*) of each model, which indicated relative support for that model, the number of parameters (*K*), and -2 times the logarithm of the likelihood. Models showed probability of survival (ϕ), probability of capture or recapture (p), and probability of entrance (b). Probabilities were either held constant (.) or were allowed to vary with time (t) and habitat type (site). $QAIC_c$ values were shown because a correction for overdispersion was used ($\hat{c} = 3.26$). The top model suggested constant survival and with no difference between the 2 habitat types.

Model	$QAIC_c$	$\Delta QAIC_c$	*w*	*K*	-2*LOG(*L*)
φ(.) p(.*site) b(t*site)	115.33	0.00	0.6443	12	301.65
φ(.*site) p(.*site) b(t*site)	117.47	2.14	0.2207	13	301.62
φ(t) p(.*site) b(t*site)	118.54	3.21	0.1297	14	298.03
φ(t*site) p(.*site) b(t*site)	124.94	9.61	0.0053	17	297.48

Table 6. – Set of models for the house mouse (*Mus musculus*) in 2012 showing Akaike's Information Criterion (AIC_c), the difference in AIC_c of each model from the best model (ΔAIC_c), the AIC weight (*w*) of each model, which indicated relative support for that model, the number of parameters (*K*), and -2 times the logarithm of the likelihood. Models showed probability of survival (ϕ), probability of capture or recapture (p), and probability of entrance (b). Probabilities were either held constant (.) or were allowed to vary with time (t) and habitat type (site). The model fit was good ($\hat{c}$ = 1.79), so overdispersion was not adjusted for. The top model suggested constant survival. The 2 models with site as a factor were excluded because no *M. musculus* were captured in the wet meadows.

Model	AIC_c	ΔAIC_c	*w*	*K*	-2*LOG(*L*)
φ(.) p(.*site) b(t*site)	182.42	0.00	0.6445	11	159.19
φ(t) p(.*site) b(t*site)	183.61	1.19	0.3555	13	155.90

Table 7. – Survival rates ($\hat{S}$) and associated standard errors (SE) and 95% confidence intervals for the deer mouse (*Peromyscus maniculatus*) at both habitat types in 2012. Survival rates were the probabilities that an individual survived from one trapping occasion to the next. Each period represented 1 month (φ_1 = May–June, φ_2 = June–July, φ_3 = July–August). Survival rates shown were the result of the model averaged estimates.

	Wetlands				Wet Meadows			
			95% Confidence Interval				95% Confidence Interval	
Period	$\hat{S}$	SE	Lower	Upper	$\hat{S}$	SE	Lower	Upper
φ_1	0.48	0.04	0.39	0.56	0.19	0.16	0.03	0.65
φ_2	0.49	0.03	0.43	0.55	0.20	0.18	0.03	0.70
φ_3	0.50	0.04	0.42	0.59	0.21	0.20	0.02	0.74

Table 8. – Survival rates ($\hat{S}$) and associated standard errors (SE) and 95% confidence intervals for the hispid cotton rat (*Sigmodon hispidus*) at both habitat types in 2012. Survival rates were the probabilities that an individual survived from one trapping occasion to the next. Each period represented 1 month (φ_1 = May–June, φ_2 = June–July, φ_3 = July–August). Survival rates shown were the result of the model averaged estimates.

	Wetlands				Wet Meadows			
			95% Confidence Interval				95% Confidence Interval	
Period	$\hat{S}$	SE	Lower	Upper	$\hat{S}$	SE	Lower	Upper
φ_1	0.33	0.11	0.16	0.57	0.34	0.12	0.15	0.60
φ_2	0.32	0.10	0.16	0.55	0.32	0.11	0.15	0.57
φ_3	0.35	0.12	0.16	0.59	0.35	0.12	0.16	0.60

Table 9. – Survival rates ($\hat{S}$) and associated standard errors (SE) and 95% confidence intervals for the house mouse (*Mus musculus*) at the wetlands in 2012. *Mus musculus* was not captured in the wet meadows in 2012. Survival rates were the probabilities that an individual survived from one trapping occasion to the next. Each period represented 1 month (φ_1 = May–June, φ_2 = June–July, φ_3 = July–August). Survival rates shown were the result of the model averaged estimates.

	Wetlands				Wet Meadows			
			95% Confidence Interval				95% Confidence Interval	
Period	$\hat{S}$	SE	Lower	Upper	$\hat{S}$	SE	Lower	Upper
φ_1	0.51	0.21	0.17	0.84	-	-	-	-
φ_2	0.62	0.21	0.22	0.90	-	-	-	-
φ_3	0.72	0.35	0.08	0.99	-	-	-	-

Table 10. – Monthly population estimates (N-hat) and associated standard errors (SE) and 95% confidence intervals for the deer mouse (*Peromyscus maniculatus*) at both habitat types in 2012.

	Wetlands				Wet Meadows			
			95% Confidence Interval				95% Confidence Interval	
Sampling Period	N-hat	SE	Lower	Upper	N-hat	SE	Lower	Upper
May	138.84	11.11	117.06	160.62	29.04	40.03	-49.41	107.49
June	162.00	10.75	140.93	183.07	15.28	20.28	-24.46	55.01
July	135.20	9.71	116.16	154.23	19.11	26.17	-32.19	70.41
August	119.83	10.32	99.61	140.04	12.75	17.48	-21.51	47.00

Table 11. – Monthly population estimates (N-hat) and associated standard errors (SE) and 95% confidence intervals for the hispid cotton rat (*Sigmodon hispidus*) at both habitat types in 2012.

	Wetlands				Wet Meadows			
			95% Confidence Interval				95% Confidence Interval	
Sampling Period	N-hat	SE	Lower	Upper	N-hat	SE	Lower	Upper
May	92.71	41.43	11.50	173.91	5.00	3.81	-2.47	12.47
June	119.42	48.31	24.74	214.10	16.69	5.60	5.70	27.67
July	119.45	48.13	25.11	213.78	16.44	5.09	6.46	26.41
August	145.79	58.80	30.54	261.03	19.71	5.34	9.25	30.17

Table 12. – Monthly population estimates (N-hat) and associated standard errors (SE) and 95% confidence intervals for the house mouse (*Mus musculus*) at the wetlands in 2012. *Mus musculus* was not captured in the wet meadows in 2012.

	Wetlands				Wet Meadows			
			95% Confidence Interval				95% Confidence Interval	
Sampling Period	N-hat	SE	Lower	Upper	N-hat	SE	Lower	Upper
May	150.08	75.97	1.18	298.99	-	-	-	-
June	156.02	74.15	10.69	301.35	-	-	-	-
July	299.96	141.88	21.88	578.03	-	-	-	-
August	478.92	233.27	21.72	936.12	-	-	-	-

Table 13. – Set of models for the deer mouse (*Peromyscus maniculatus*) in 2013 showing Akaike's Information Criterion (AIC_c), the difference in AIC_c of each model from the best model (ΔAIC_c), the AIC weight (*w*) of each model, which indicated relative support for that model, the number of parameters (*K*), and -2 times the logarithm of the likelihood. Models showed probability of survival (ϕ), probability of capture or recapture (p), and probability of entrance (b). Probabilities were either held constant (.) or were allowed to vary with time (t) and habitat type (site). The model fit was poor ($\hat{c} = 10.97$), but was likely due to age structure in the data, which cannot be accounted for in the POPAN formulation, thus overdispersion was not adjusted for. The top model suggested constant survival and with no difference between the 2 habitat types.

Model	AIC_c	ΔAIC_c	*w*	*K*	-2*LOG(*L*)
φ(.) p(.*site) b(t*site)	281.19	0.00	0.5503	9	262.41
φ(.*site) p(.*site) b(t*site)	283.10	1.91	0.2118	10	262.15
φ(t) p(.*site) b(t*site)	283.29	2.10	0.1927	10	262.33
φ(t*site) p(.*site) b(t*site)	286.19	5.00	0.0451	12	260.83

Table 14. – Survival rates ($\hat{S}$) and associated standard errors (SE) and 95% confidence intervals for the deer mouse (*Peromyscus maniculatus*) at both habitat types in 2013. Survival rates were the probabilities that an individual survived from one trapping occasion to the next. Each period represented 1 month (φ_1 = May–June, φ_2 = June–July). Survival rates shown were the result of the model averaged estimates.

	Wetlands				Wet Meadows			
			95% Confidence Interval				95% Confidence Interval	
Period	$\hat{S}$	SE	Lower	Upper	$\hat{S}$	SE	Lower	Upper
φ_1	0.47	0.05	0.38	0.56	0.48	0.11	0.29	0.68
φ_2	0.46	0.05	0.37	0.55	0.49	0.13	0.26	0.73

Table 15. – Monthly population estimates (N-hat) and associated standard errors (SE) and 95% confidence intervals for the deer mouse (*Peromyscus maniculatus*) at both habitat types in 2013.

	Wetlands				Wet Meadows			
			95% Confidence Interval				95% Confidence Interval	
Sampling Period	N-hat	SE	Lower	Upper	N-hat	SE	Lower	Upper
May	69.00	5.81	57.62	80.38	35.35	13.46	8.96	61.73
June	65.77	5.16	55.67	75.88	25.02	7.85	9.63	40.41
July	62.29	5.50	51.51	73.07	20.28	7.65	5.29	35.26

Table 16. – Small mammal species richness (S), evenness (J'), and Shannon-Weiner index (H') by habitat type for 2012 and 2013.

Site	2012			2013		
	S	J'	H'	S	J'	H'
Wet Meadows	5	0.69	1.12	5	0.44	0.71
Wetlands	6	0.71	1.28	5	0.47	0.76

Table 17. – Set of models for the overall small mammal communities in 2012 showing Akaike's Information Criterion (AIC_c), the difference in AIC_c of each model from the best model (ΔAIC_c), the AIC weight (*w*) of each model, which indicated relative support for that model, the number of parameters (*K*), and -2 times the logarithm of the likelihood. Models showed probability of survival (ϕ), probability of capture or recapture (p), and probability of entrance (b). Probabilities were either held constant (.) or were allowed to vary with time (t) and habitat type (site). The model fit was very poor ($\hat{c} = 22.29$), but was likely due to age structure in the data, which cannot be accounted for in the POPAN formulation, thus overdispersion was not adjusted for. The top model suggested constant survival and a difference between the 2 habitat types.

Model	AIC_c	ΔAIC_c	*w*	*K*	-2*LOG(*L*)
φ(.*site) p(.*site) b(t*site)	1245.53	0.00	0.9018	12	1221.26
φ(t*site) p(.*site) b(t*site)	1250.21	4.68	0.0869	16	1217.74
φ(.) p(.*site) b(t*site)	1255.11	9.58	0.0075	11	1232.89
φ(t) p(.*site) b(t*site)	1256.46	10.93	0.0038	13	1230.15

Table 18. – Survival rates ($\hat{S}$) and associated standard errors (SE) and 95% confidence intervals for the overall small mammal communities at both habitat types in 2012. Survival rates were the probabilities that an individual survived from one trapping occasion to the next. Each period represented 1 month (φ_1 = May–June, φ_2 = June–July, φ_3 = July–August). Survival rates shown were the result of the top model and were not averaged estimates.

	Wetlands				Wet Meadows			
			95% Confidence Interval				95% Confidence Interval	
Period	$\hat{S}$	SE	Lower	Upper	$\hat{S}$	SE	Lower	Upper
φ_1	0.45	0.03	0.40	0.51	0.23	0.05	0.14	0.35
φ_2	0.45	0.03	0.40	0.51	0.23	0.05	0.14	0.35
φ_3	0.45	0.03	0.40	0.51	0.23	0.05	0.14	0.35

Table 19. – Monthly community estimates (N-hat) and associated standard errors (SE) and 95% confidence intervals for the overall small mammal communities at both habitat types in 2012.

	Wetlands				Wet Meadows			
			95% Confidence Interval				95% Confidence Interval	
Sampling Period	N-hat	SE	Lower	Upper	N-hat	SE	Lower	Upper
May	341.72	32.15	284.30	410.75	21.00	4.01	14.49	30.44
June	405.09	32.79	345.75	474.61	27.06	4.05	20.21	36.24
July	419.61	34.42	357.39	492.66	29.53	4.12	22.49	38.78
August	470.12	39.28	399.22	553.61	26.54	4.02	19.76	35.64

Table 20. – Super-population estimates (N*-hat) and associated standard errors (SE) and 95% confidence intervals for the overall small mammal communities at each habitat type in 2012.

			95% Confidence Interval	
Habitat Type	N*-hat	SE	Lower	Upper
Wetlands	1461.88	64.08	1341.58	1592.96
Wet Meadows	153.62	12.43	131.14	179.97

Table 21. – Set of models for the overall small mammal communities in 2013 showing Akaike's Information Criterion (AIC_c), the difference in AIC_c of each model from the best model (ΔAIC_c), the AIC weight (*w*) of each model, which indicated relative support for that model, the number of parameters (*K*), and -2 times the logarithm of the likelihood. Models showed probability of survival (ϕ), probability of capture or recapture (p), and probability of entrance (b). Probabilities were either held constant (.) were allowed to vary with time (t) and habitat type (site). The model fit was poor ($\hat{c} = 9.89$), but was likely due to age structure in the data, which cannot be accounted for in the POPAN formulation, thus overdispersion was not adjusted for. The top model suggested constant survival and with no difference between the 2 habitat types.

Model	AIC_c	ΔAIC_c	*w*	*K*	-2*LOG(*L*)
φ(.) p(.*site) b(t*site)	317.69	0.00	0.5311	9	299.05
φ(.*site) p(.*site) b(t*site)	319.35	1.67	0.2301	10	298.57
φ(t) p(.*site) b(t*site)	319.76	2.08	0.1879	10	298.98
φ(t*site) p(.*site) b(t*site)	322.41	4.72	0.0500	12	297.29

Table 22. – Survival rates ($\hat{S}$) and associated standard errors (SE) and 95% confidence intervals for the overall small mammal communities at both habitat types in 2013. Survival rates were the probabilities that an individual survived from one trapping occasion to the next. Each period represented 1 month (φ_1 = May–June, φ_2 = June–July). Survival rates shown were the result of the model averaged estimates.

	Wetlands				Wet Meadows			
			95% Confidence Interval				95% Confidence Interval	
Period	$\hat{S}$	SE	Lower	Upper	$\hat{S}$	SE	Lower	Upper
φ_1	0.44	0.04	0.36	0.53	0.47	0.11	0.27	0.68
φ_2	0.44	0.04	0.36	0.52	0.48	0.14	0.23	0.73

Table 23. – Monthly community estimates (N-hat) and associated standard errors (SE) and 95% confidence intervals for the overall small mammal communities at both habitat types in 2013.

	Wetlands				Wet Meadows			
			95% Confidence Interval				95% Confidence Interval	
Sampling Period	N-hat	SE	Lower	Upper	N-hat	SE	Lower	Upper
May	82.00	6.55	69.16	94.84	34.30	13.17	8.49	60.11
June	76.84	5.85	65.37	88.30	28.93	9.42	10.47	47.39
July	82.54	6.25	70.28	94.79	25.63	9.65	6.70	44.55

Table 24. – Super-population estimates (N*-hat) and associated standard errors (SE) and 95% confidence intervals for the overall small mammal communities at each habitat type in 2013.

			95% Confidence Interval	
Habitat Type	N*-hat	SE	Lower	Upper
Wetlands	214.06	5.84	202.61	225.51
Wet Meadows	69.86	13.24	43.91	95.81

Table 25. – Mann-Whitney *U*-tests used to compare the ranked means of the cover (%) of vegetational cover types between the 2 habitat types for each month in 2012.

Cover Type	Wetland		Wet Meadow		Mann-Whitney	
	Mean	SE	Mean	SE	U	P
May						
Grasses	6.39	0.98	47.47	2.30	2439.0	<0.001[a]
Forbs	32.94	2.21	15.14	1.68	7662.5	<0.001[a]
Sedges/Rushes	10.05	0.88	3.58	0.33	7890.0	<0.001[a]
Bare Ground	28.69	2.60	9.64	1.17	7622.0	<0.001[a]
Litter	23.44	1.57	16.61	0.88	11054.0	0.020
Standing Dead Vegetation	18.23	1.44	37.08	2.32	7560.5	<0.001[a]
June						
Grasses	6.17	1.08	52.35	2.32	2194.0	<0.001[a]
Forbs	50.59	2.74	16.98	1.70	5433.5	<0.001[a]
Sedges/Rushes	6.92	0.72	3.34	0.26	9556.0	<0.001[a]
Bare Ground	22.56	2.35	8.84	1.03	7945.0	<0.001[a]
Litter	17.36	1.34	13.64	0.56	11798.0	0.860
Standing Dead Vegetation	15.75	1.47	33.49	1.95	5888.0	<0.001[a]
July						
Grasses	2.89	0.17	45.94	2.48	2507.5	<0.001[a]
Forbs	50.08	2.78	15.84	1.75	5448.0	<0.001[a]
Sedges/Rushes	3.13	0.22	2.67	0.12	11484.0	0.070
Bare Ground	24.42	2.39	12.11	1.27	9009.5	<0.001[a]
Litter	17.48	1.42	18.81	1.26	10600.5	0.074
Standing Dead Vegetation	20.05	1.81	23.54	1.72	10232.5	0.024
August						
Grasses	2.81	0.15	46.26	2.44	2244.0	<0.001[a]
Forbs	52.13	2.75	17.82	1.86	5518.5	<0.001[a]
Sedges/Rushes	2.50	0.00	2.58	0.08	11840.0	0.300
Bare Ground	27.39	2.42	12.47	1.36	8297.5	<0.001[a]
Litter	16.78	1.33	20.81	1.15	9197.0	<0.001[a]
Standing Dead Vegetation	14.83	1.61	22.52	1.47	7423.5	<0.001[a]

[a] Difference between wetland and wet meadow means is significant at $\alpha = 0.01324$ level, based on the Benjamin and Yekutieli correction for alpha (Narum 2006).

Table 26. – Mann-Whitney *U*-tests used to compare the ranked means of the height of the standing dead vegetation between the 2 habitat types for each month in 2012.

Cover Type	Wetland		Wet Meadow		Mann-Whitney	
	Mean	SE	Mean	SE	U	P
May	53.74	3.39	29.99	1.90	1902.5	<0.001[a]
June	54.31	3.29	37.46	2.28	2469.0	<0.001[a]
July	58.96	3.12	41.91	1.95	2281.5	<0.001[a]
August	57.05	4.20	42.45	1.89	2747.0	0.007[a]

[a] Difference between wetland and wet meadow means is significant at $\alpha = 0.024$ level, based on the Benjamin and Yekutieli correction for alpha (Narum 2006).

Table 27. – Friedman's tests used to determine any significant trends within each vegetational cover type in the wetlands in 2012.

	N	df	χ^2	p-value
Grasses	160	3	43.21	<0.001[a]
Forbs	160	3	78.63	<0.001[a]
Sedges/Rushes	160	3	153.94	<0.001[a]
Bare ground	160	3	18.80	<0.001[a]
Litter	160	3	40.07	<0.001[a]
Standing Dead Vegetation	160	3	37.24	<0.001[a]
Height of Standing Dead Vegetation	85	3	5.58	0.134

[a] Difference over time of each vegetational cover type's ranked mean is significant at $\alpha = 0.05$ level.

Table 28. – Wilcoxon signed-rank tests used to determine which months differed significantly for each of the vegetational cover types in the wetlands in 2012.

	May - June		May- July		May - Aug		June - July		June - Aug		July - Aug	
	Z	p-value	Z	p-value	Z	p-value	Z	p-value	Z	p-value	Z	p-value
Grasses	-0.70	0.484	-4.07	<0.001[a]	-3.98	<0.001[a]	-3.89	<0.001[a]	-3.75	<0.001[a]	-0.45	0.655
Forbs	-0.65	<0.001[a]	-6.48	<0.001[a]	-6.93	<0.001[a]	-0.15	0.880	-0.98	0.328	-0.90	0.369
Sedges/ Rushes	-3.74	<0.001[a]	-7.62	<0.001[a]	-8.06	<0.001[a]	-5.60	<0.001[a]	-6.09	<0.001[a]	-2.83	0.005[a]
Bare ground	-3.91	<0.001[a]	-2.53	0.011[a]	-0.53	0.594	-2.43	0.015[a]	-4.29	<0.001[a]	-2.89	0.004[a]
Litter	-4.28	<0.001[a]	-4.16	<0.001[a]	-3.77	<0.001[a]	-0.32	0.753	-0.22	0.823	-0.34	0.737
Standing Dead Vegetation	-2.50	0.013[a]	-1.16	0.244	-2.76	0.006[a]	-3.23	<0.001[a]	-1.22	0.223	-5.23	<0.001[a]
Height of Dead Vegetation	-	-	-	-	-	-	-	-	-	-	-	-

[a] Difference between months of each vegetational cover type's ranked mean is significant at $\alpha = 0.02041$ level, based on the Benjamin and Yekutieli correction for alpha (Narum 2006).

Table 29. – Friedman's tests used to determine any significant trends within each vegetational cover type in the wet meadows in 2012.

	N	df	χ^2	p-value
Grasses	149	3	17.10	0.001[a]
Forbs	149	3	3.55	0.315
Sedges/Rushes	149	3	15.60	0.001[a]
Bare ground	149	3	18.70	<0.001[a]
Litter	149	3	25.77	<0.001[a]
Standing Dead Vegetation	149	3	69.73	<0.001[a]
Height of Standing Dead Vegetation	85	3	47.76	<0.001[a]

[a] Difference over time of each vegetational cover type's ranked mean is significant at $\alpha = 0.05$ level.

Table 30. – Wilcoxon signed-rank tests used to determine which months differed significantly for each of the vegetational cover types in the wet meadows in 2012.

	May - June		May- July		May - Aug		June - July		June - Aug		July - Aug	
	Z	p-value	Z	p-value	Z	p-value	Z	p-value	Z	p-value	Z	p-value
Grasses	-4.05	<0.001[a]	-0.26	0.794	-0.29	0.770	-3.68	<0.001[a]	-3.18	0.001[a]	-0.02	0.984
Forbs	-	-	-	-	-	-	-	-	-	-	-	-
Sedges/ Rushes	-1.00	0.317	-3.21	0.001[a]	-3.36	0.001[a]	-2.83	0.005[a]	-3.00	0.003[a]	-1.00	0.317
Bare ground	-1.86	0.063	-1.79	0.073	-2.25	0.025	-3.11	0.002[a]	-4.08	<0.001[a]	-0.79	0.432
Litter	-2.28	0.023	-2.95	0.003[a]	-4.32	<0.001[a]	-4.35	<0.001[a]	-6.00	<0.001[a]	-1.32	0.188
Standing Dead Vegetation	-3.51	<0.001[a]	-6.66	<0.001[a]	-6.18	<0.001[a]	-5.56	<0.001[a]	-5.44	<0.001[a]	-1.41	0.160
Height of Dead Vegetation	-4.41	<0.001[a]	-5.56	<0.001[a]	-5.29	<0.001[a]	-2.13	0.033	-2.20	0.028	-0.07	0.955

[a] Difference between months of each vegetational cover type's ranked mean is significant at $\alpha = 0.02041$ level, based on the Benjamin and Yekutieli correction for alpha (Narum 2006).

Table 31. – Mann-Whitney *U*-tests used to compare the ranked means of the cover (%) of vegetational cover types between the 2 habitat types for each month in 2013.

Cover Type	Wetland		Wet Meadow		Mann-Whitney	
	Mean	SE	Mean	SE	U	P
May						
Grasses	8.08	0.98	61.44	1.66	781.5	<0.001[a]
Forbs	53.61	2.44	12.52	1.18	3969.0	<0.001[a]
Sedges/Rushes	2.97	0.19	5.73	0.66	11096.0	<0.001[a]
Bare Ground	26.13	2.56	12.48	1.10	10707.5	0.006[a]
Litter	9.47	1.08	11.67	0.77	9799.0	<0.001[a]
Standing Dead Vegetation	11.78	1.09	8.30	0.61	11936.5	0.235
June						
Grasses	6.77	0.84	57.16	1.92	1070.5	<0.001[a]
Forbs	66.48	2.37	11.69	1.23	2467.5	<0.001[a]
Sedges/Rushes	2.97	0.19	5.45	0.60	11102.0	<0.001[a]
Bare Ground	23.08	2.42	14.58	1.19	11816.0	0.203
Litter	6.22	0.82	11.47	0.97	8781.5	<0.001[a]
Standing Dead Vegetation	9.00	0.92	13.28	1.06	10098.0	<0.001[a]
July						
Grasses	5.97	0.67	52.80	2.19	2100.0	<0.001[a]
Forbs	63.00	2.46	13.03	1.36	3079.5	<0.001[a]
Sedges/Rushes	2.81	0.15	3.80	0.39	12076.0	0.024
Bare Ground	23.53	2.41	15.70	1.36	11746.0	0.174
Litter	5.89	0.78	10.09	0.87	9315.0	<0.001[a]
Standing Dead Vegetation	8.61	0.72	16.66	1.43	9697.5	<0.001[a]

[a] Difference between wetland and wet meadow means is significant at $\alpha = 0.01431$ level, based on the Benjamin and Yekutieli correction for alpha (Narum 2006).

Table 32. – Mann-Whitney *U*-tests used to compare the ranked the means of the height of the standing dead vegetation between the 2 habitat types for each month in 2013.

Cover Type	Wetland		Wet Meadow		Mann-Whitney	
	Mean	SE	Mean	SE	U	P
May	63.05	5.17	35.76	1.45	4550.5	0.185
June	60.07	4.89	37.66	1.25	4744.0	0.391
July	59.85	4.49	41.18	1.25	4821.0	0.501

[a] Difference between wetland and wet meadow means is significant at $\alpha = 0.027$ level, based on the Benjamin and Yekutieli correction for alpha (Narum 2006).

Table 33. – Friedman's tests used to determine any significant trends within each vegetational cover type in the wetlands in 2013.

	N	df	χ^2	p-value
Grasses	160	2	21.81	<0.001[a]
Forbs	160	2	65.63	<0.001[a]
Sedges/Rushes	160	2	0.89	0.641
Bare Ground	160	2	1.07	0.585
Litter	160	2	47.28	<0.001[a]
Standing Dead Vegetation	160	2	24.05	<0.001[a]
Height of Standing Dead Vegetation	101	2	0.60	0.740

[a] Difference over time of each vegetational cover type's ranked mean is significant at $\alpha = 0.05$ level.

Table 34. – Wilcoxon signed-rank tests used to determine which months differed significantly for each of the vegetational cover types in the wetlands in 2013.

	May - June		May- July		June - July	
	Z	p-value	Z	p-value	Z	p-value
Grasses	-2.46	0.014[a]	-3.72	<0.001[a]	-1.79	0.074
Forbs	-6.45	<0.001[a]	-5.55	<0.001[a]	-2.29	0.022[a]
Sedges/Rushes	-	-	-	-	-	-
Bare Ground	-	-	-	-	-	-
Litter	-4.50	<0.001[a]	-4.75	<0.001[a]	-0.84	0.403
Standing Dead Vegetation	-4.12	<0.001[a]	-4.02	<0.001[a]	-1.25	0.213
Height of Dead Vegetation	-	-	-	-	-	-

[a] Difference between months of each vegetational cover type's ranked mean is significant at $\alpha = 0.027$ level, based on the Benjamin and Yekutieli correction for alpha (Narum 2006).

Table 35. – Friedman's tests used to determine any significant trends within each vegetational cover type in the wet meadows in 2013.

	N	df	χ^2	p-value
Grasses	160	2	21.67	<0.001[a]
Forbs	160	2	2.81	0.245
Sedges/Rushes	160	2	17.31	<0.001[a]
Bare Ground	160	2	5.71	0.057
Litter	160	2	8.44	0.015[a]
Standing Dead Vegetation	160	2	19.60	<0.001[a]
Height of Standing Dead Vegetation	101	2	20.14	<0.001[a]

[a] Difference over time of each vegetational cover type's ranked mean is significant at $\alpha =$ 0.05 level.

Table 36. – Wilcoxon signed-rank tests used to determine which months differed significantly for each of the vegetational cover types in the wet meadows in 2013.

	May - June		May- July		June - July	
	Z	p-value	Z	p-value	Z	p-value
Grasses	-3.04	0.002[a]	-4.31	<0.001[a]	-3.49	<0.001[a]
Forbs	-	-	-	-	-	-
Sedges/Rushes	-0.60	0.547	-3.14	0.002[a]	-3.57	<0.001[a]
Bare Ground	-	-	-	-	-	-
Litter	-0.08	0.940	-1.49	0.137	-1.92	0.054
Standing Dead Vegetation	-4.42	<0.001[a]	-4.80	<0.001[a]	-3.44	0.001[a]
Height of Dead Vegetation	-2.41	0.016[a]	-3.89	<0.001[a]	-3.78	<0.001[a]

[a] Difference between months of each vegetational cover type's ranked mean is significant at α = 0.027 level, based on the Benjamin and Yekutieli correction for alpha (Narum 2006).

Fig. 1. – Drought ratings for Barton County, KS, where the Cheyenne Bottoms basin was located. Data were adapted from the U.S. Drought Monitor website (U.S. Drought Monitor 2014).

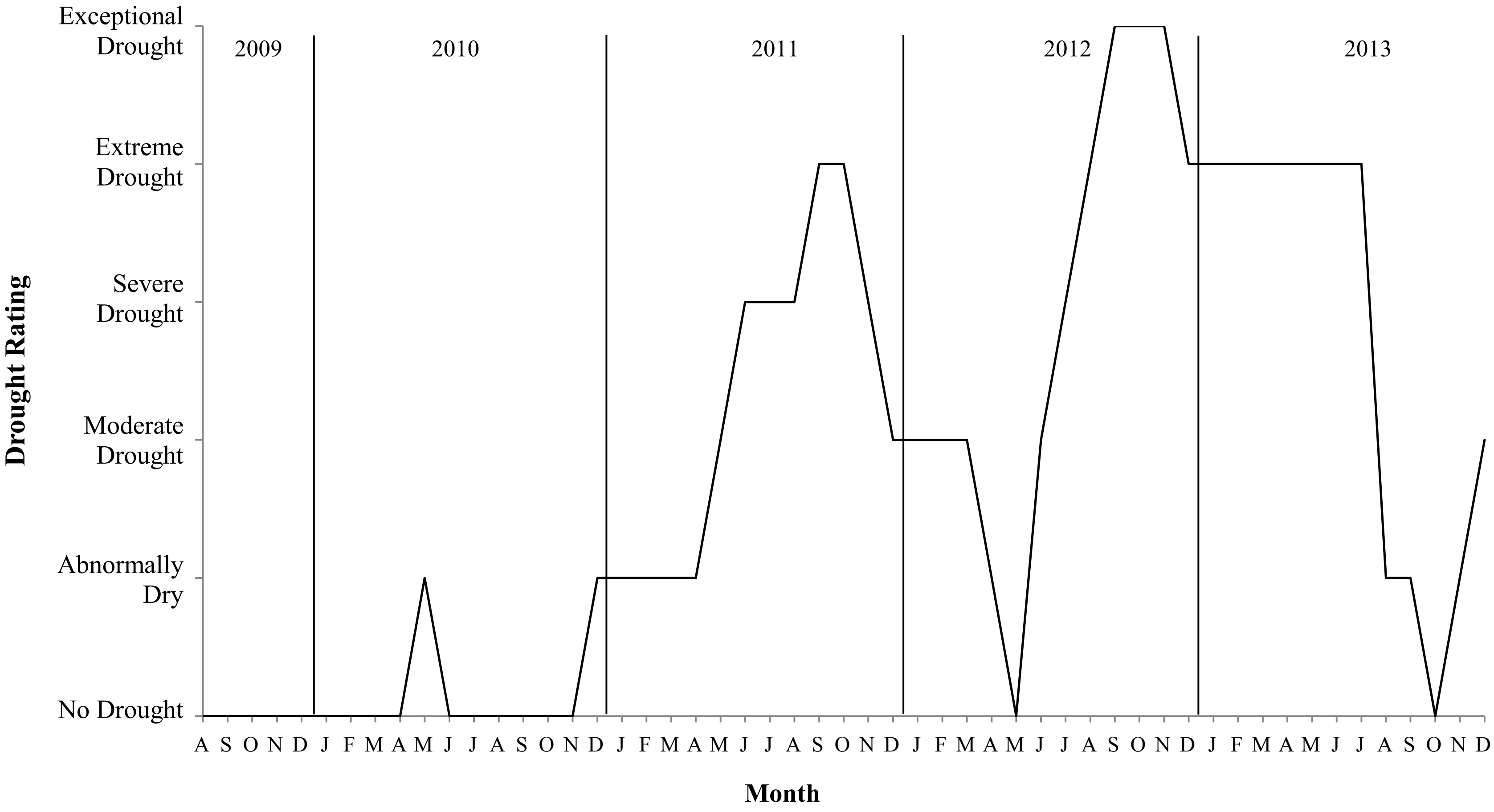


Fig. 2. – POPAN parameters in a study with 4 sampling occasions ($t_1 – t_4$). Probability of capture was represented by p_i at occasion i. Probability of an animal surviving from occasion i to occasion $i + 1$ was represented by ϕ_i. Probability that an animal entered the population from the super-population between occasions i and $i + 1$ was represented by b_i.

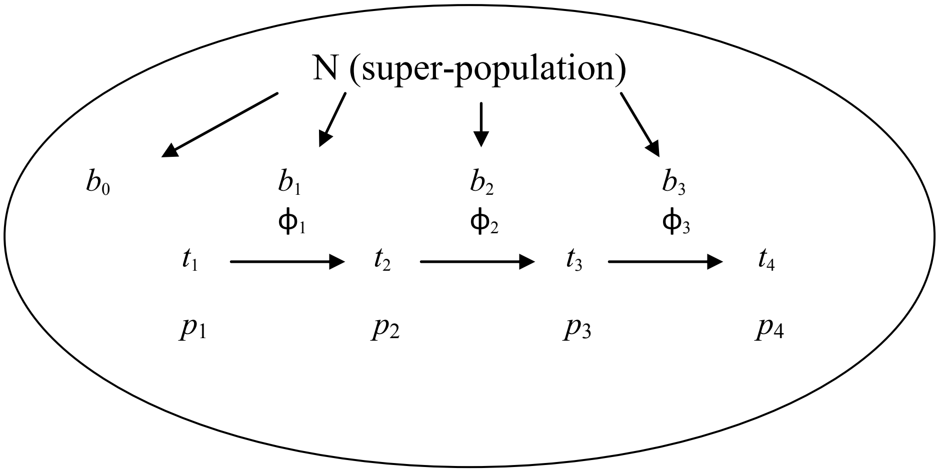


Fig. 3. – Aerial cover and standard error of grasses in the wetlands (solid) and wet meadows (dashed) in a) 2012 and b) 2013.

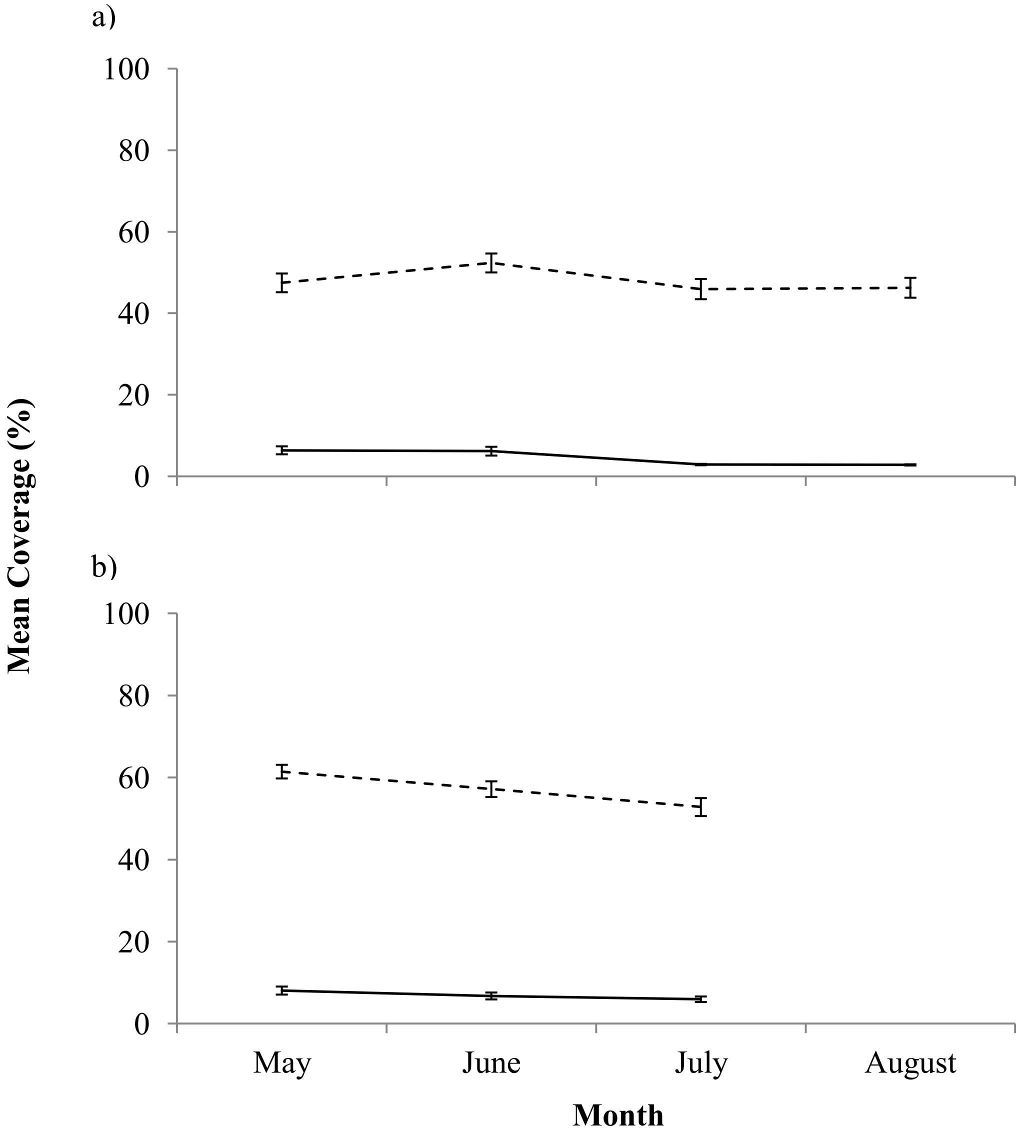


Fig. 4. – Aerial cover and standard error of forbs in the wetlands (solid) and wet meadows (dashed) in a) 2012 and b) 2013.

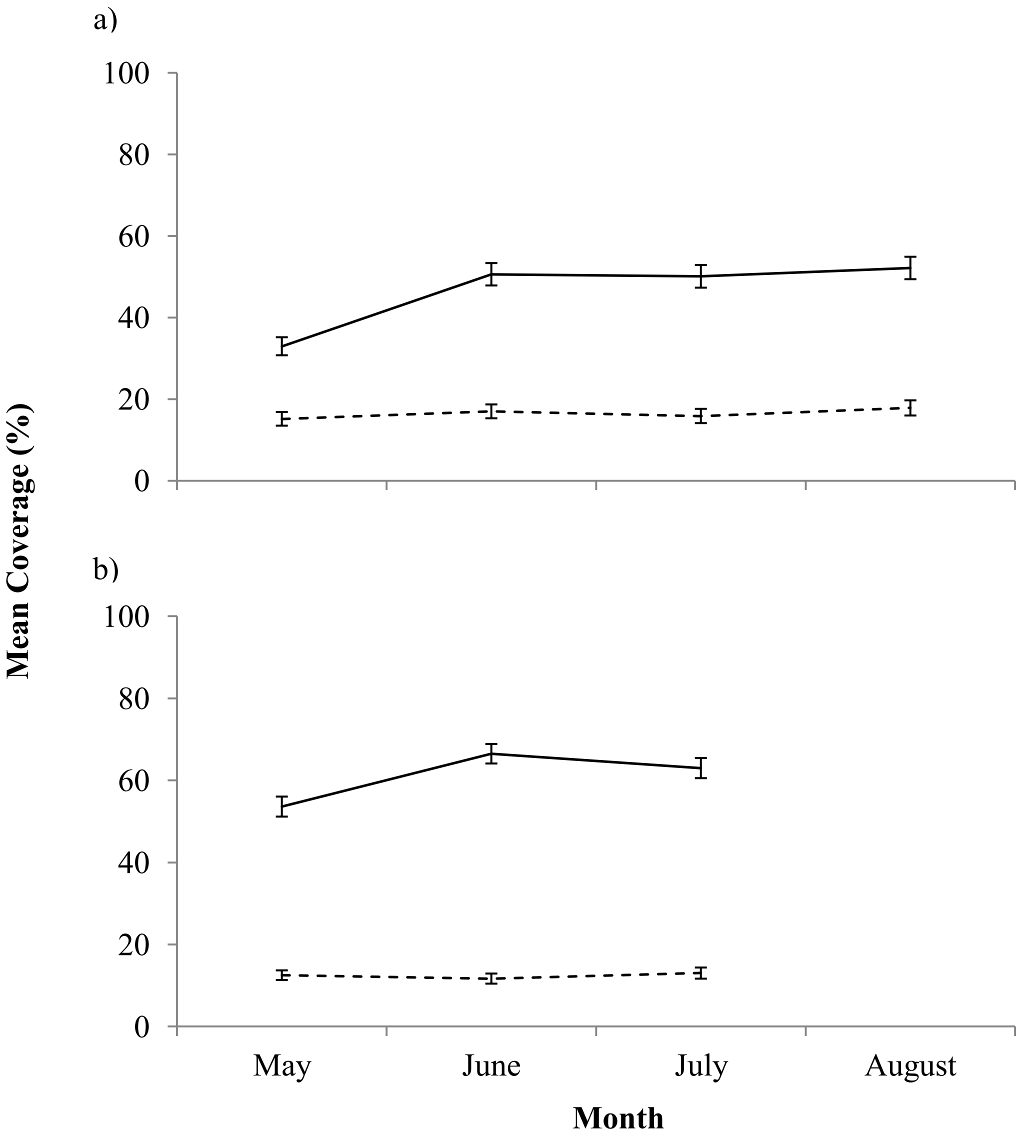


Fig. 5. – Aerial cover and standard error of sedges and rushes in the wetlands (solid) and wet meadows (dashed) in a) 2012 and b) 2013.

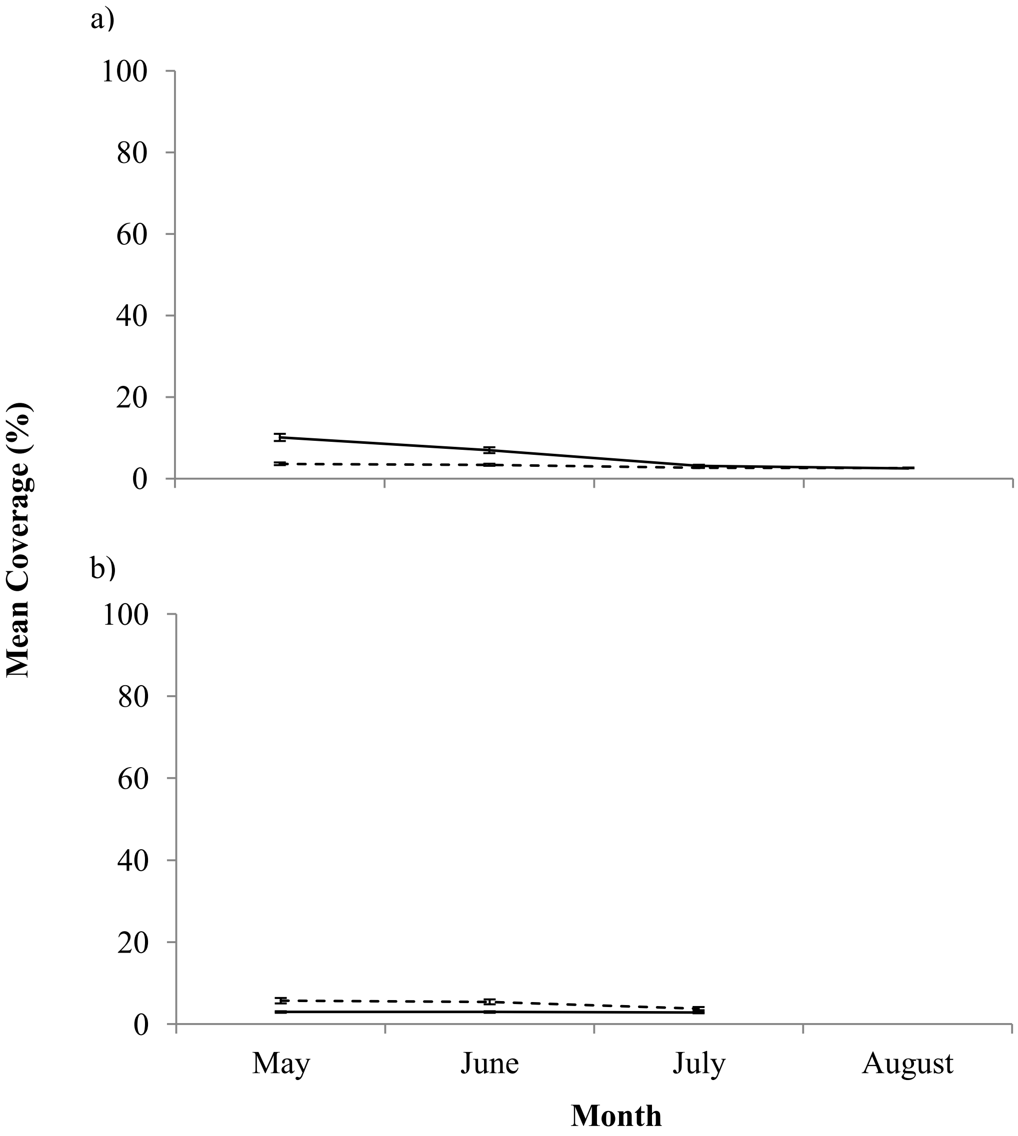


Fig. 6. – Aerial cover and standard error of bare ground in the wetlands (solid) and wet meadows (dashed) in a) 2012 and b) 2013.

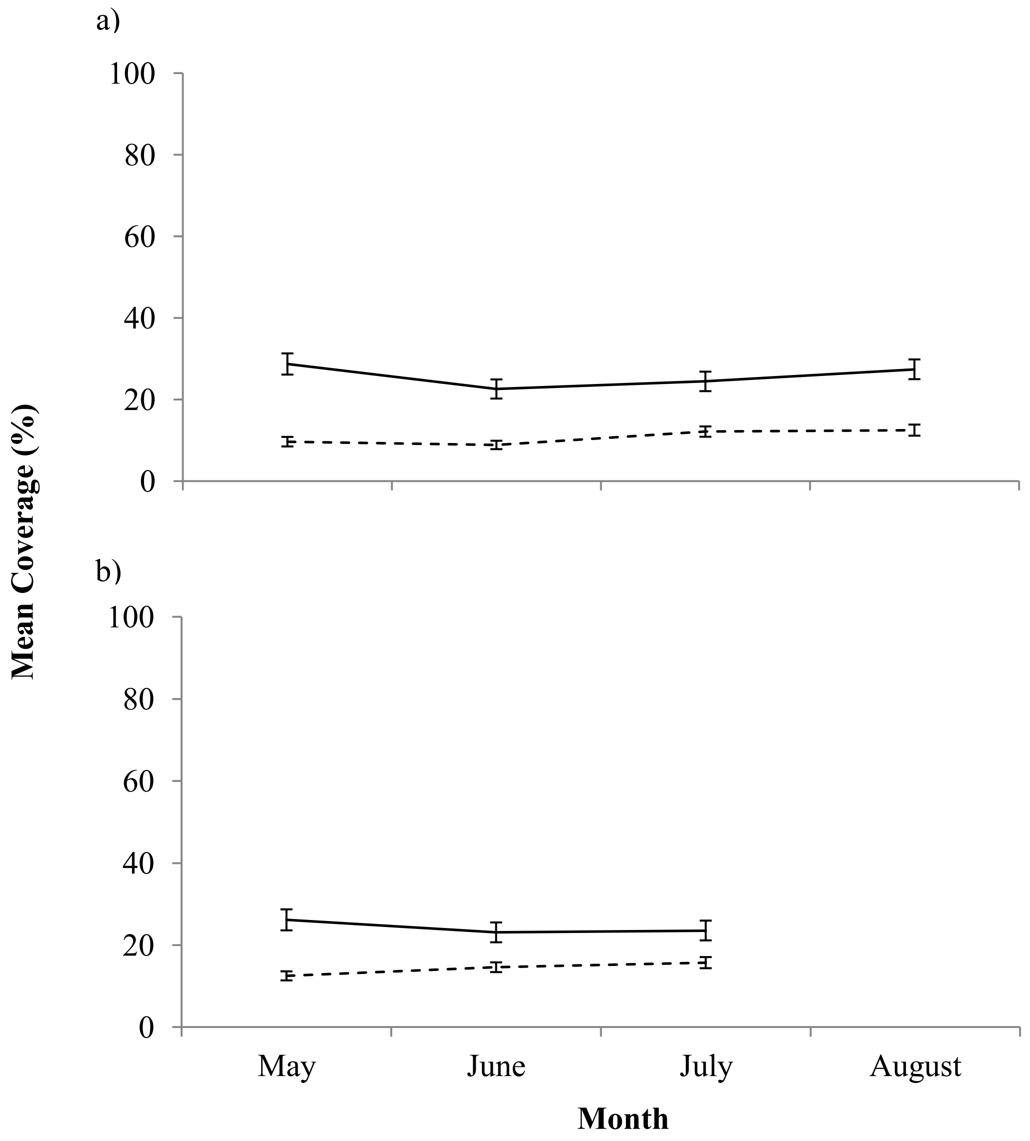


Fig. 7. – Aerial cover and standard error of litter in the wetlands (solid) and wet meadows (dashed) in a) 2012 and b) 2013.

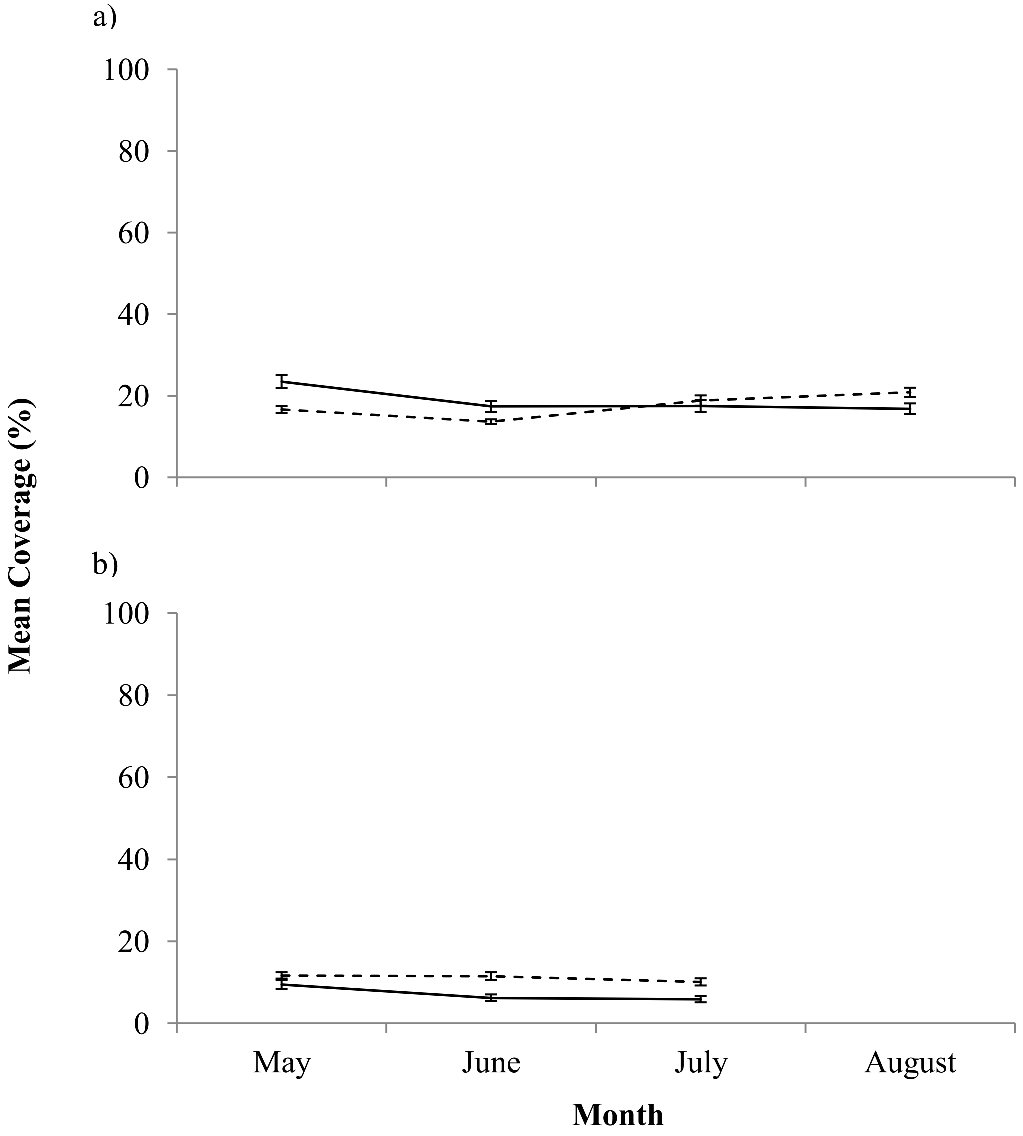


Fig. 8. – Aerial cover and standard error of standing dead vegetation in the wetlands (solid) and wet meadows (dashed) in a) 2012 and b) 2013.

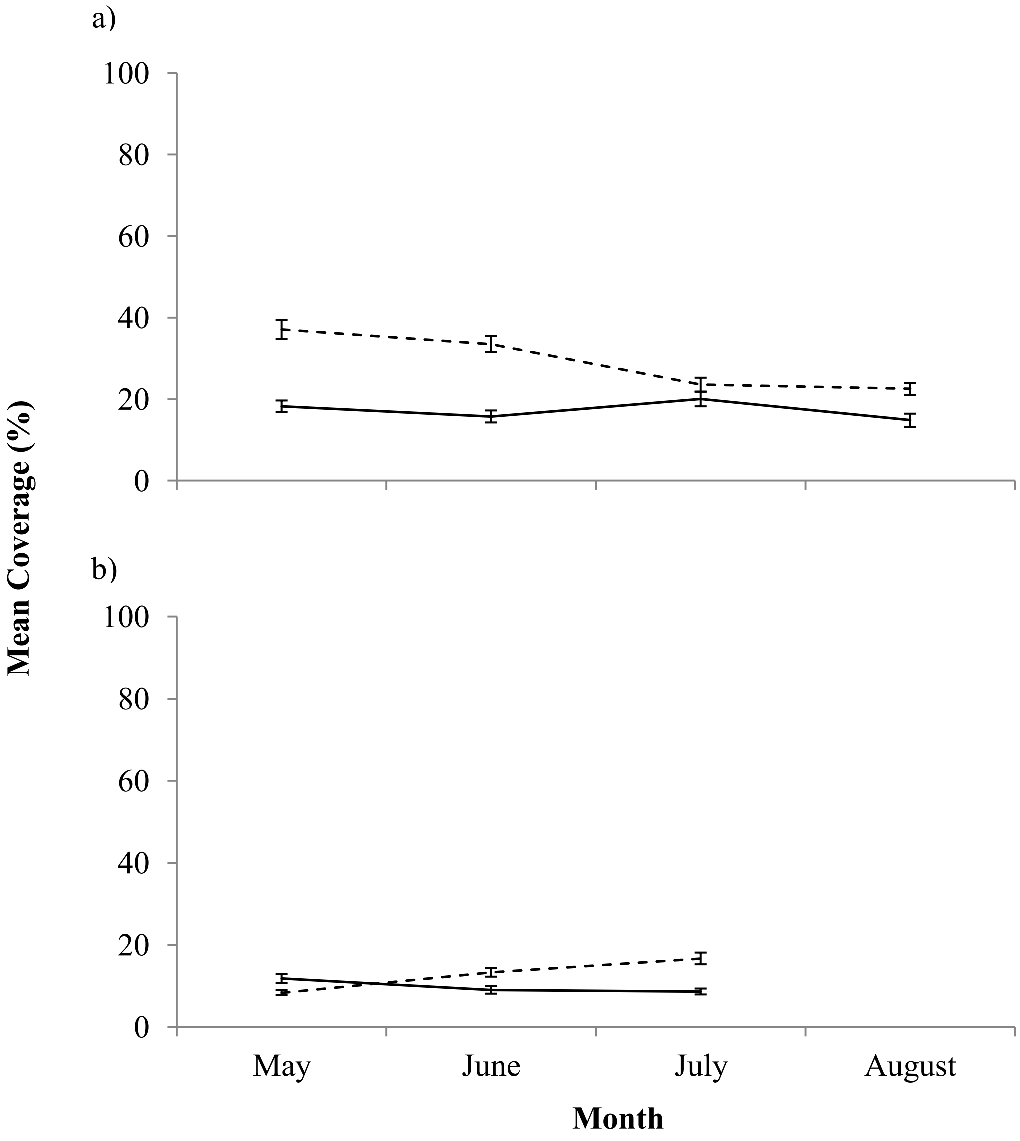


Fig. 9. – Height and standard error of the standing dead vegetation in the wetlands (solid) and wet meadows (dashed) in a) 2012 and b) 2013.

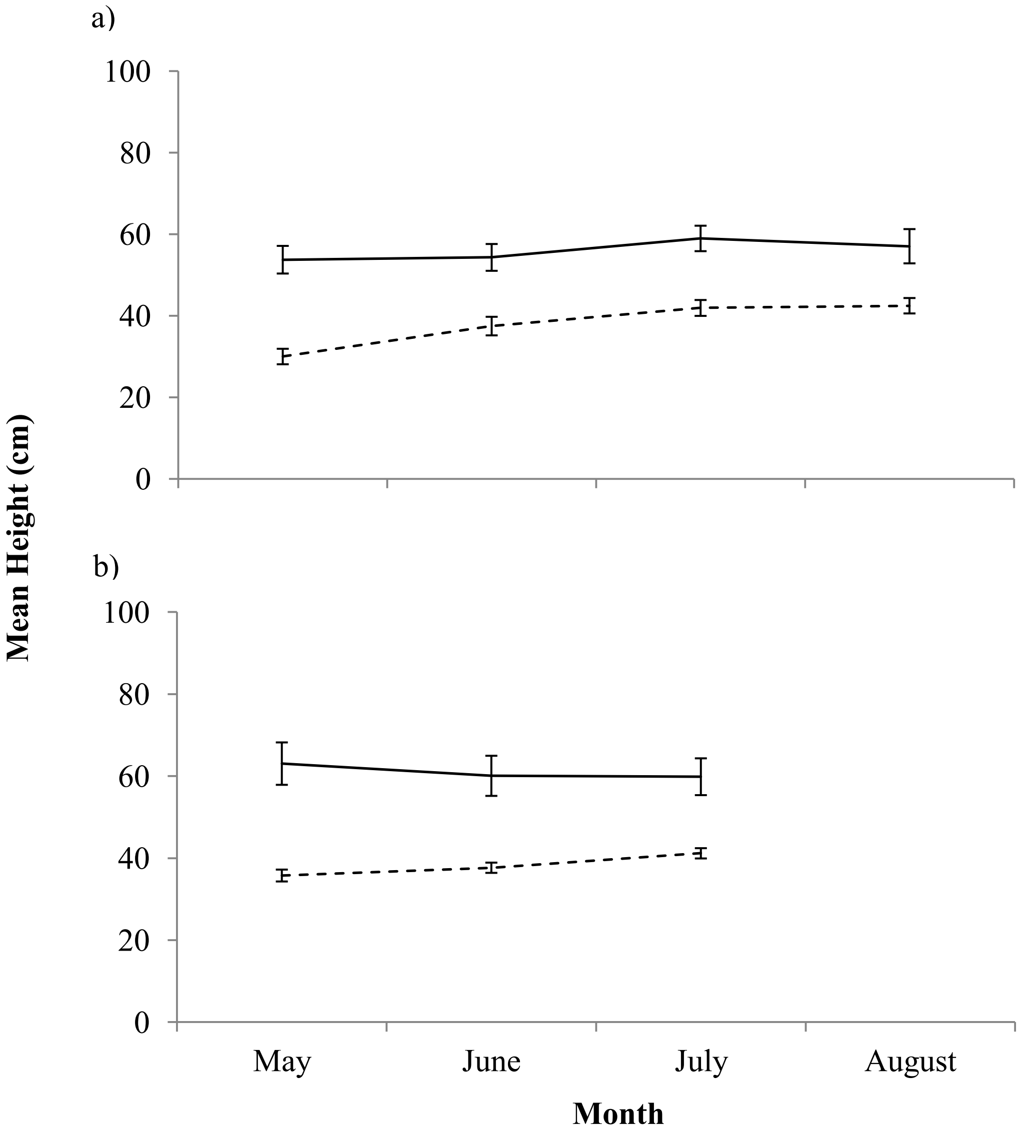


Fig. 10. – Number of individual small mammals captured (bars) each month in the wetlands in 2012, and the cover (%) of forbs (solid) and grasses (dashed).

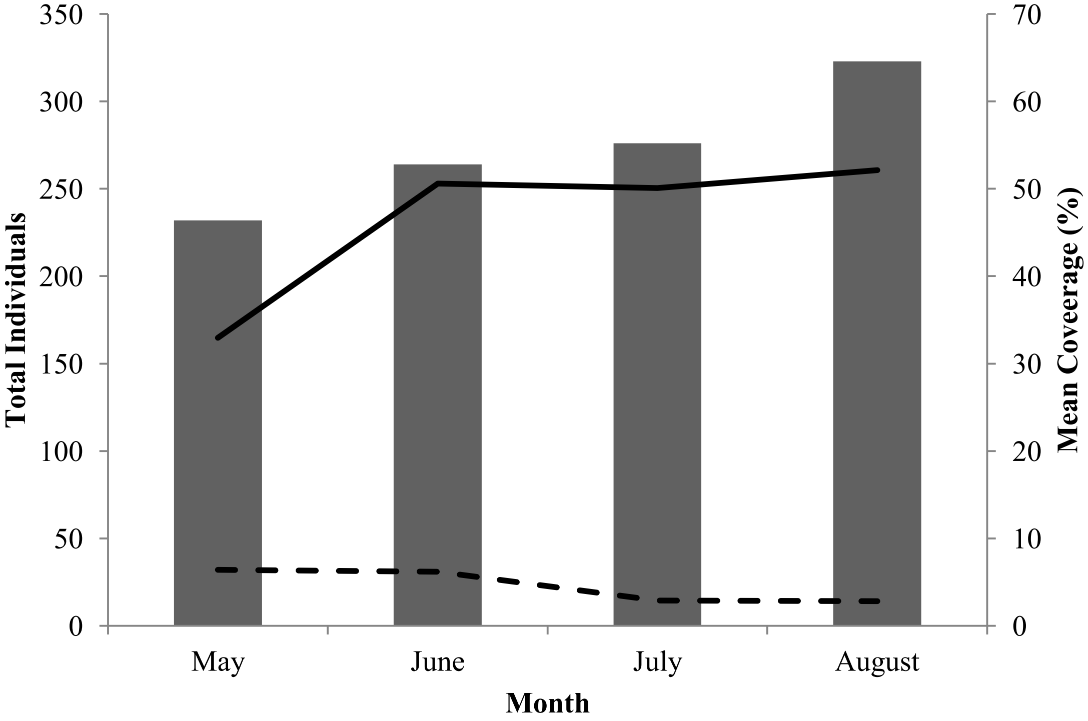


Fig. 11. – Number of individual small mammals captured (bars) each month in the wetlands in 2013, and the cover (%) of forbs (solid) and grasses (dashed).

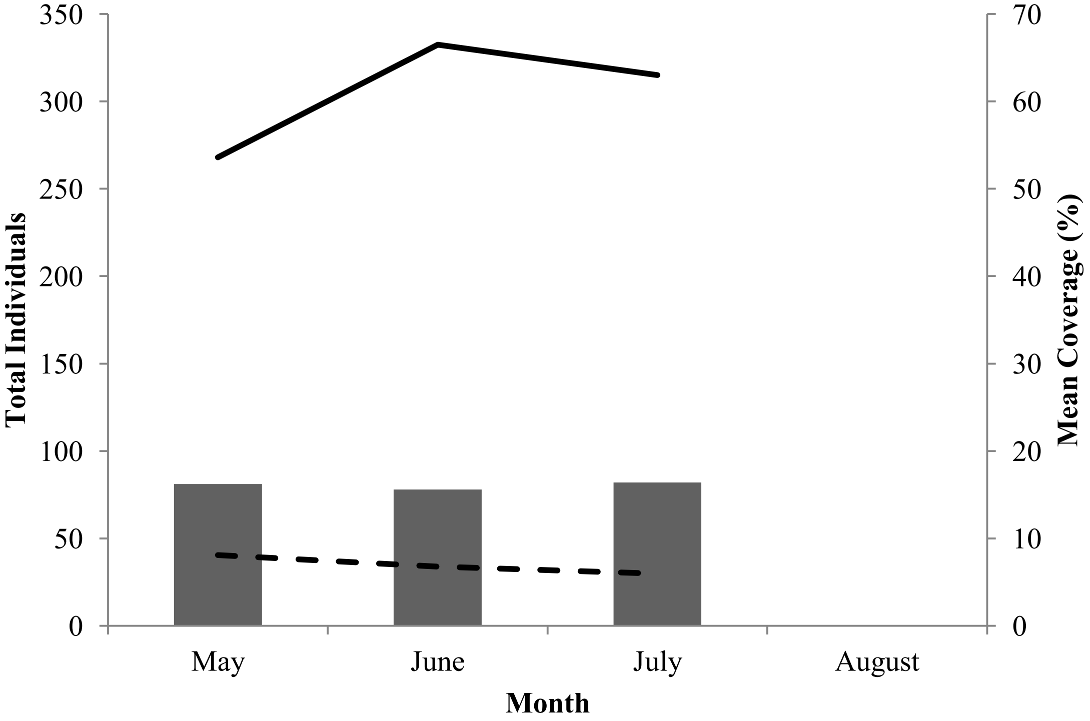


Fig. 12. – Number of individual small mammals captured (bars) each month in the wet meadows in 2012, and the cover (%) of forbs (solid) and grasses (dashed).

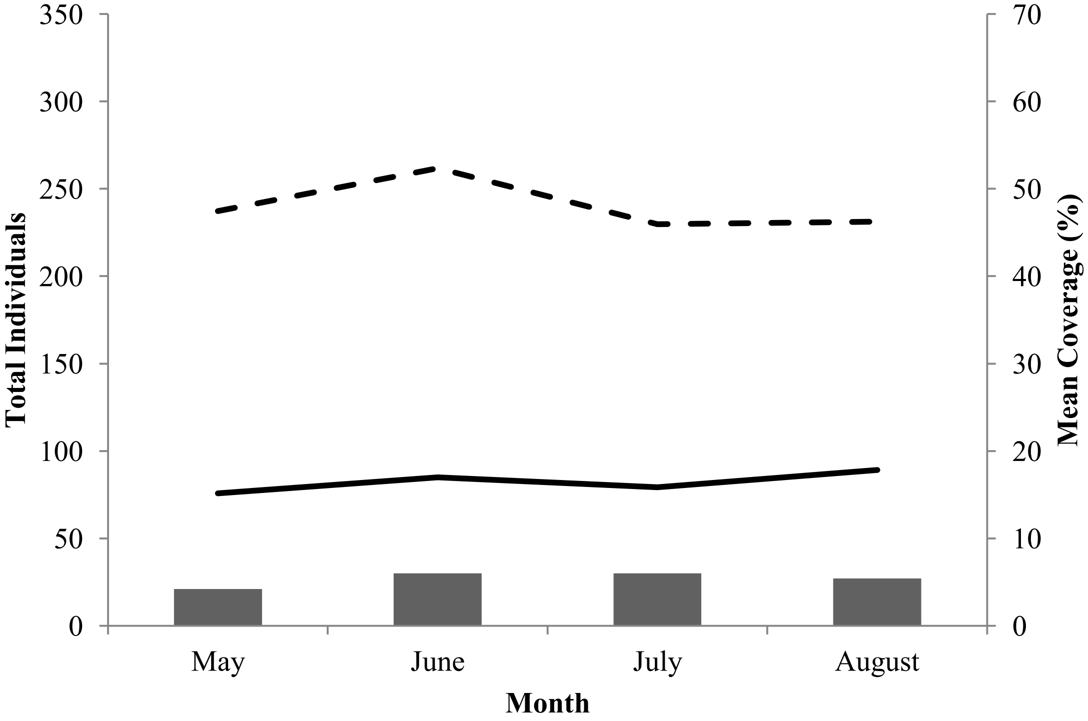


Fig. 13. – Number of individual small mammals captured (bars) each month in the wet meadows in 2013, and the cover (%) of forbs (solid) and grasses (dashed).

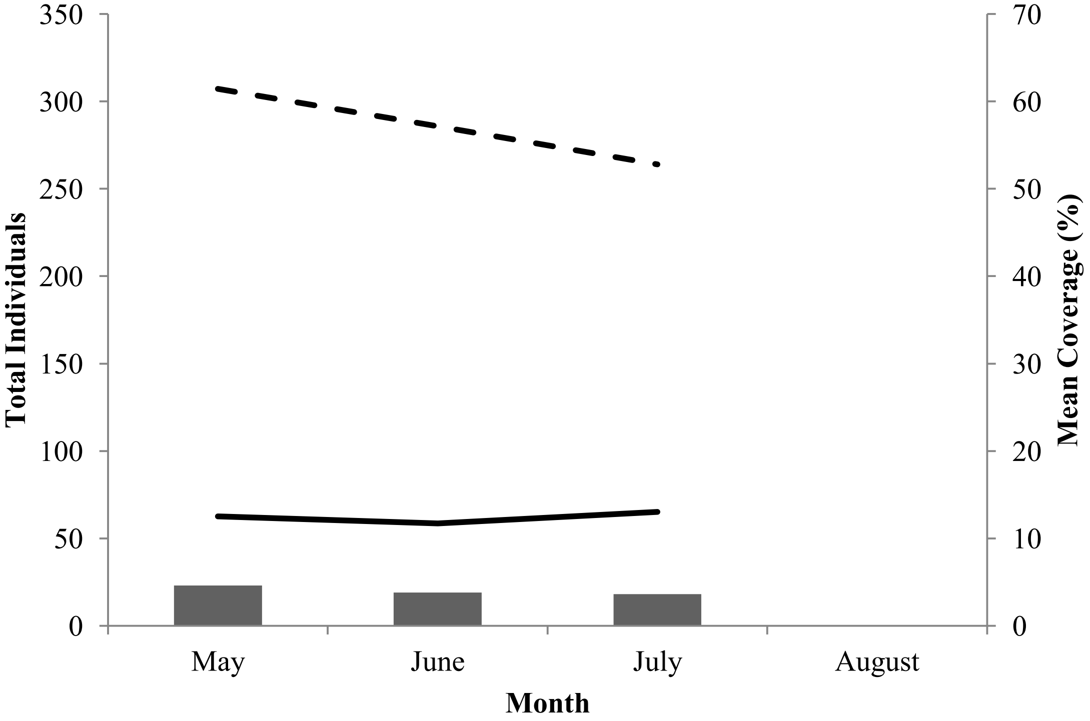


Fig. 14. – Number of individual small mammals by species captured each month in the wetlands in 2012.

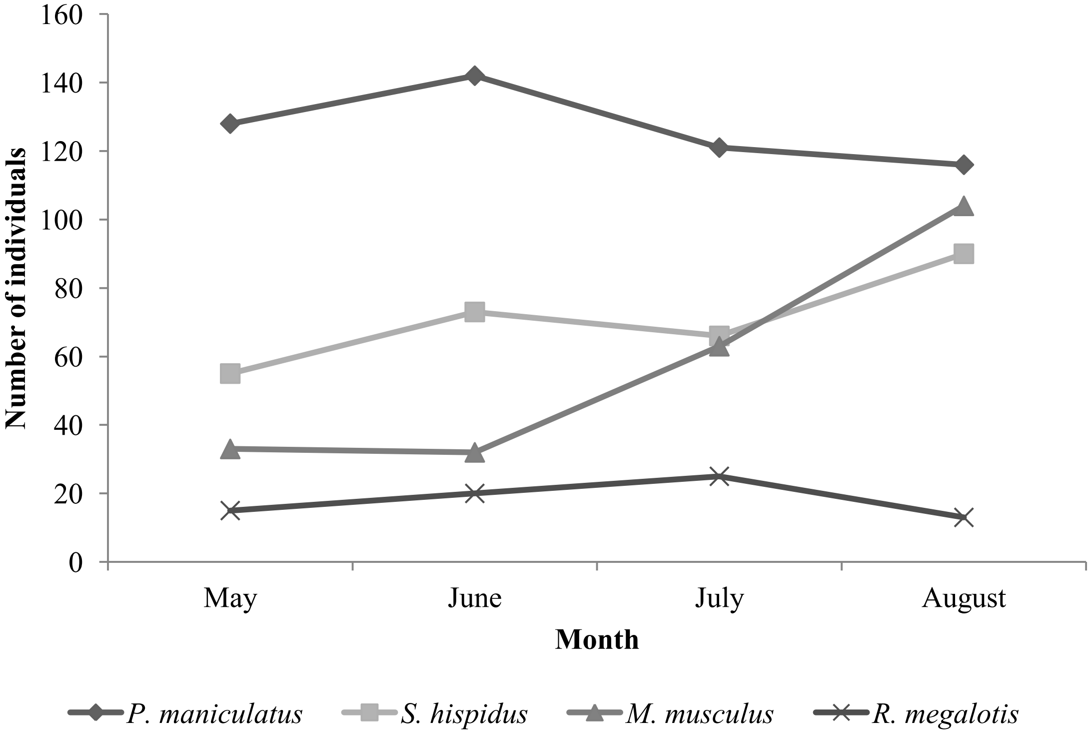


Fig. 15. – Daily high temperatures (°C) for the winter and spring months in 2011–2012 (dark line) and 2012–2013 (light line). The solid trend line was for the 2011–2012 winter and spring data, and the dashed trend line was for the 2012–2013 winter and spring data. Data were from the National Oceanic and Atmospheric Administration's weather station at Quivira National Wildlife Refuge (ID: GHCND:USR0000KSTA) approximately 64.4 km (40 mi) from the Cheyenne Bottoms Wildlife Area (NOAA 2014).

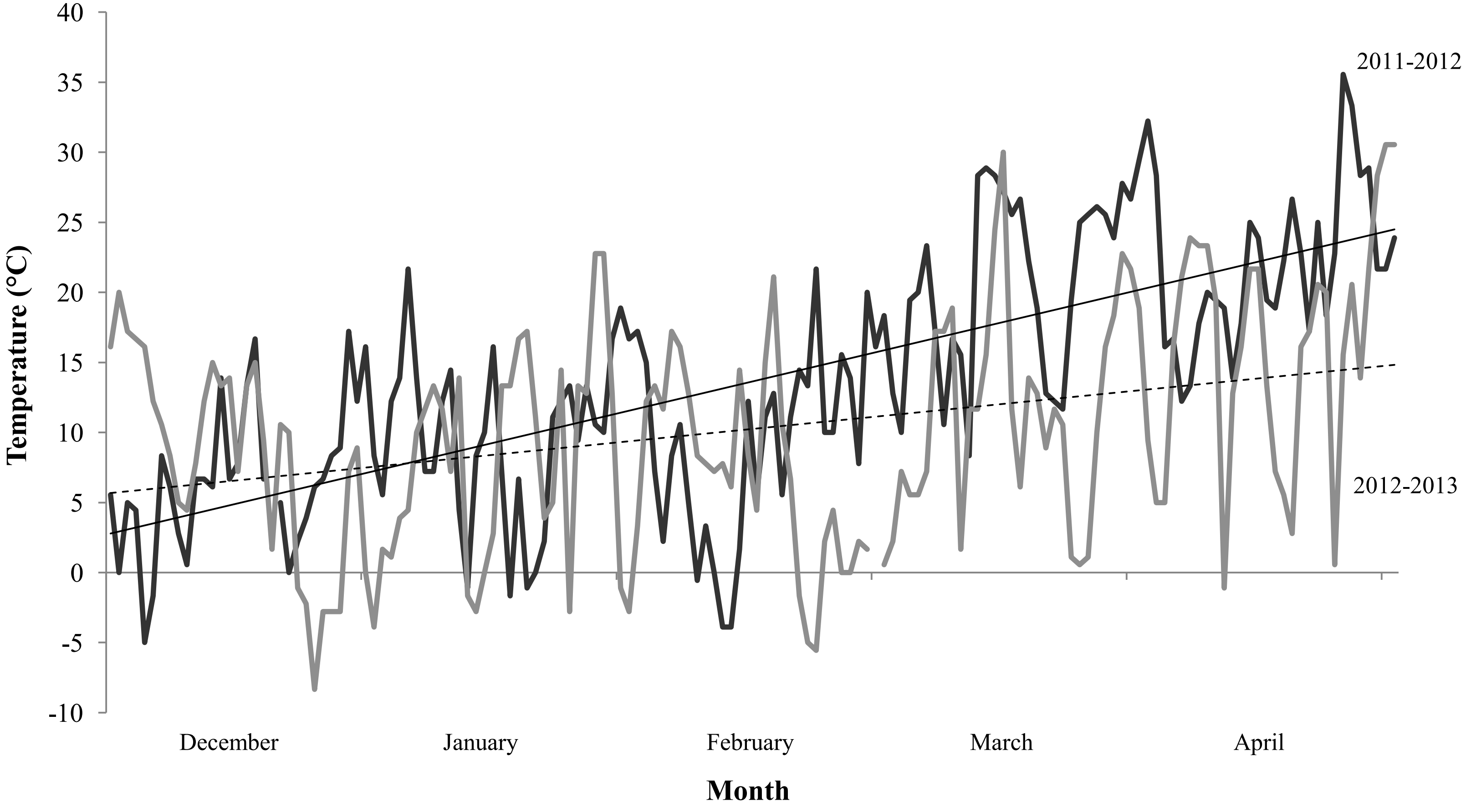
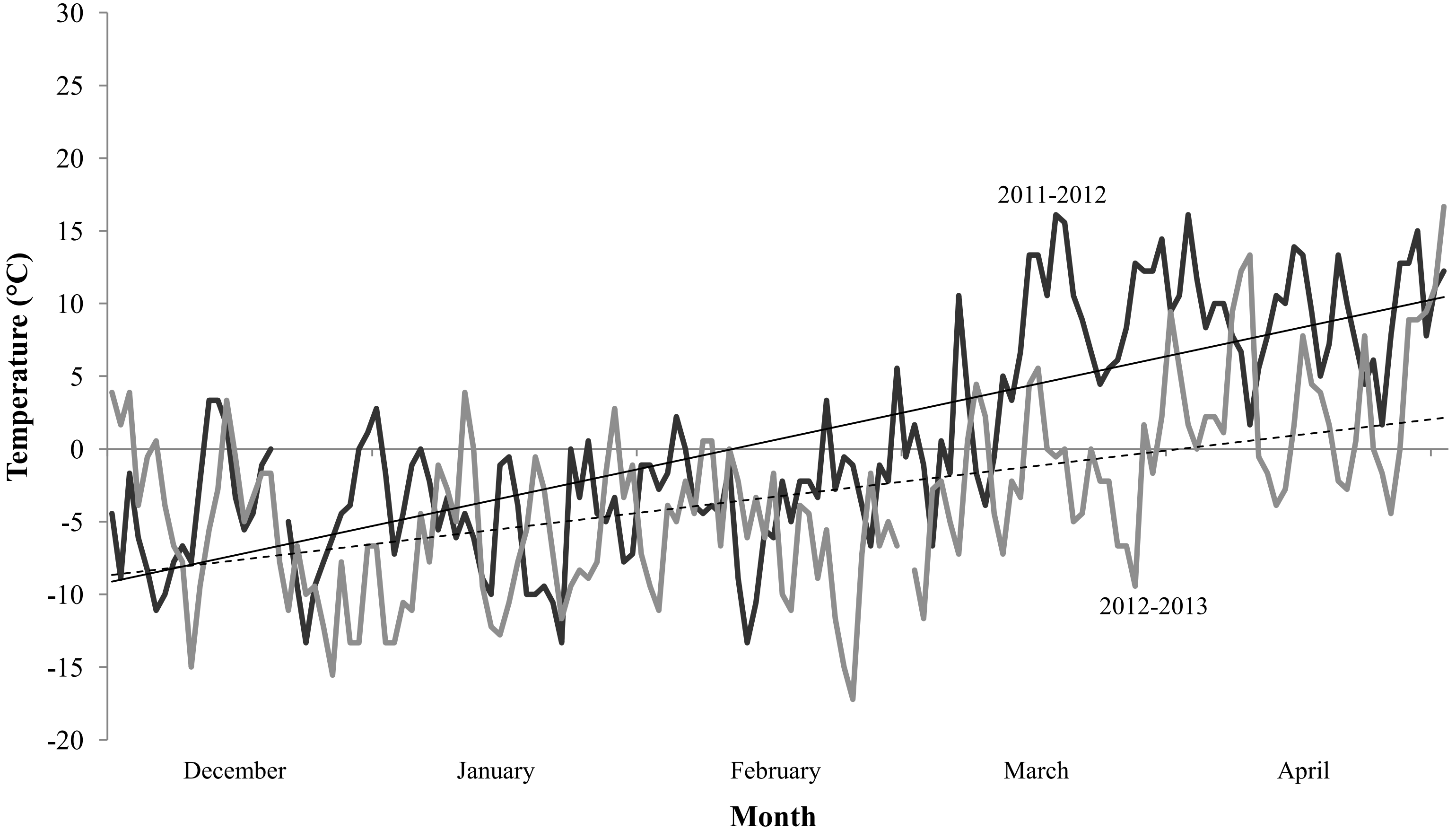
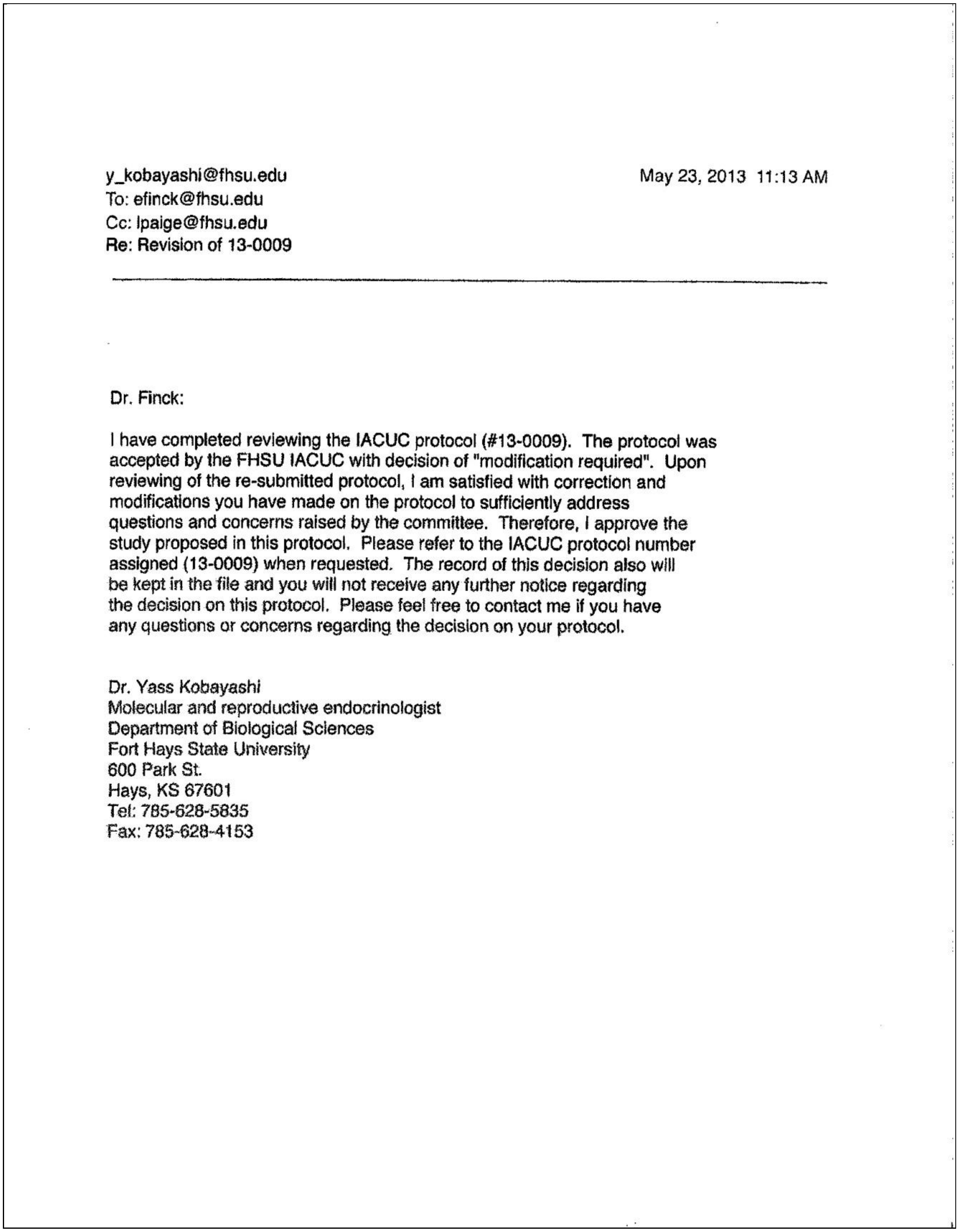


Fig. 16. – Daily low temperatures (°C) for the winter and spring months in 2011–2012 (dark line) and 2012–2013 (light line). The solid trend line was for the 2011–2012 winter and spring data, and the dashed trend line was for the 2012–2013 winter and spring data. Data were from the National Oceanic and Atmospheric Administration's weather station at Quivira National Wildlife Refuge (ID: GHCND:USR0000KSTA) approximately 64.4 km (40 mi) from the Cheyenne Bottoms Wildlife Area (NOAA 2014).



Appendix 1. – Project approval by the Fort Hays State University Institutional Animal Care and Use Committee protocol number 013-0009.



y_kobayashi@fhsu.edu May 23, 2013 11:13 AM
To: efinck@fhsu.edu
Cc: lpaige@fhsu.edu
Re: Revision of 13-0009

Dr. Finck:

I have completed reviewing the IACUC protocol (#13-0009). The protocol was accepted by the FHSU IACUC with decision of "modification required". Upon reviewing of the re-submitted protocol, I am satisfied with correction and modifications you have made on the protocol to sufficiently address questions and concerns raised by the committee. Therefore, I approve the study proposed in this protocol. Please refer to the IACUC protocol number assigned (13-0009) when requested. The record of this decision also will be kept in the file and you will not receive any further notice regarding the decision on this protocol. Please feel free to contact me if you have any questions or concerns regarding the decision on your protocol.

Dr. Yass Kobayashi
Molecular and reproductive endocrinologist
Department of Biological Sciences
Fort Hays State University
600 Park St.
Hays, KS 67601
Tel: 785-628-5835
Fax: 785-628-4153

Appendix 2. – Species of plants found in each habitat type (W = wetlands, M = wet meadows). The cover type that each species was assigned is given (G = grasses, F = forbs, S/R = sedges/rushes).

Common Name*	Scientific Name*	Habitat Type	Cover Type
Annual Marsh Elder	*Iva annua* L.	M	F
Annual Yellow Sweetclover	*Melilotus indicus* (L.) All.	W	F
Aster spp.		M	F
Burningbush	*Bassia scoparia* (L.) A.J. Scott	M, W	F
Bushy Knotweed	*Polygonum ramosissimum* Michx.	W	F
California Loosestrife	*Lythrum californicum* Torr. & A. Gray	M	F
Canadian Horseweed	*Conyza canadensis* (L.) Cronquist	W	F
Common Milkweed	*Asclepias syriaca* L.	M	F
Common Pepperweed	*Lepidium densiflorum* Schrad.	M	F
Common Sunflower	*Helianthus annuus* L.	M, W	F
Cuman Ragweed	*Ambrosia psilostachya* DC.	M, W	F
Curly Dock	*Rumex crispus* L.	M, W	F
Curlycup Gumweed	*Grindelia squarrosa* (Pursh) Dunal	M	F
Curlytop Knotweed	*Polygonum lapathifolium* L.	W	F
Golden Tickseed	*Coreopsis tinctoria* Nutt.	M	F
Illinois Bundleflower	*Desmanthus illinoensis* (Michx.) MacMill. ex B.L. Rob. & Fernald	M, W	F
Lambsquarters	*Chenopodium album* L.	W	F
Little Hogweed	*Portulaca oleracea* L.	W	F
Pale Dock	*Rumex altissimus* Alph. Wood	M	F
Pigweed	*Amaranthus* spp.	W	F
Prickly Lettuce	*Lactuca serriola* L.	W	F
Saltmarsh Aster	*Symphyotrichum* spp.	W	F

Appendix 2. – Continued.

Common Name*	Scientific Name*	Habitat Type	Cover Type
Spanish Gold	*Grindelia papposa* G.L. Nesom & Suh	M	F
Velvetleaf	*Abutilon theophrasti* Medik.	W	F
Western Yarrow	*Achillea millefolium* L. var. occidentalis DC.	M	F
Whorled Milkweed	*Asclepias verticillata* L.	M	F
Alkali Sacaton	*Sporobolus airoides* (Torr.) Torr.	M	G
Big Bluestem	*Andropogon gerardii* Vitman	M	G
Blue Grama	*Bouteloua gracilis* (Willd. ex Kunth) Lag. ex Griffiths	M	G
Buffalograss	*Bouteloua dactyloides* (Nutt.) J.T. Columbus	M	G
Cheatgrass	*Bromus tectorum* L.	M	G
Composite Dropseed	*Sporobolus compositus* (Poir.) Merr.	M	G
Foxtail Barley	*Hordeum jubatum* L.	M, W	G
Indiangrass	*Sorghastrum nutans* (L.) Nash	M	G
Intermediate Wheatgrass	*Thinopyrum intermedium* (Host) Barkworth & D.R. Dewey	M	G
Kentucky Bluegrass	*Poa pratensis* L.	M	G
Little Barley	*Hordeum pusillum* Nutt.	M, W	G
Saltgrass	*Distichlis spicata* (L.) Greene	M, W	G
Sand Dropseed	*Sporobolus cryptandrus* (Torr.) A. Gray	M	G
Sideoats Grama	*Bouteloua curtipendula* (Michx.) Torr.	M	G
Switchgrass	*Panicum virgatum* L.	M	G
Tumble Windmill Grass	*Chloris verticillata* Nutt.	M	G
Tumblegrass	*Schedonnardus paniculatus* (Nutt.) Trel.	M	G
Western Wheatgrass	*Pascopyrum smithii* (Rydb.) Á. Löve	M, W	G

Appendix 2. – Continued.

Common Name*	Scientific Name*	Habitat Type	Cover Type
Witchgrass	*Panicum capillare* L.	M	G
Cattail	*Typha* spp.	W	S/R
Cosmopolitan Bulrush	*Schoenoplectus maritimus* (L.) Lye	W	S/R
Sedge	*Carex* spp.	M	S/R
Softstem Bulrush	*Schoenoplectus tabernaemontani* (C.C. Gmel.) Palla	W	S/R
Spikerush	*Eleocharis* spp.	M	S/R
Dodder	*Cuscuta* spp.	W	N/A

*Notation follows that of the PLANTS Database (USDA and NRCS 2014).