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Northeast Region
Boston, Massachusetts

Inventory of Small Mammals at Cape Cod National Seashore with Recommendations for Long-Term Monitoring

Technical Report NPS/NER/NRTR--2006/047

ON THE COVER
White footed mouse; *Peromyscus leucopus*
Photograph by: Kelly Boland

Inventory of Small Mammals at Cape Cod National Seashore with Recommendations for Long-Term Monitoring

Technical Report NPS/NER/NRTR--2006/047

Robert P. Cook, Kelly M. Boland, and Tressa Dolbeare

Cape Cod National Seashore
Wellfleet, MA 02661

July 2006

U.S. Department of the Interior
National Park Service
Northeast Region
Boston, Massachusetts

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Table of Contents

Tables vi

Figures viii

Appendices xi

Acknowledgements xii

Part One: 1

Background and Justification for the Small Mammal Monitoring Protocol 1

Introduction 1

Inventory and Monitoring Questions 3

Materials and Methods 5

Field Procedures 5

Estimating Abundance 8

Estimating Survival 8

Variation in Abundance 9

Power Analysis 9

Species Diversity 10

Habitat Variables 10

Habitat Relationships 11

Results 13

Raw Captures 13

Estimating Abundance 13

Estimating Survival 18

Variation in Abundance 21

Power Analysis 30
Community Structure 39
Habitat Relationships 45
Discussion 49
Estimating Abundance 49
Estimating Survival 51
Variation in Abundance 51
Power Analysis 76
Community Analysis 79
Adequacy of Species Detection 83
Conclusions/Recommendations 85
CACO Mammals 85
Estimation of Abundance 85
Estimation of Survival 86
Monitoring Program 86
Part Two: 89
Small Mammal Monitoring Protocol 89
Introduction 89
Safety Considerations 89
Field Methods 90
Data Entry 98
Data Analysis 98
Vegetation Sampling 101

Use of Vegetation Data in Habitat Analysis 106

Literature Cited 107

Tables

Table 1. Names and location of small mammal sampling sites. Coordinates are for southwest corner of sampling site. All sites are oriented N-S, E-W. 6

Table 2. Description of habitat variables measured at each small mammal sampling site..... 12

Table 3. Summary of raw captures (#inds, #capture events), percent composition of individuals captured, and catch per 100 trap nights, by species, for 2000 and 2001. All sites and sampling periods combined. 14

Table 4. Species composition of individuals captured, by habitat, based on all individuals captured in 2000 and 2001. 15

Table 5. Correlation between abundance measures based on counts of individuals and capture events, an index (individuals/100 trap nights), and numerical estimators of population size derived from program CAPTURE (N_{CAP}) and the modified Lincoln-Petersen method (N_{LP}). Marked correlations (*) are significant at $p < .05$. n=133. 17

Table 6. Estimated monthly survival of small mammals over the course of the trapping season, late spring through late summer. Program RELEASE goodness of fit tests indicate data were insufficient. 20

Table 7. Variation in estimated abundance due to year, site, and habitat. Analysis of variance results (values of p) for three way ANOVA, with sites nested in habitat. Significant values $p < .05$ are in bold. 22

Table 8. Mean estimated abundance (N_{LP}) by year, for each of 11 species at Cape Cod National Seashore. Bold indicates species where means are significantly different ($p < .05$). 22

Table 9. Variation in estimated abundance (ANOVA) for each species/group due to season, habitat and site (nested in habitat). Significant values ($p<.05$) are in bold. 25

Table 10. Mean estimated abundance, by habitat type, for each of 11 species at Cape Cod National Seashore. Bold indicates species where habitat means are significantly different at $p<0.05$. Means with same subscript belong to same homogeneous group. 25

Table 11. Mean estimated abundance by sampling period/season, for each of 11 species at Cape Cod National Seashore. Bold indicates species where means are significantly different. Means with same subscript belong to same homogeneous group. 28

Table 12. Mean estimated abundance, by site, for each of 11 species at Cape Cod National Seashore. Means with same subscript belong to same homogeneous group. 29

Table 13. Power to detect park-wide changes in estimated total abundance. 31

Table 14. Power to detect change in estimated total abundance at least and most variable sites based on four temporal replicates per sampling season. 32

Table 15. Power to detect change in estimated total abundance at least and most variable sites, based on replicate four only. 33

Table 16. Power to detect park-wide changes in estimated PELE abundance. 35

Table 17. Power to detect change in estimated PELE abundance least and most variable sites, based on four temporal replicates per sampling season. 36

Table 18. Power to detect change in estimated PELE abundance at least and most variable site, based on replicate four only. 37

Table 19. Power to detect change in estimated ZAHU abundance, at 6 sites for both all four replicates and only replicate four. 38

Table 20. Similarity of small mammal community at sample sites. Bray-Curtis similarity matrix based on mean estimated abundance of each species at each site. Bold indicates sites with similar mammal communities, determined by two-way ANOSIM (sites x season). 41

Table 21. Measures of species diversity (H'(log e)), evenness (J'), number of species recorded (S), and mean estimated total abundance for sample sites and by habitat. H' and J' were calculated using estimated abundance..... 44

Table 22. Eigenvalues and eigenvectors of significant habitat principal components. 46

Table 23. Results of forward stepwise linear regression of small mammal estimated abundance on habitat principal component scores. Significant relationships (P<0.05) are shown in bold. 48

Table 24. Comparison of small mammal capture rates at CACO with other, similar studies. Abundance method refers to manner in which abundance was expressed or calculated. CHIS=Channel Islands National Park, CA. DENA=Denali National Park, AK. 50

Table 25. Coefficients of variation in estimated abundance based on all 4 replicates and Replicate 4 only. ***denotes locations where species was not captured..... 78

Table 26. Small mammal field data sheet. 94

Table 27. Sky codes. 95

Table 28. Overstory, understory, and tree sampling datasheet. In each plot, estimate cover clases to nearest 5%. 104

Table 29. Groundcover vegetation sampling datasheet. 105

Figures

Figure1. Small mammal trapping sites at Cape Cod National Seashore. 7

Figure 2. Frequency distribution of total number of individuals of a species captured alive during a weekly sampling session. Only instances where at least one live individual was captured are included. 15

Figure 3. Frequency distribution of total number of individuals of a species captured at a site for an entire season. 19

Figure 4. Mean annual estimated total abundance by site. There were no significant site-specific, inter-year differences..... 23

Figure 5. Variation in overall mean estimated abundance by season. For each species or species group, estimated abundance is mean of all sites for 2000 and 2001. 26

Figure 6. Variation in park-wide estimated abundance of each species or species group over the course of sampling. Data are means of all sites. 26

Figure 7. Ordination of small mammal communities at each sampling site. Based on mean estimated abundance of each species at each site. 40

Figure 8. Ordination of small mammal communities by habitat type. Based on mean estimated abundance of each species in each site habitat type. 42

Figure 9. Ordination of sites by Principal Components Analysis based on habitat data. 47

Figure 10. Mean annual estimated abundance of PELE by site. Site-specific, inter-year comparisons found no significant differences. 54

Figure 11. Mean annual estimated abundance of MIPE by site. There were significant inter-year differences at sites Cemetary Oak (CMO) (p=0.01) and Pilgrim Heights Grassland (PHG) (p=0.02). 55

Figure 12. Mean annual estimated abundance of SOCI by site. There were no site-specific, inter-year differences. 56

Figure 13. Mean annual estimated abundance of CLGA by site. There were significant inter-year differences at Cemetary Oak (CMO) (p=0.0007). 57

Figure 14. Mean annual estimated abundance of ZAHU by site. There were no significant site-specific inter-year differences. 58

Figure 15. Mean annual estimated abundance of BLBR by site. There were no significant site-specific inter-year differences. 59

Figure 16. Mean annual estimated abundance of GLVO by site. There were no significant site-specific inter-year differences. 60

Figure 17. Temporal variation in estimated abundance of each species at Bound Brook Heathland over the course of sampling. 62

Figure 18. Temporal variation in estimated abundance of each species at Bound Brook Pine Forest over the course of sampling. 62

Figure 19. Temporal variation in estimated abundance of each species at Cemetery Oak Forest over the course of sampling. 63

Figure 20. Temporal variation in estimated abundance of each species at Duck Harbor Wetland over the course of sampling. 63

Figure 21. Temporal variation in estimated abundance of each species at Fort Hill Grassland over the course of sampling..... 64

Figure 22. Temporal variation in estimated abundance of each species at High Toss Wetland over the course of sampling. 64

Figure 23. Temporal variation in estimated abundance of each species group at Longnook Oak Forest over the course of sampling. 65

Figure 24. Temporal variation in estimated abundance of each species at Longnook Pine Forest over the course of sampling. 65

Figure 25. Temporal variation in estimated abundance of each species at Marconi Heathland over the course of sampling..... 66

Figure 26. Temporal variation in estimated abundance of each species at Pilgrim Heights Grassland over the course of sampling. 66

Figure 27. Estimated total abundance over time, by site, over all sampling periods..... 67

Figure 28. Estimated abundance over time, by site, over all sampling periods for *Peromyscus leucopus*..... 67

Figure 29. Estimated abundance over time, for all sites, over all sampling periods for *Microtus pennsylvanicus*. 68

Figure 30. Estimated abundance over time, by site, over all sampling periods for *Clethrionomys gapperi*. 68

Figure 31. Estimated abundance over time, by site, over all sampling periods for *Zapus hudsonius*..... 69

Figure 32. Estimated abundance over time, for all sites, over all sampling periods for *Sorex cinereus*. 69

Figure 33. Estimated abundance over time, by site, over all sampling periods for *Blarina brevicauda*. 70

Figure 34. Total abundance of sites relative to their position in habitat space, as determined by principal components analysis of site habitat data. Site total abundance is expressed by size of circle. 74

Figure 35. Species diversity at sites relative to their location in habitat space as determined by principal components analysis. Site diversity is expressed by size of circle. 82

Figure 36. Mammal trapping grid layout. Shaded areas represent vegetation sampling plots. .. 91

Figure 37. Percent cover visual aid. 102

Appendices

Appendix 1. Values of mammal habitat variables used in Principal Components Analysis. Basal area is expressed as cm^2 and represents the sum of basal areas for the trees of interest recorded in the five vegetation sampling plots within a site's trapping grid. 114

Appendix 2. Population estimates for all species present (total abundance), total rodents, total shrews, and each individual species, for all sites and trapping sessions. 115

Appendix 3. List of equipment and materials necessary for a single replicate season of small mammal sampling. 117

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Part One: Background and Justification for the Small Mammal Monitoring Protocol

Introduction

Small mammals are an important component of Cape Cod National Seashore's fauna. In addition to their direct contribution to species richness, they play a major role in trophic dynamics, consuming plant material and invertebrates, and in turn serving as prey items for snakes, raptorial birds, and small to mid-sized carnivorous mammals. Through these relationships, small mammals may directly influence population levels of insect pests and disease vectors such as gypsy moths and deer ticks, as well as certain regionally rare hawks and owls, and have a cascading effect up and down the "food chain". Moreover, the abundance and composition of small mammal communities can affect the structure, species composition, and successional trends of plant communities (Ostfeld 2002).

At CACO, red fox (*Vulpes vulpes*) and other carnivores prey upon nests of colonial waterbirds and shorebirds such as the federally threatened piping plover. Since small mammals serve as a food source for these predators, variation in their abundance may affect predation pressure on these birds (Bennett 1998). Small mammal abundance and community structure at CACO is influenced by agents of change such as fire suppression, exotic species introduction, habitat succession, weather, and mast production (Wolff 1996). Some of the mechanisms by which these agents of change affect small mammal abundance include loss of herbaceous dominated habitat from suppression of fire, human caused increases in predators such as skunks and red foxes ("subsidized predators"), and declines in native forage material through replacement by exotic plants. In addition, small mammals also act as agents of change, influencing abundance of predators and acting to control outbreaks of pest species such as gypsy moth (Elkinton et al. 1996, Ostfeld 1996). Since small mammal species vary in their preferences for tree seedlings, their abundance and species composition also influences the rate and species composition in old field succession (Ostfeld et al. 1997).

Because of their ecological importance, and the potential for their populations to respond to numerous "agents of change" operating in and adjacent to CCNS (as well as become "agents of change" themselves) small mammals have been identified for possible monitoring as part of Cape Cod National Seashore's Long Term Monitoring Program (Roman and Barrett 1999). To facilitate this, a monitoring protocol was developed (Bennett 1998), based largely on protocols from Denali National Park (Rexstad 1996). Similar protocols are employed at Channel Islands National Park (Fellers et al.1988) and at Organ Pipe Cactus National Monument (Petryszyn undated). This protocol was implemented in 2000 and 2001 for two purposes. First, to provide Cape Cod National Seashore with its first-ever quantitative inventory of small mammals and analysis of their habitat relationships. Second, to evaluate the utility of this protocol and the feasibility of its use for long term monitoring. In particular, since the protocol attempts to estimate several population parameters (e.g. abundance, survival, recruitment), and there are generally a number of methods and models for doing so, an important goal was to identify which parameters could realistically be estimated, and determine optimal field and analytical methods.

Inventory and Monitoring Questions

While there was general knowledge of species occurrence at CACO (Prescott 1994, Spitzer 1976), the small mammals of CACO had never been systematically inventoried. Thus, the inventory questions were;

1. What is the distribution, abundance, and species composition of small mammal communities in oak and pine forest, wetland, grassland and heathland habitats at CACO?
2. To what extent do small mammal communities differ between these habitat types?
3. What habitat features influence the abundance and structure of small mammal communities at CACO?
4. To what extent are there seasonal differences in the abundance of small mammals at CACO?.

Since the small mammals known to occur at CACO are both habitat generalists and specialists, it is anticipated that successional changes occurring across CACO's landscape will bring about changes in species abundance and community structure. Thus, the primary monitoring questions regarding small mammals at CACO relate to trends in small mammal abundance and community structure over the long term and the relationship between these trends and habitat change. Specific monitoring questions are;

1. What are the temporal trends in the occurrence, distribution, abundance and composition of small mammal communities at CACO?

Expressed as specific hypotheses, these would be:
1a. The long term abundance of small mammals (in total and for individual species) at CACO shows no significant trend.
1b. The frequency of occurrence or site occupancy rate of individual species of small mammals does not change over the long term.
1c. The composition of small mammal communities does not change over time on a park-wide or site specific level.

2. Are there long term changes in environmental variables such as plant community structure and composition, rainfall, and soil moisture? To what extent are these changes the result of fire suppression, invasive alien species, or adjacent development?

3. Are long term changes in small mammal abundance, occurrence, and species composition correlated with long term changes in these environmental variables?

While these are the fundamental questions that potentially could be addressed through monitoring, should the need arise, small mammal monitoring could also be coupled with monitoring of other environmental variables (e.g. predator or prey species, food sources, etc.) to interpret trends in these relationships.

Materials and Methods

Field Procedures

Testing of a small mammal monitoring protocol proposed for Cape Cod National Seashore (Bennett 1998) was conducted in 2000 and 2001. Sampling was conducted in the major upland and lowland habitat types present in the park, heathland, wetland, grassland, oak forest, and pine forest. For each habitat type, 1990 vegetation maps were used to identify potential sampling sites, i.e. polygons of a habitat type large enough to overlay a grid containing a least five one ha square plots. Two sites per habitat type were then selected randomly, for a total of 10 sample sites (Figure 1, Table 1). For each site selected, a specific trapping plot was then randomly selected from the grid overlay. However, trapping plots within 20 m of any road (paved or unpaved) or containing a trail were passed over in the selection process.

At each site, a trapping grid was established. One hundred (100) Sherman live traps (model LFATGD, H.B. Sherman Traps, Tallahassee, FL) were deployed at 10 m intervals in a 90 m X 90 m (0.81 ha) north-oriented square grid. This grid size, shape, and trap spacing was chosen based on small mammal densities previously reported from Cape Cod (Adler 1988) and the recommendations of Rexstad (1996), who found that trapping webs violated assumptions of high detection probability at their center. Traps were baited with a mixture of peanut butter, rolled oats, and cooked bacon. In the initial replicate (late spring, 2000), traps were set Monday morning, checked that afternoon, and every morning and afternoon thereafter until removed on Thursday afternoon. This resulted in four afternoon trap checks and three morning trap checks. Because 89% of the 228 capture events recorded during this initial replicate were during the morning trap check, the trapping schedule was altered. Thereafter, traps were set Monday, removed Friday, and checked each morning, providing four sampling occasions. While checking traps once rather than twice daily can be a factor in higher rates of trap mortality, results from 2000 indicate there were no significant differences due to this factor. Trap mortality of masked shrew (*Sorex cinereus*) was 78% during replicate one, when traps were checked twice/daily and 72% during the remaining replicates (χ^2 =0.02, df=1, p=0.88). For all rodents, trap mortality was 1.8% in replicate one and 2.8% during the remaining replicates (χ^2 =0.68, df=1, p=0.41).

Table 1. Names and location of small mammal sampling sites. Coordinates are for southwest corner of sampling site. All sites are oriented N-S, E-W.

Site Code	Location	Habitat Type	Easting	Northing
FHG	Fort Hill Grassland	Grassland	419986	4630214
PHG	Pilgrim Heights Grassland	Grassland	407897	4656317
MCH	Marconi Heathland	Heathland	419051	4639483
BBH	Bound Brook Heathland	Heathland	411522	4645841
CMO	Cemetery Oak	Oak Forest	412683	4647798
LNO	Longnook Oak	Oak Forest	413329	4652233
BBP	Bound Brook Pine	Pine Forest	412047	4645286
LNP	Longnook Pine	Pine Forest	412744	4652876
HTW	High Toss Wetland	Wetland	412562	4644051
DHW	Duck Harbor Wetland	Wetland	410817	4644460

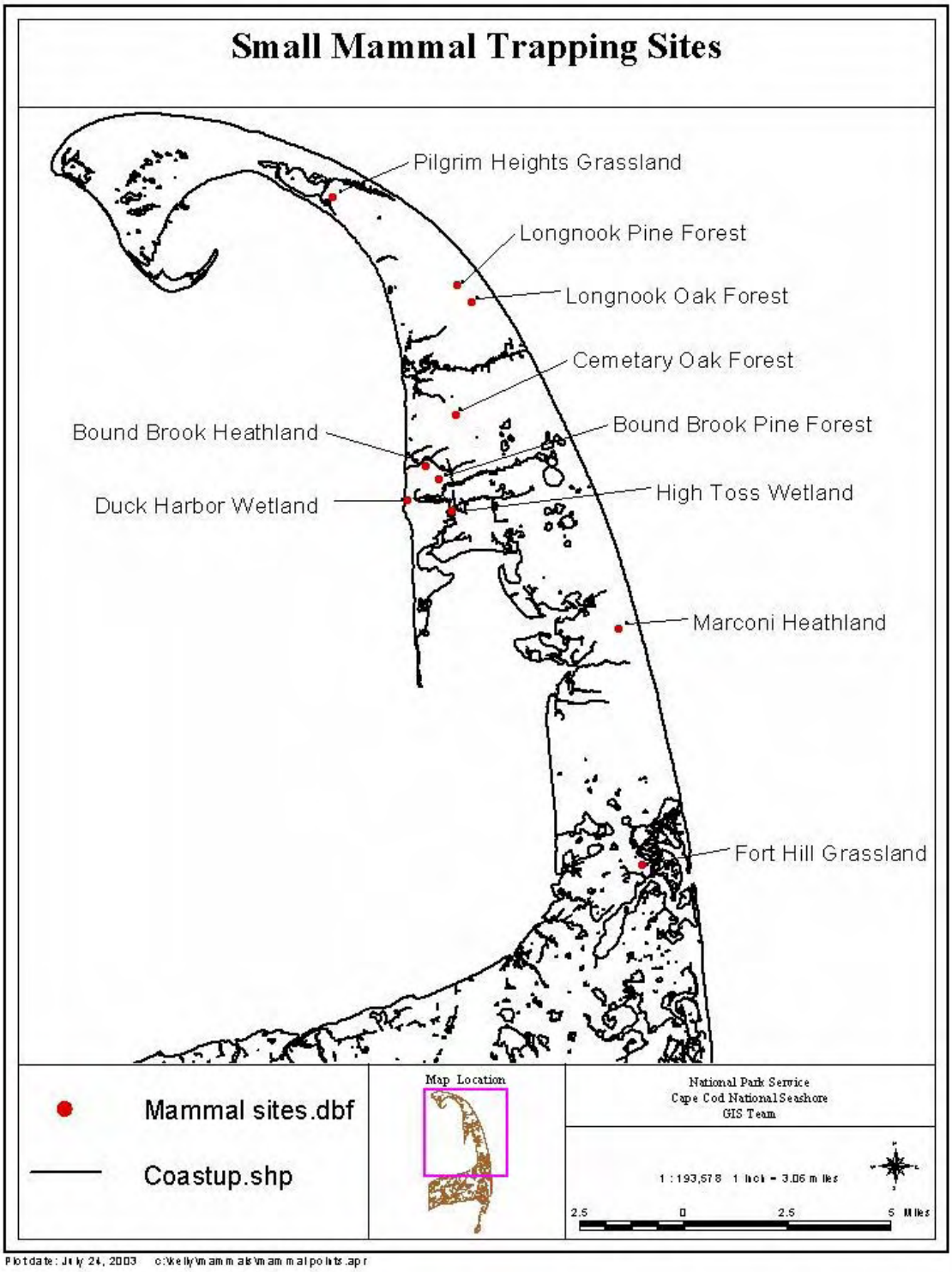


Figure1. Small mammal trapping sites at Cape Cod National Seashore.

While this weekly sampling session duration fit in well operationally with a traditional five-day work week, its use was also based on findings that assumptions of population closure (inherent in closed population estimation models) in small mammals do not extend well beyond a four day trapping session (Rexstad 1996). Two sites (never the same habitat) were trapped each week, with 5 weeks required to complete a replicate of sampling at each of the 10 sites.

To determine the extent of seasonal changes in small mammal abundance and estimate survival and recruitment rates over the course of the growing season using the robust design (Pollock et al. 1990), a total of four seasonal replicates of sampling were conducted (replicate 1=late spring, replicate 2=early summer, replicate 3 = mid summer, and replicate 4=late summer). Outside dates of trapping were 15 May to 29 September 2000, and 14 May to 28 September 2001. Trapping effort in 2000 was 1,500 trap nights/site (300 in replicate one and 400/replicate thereafter), for a grand total of 15,000 trap nights. In 2001, trapping effort was 1,600 trap nights/site (400/replicate), for a grand total of 16,000 trap nights. All animals captured were identified to species, weighed, measured, sexed, aged, marked with a Passive Integrated Transponder (PIT tag) and then released at point of capture.

Estimating Abundance

Population size (N) of each species, for a given site and weekly trapping session was estimated using the multiple mark and recapture methods of program CAPTURE (Otis et al. 1978, Rexstad and Burnham 1991). Estimates of N using CAPTURE are based on the model selected as best fit by the program. Whenever capture-recapture data were inadequate to estimate N, we used the number of individuals captured as the estimate of abundance (Rexstad 1999). For shrews (*Sorex* and *Blarina*) which rarely survive to be recaptured, the generalized removal estimator (M_{bh}; White et al. 1982) was used (Rexstad 2001). Where this failed we used total capture events.

Since numbers of live individuals captured usually were far fewer than the "several times larger than 10 or 20" needed for CAPTURE to produce satisfactory results (White et al. 1982), N was also estimated using Chapman's modified Lincoln-Petersen method (Nichols and Dickman 1996). Similar to Ellison and Van Riper (1998) we condensed trapping data from four daily trapping occasions into two, because Lincoln-Petersen uses only two trapping periods. Shrews, however, were not estimable using Lincoln-Petersen due to rarity of recapture.

To provide for comparison with numeric estimators of N, we also generated indices of abundance based on total individuals captured (M_{t+1}), total capture events (n .), and catch per 100 trap nights, corrected for disturbed traps.

Estimating Survival

Survival rates for a given species at a given site were estimated using program MARK (White and Burnham 1999) using the "Recaptures Only" option, which estimates survival rates using the standard Cormack-Jolly-Seber Model. Survival rates over the course of the trapping season (late spring thru late summer) were calculated based on capture history data that had been configured in accordance with the "robust design" (Pollock et al. 1990). In this, each week-long replicate is considered a major or primary sampling occasion, and each day within that week a secondary or

minor occasion. For the purposes of survival estimation over the trapping season, all the secondary occasions within a primary occasion are compressed into a single value indicating whether or not an individual was captured during a given primary session. MARK was used to evaluate models of constant and time dependant survival (Phi) and capture probability (p) and determine, based on the Akaike Information Criterion (AIC), which model best fit the data. Since applicability of reduced parameter models (e.g. constant survival and/or capture probability) depends on the goodness of fit of generalized (i.e. time specific) models (Cooch and White 2001), goodness of fit of generalized models (i. e. time specific Phi and p) was tested using MARK's Program RELEASE goodness of fit test.

Variation in Abundance

Analysis of variation in estimated abundance of individual species and species' groups used the modified Lincoln-Petersen abundance estimator for rodent species and the generalized removal model for shrew species. When numbers of captures were insufficient for these methods to work, we used weekly trapping session total number of unique individuals for rodents and total number of capture events for shrews. Shapiro-Wilk's test revealed that population estimation data were not normally distributed. Consequently, all abundance data were log transformed (Ln (X+1)) (Zar 1996) prior to analysis.

Variation in estimated abundance due to year, seasons, habitats, and sites was analyzed with Analysis of Variance (ANOVA). While these are four sources of variation, it was not possible to conduct an analysis of all four factors at once. Analysis was based on sites being nested within habitats, and, since seasonal estimates of abundance represented repeated measurements on the same trapping grid, seasonal variation was analyzed as a repeated measurements ANOVA . Analysis of annual variation was based on a three factor ANOVA consisting of factors Year, Habitat, and Site nested in Habitat. Analysis of variation due to the remaining three factors was based on a repeated measures design, with Habitat, Sites nested in Habitat, and Seasons as repeated measurements at each site.

Power Analysis

Program MONITOR (Gibbs 1995) was used to conduct power analysis to determine the power to detect changes in abundance of differing magnitudes for long term monitoring scenarios of varying spatial and temporal replication. The power to monitor changes in abundance was estimated for monitoring over a 20-25 year period, based on sampling every 1, 2 ,3, and 5 years. For total abundance (all species summed), as well as for some individual species (PELE) means and standard deviations were calculated for each of the ten sites, for monitoring scenarios involving four temporal replicates/year and only one/year, conducted in late summer (Replicate 4). Similarly, this was done for ZAHU, based on the four sites in the grassland and wetland habitats this species occurred in. Since these simulations only estimated the power of statements regarding overall park-wide trends, power was also estimated for individual sites, using data from the most and least variable sites. Since abundance data were lognormally distributed, trend type was set to exponential (Gibbs 1995). Other input settings were as follows: a=0.05; two tails (to detect both increase and decrease); constant added in log transformation =1; trend variation=0 (Gibbs pers. comm.); rounding=integers; and replications =500.

Community Structure

Mammal community structure (i.e. species composition) was examined in a number of ways. Multi-dimensional scaling (MDS) was used to ordinate the relationship of species composition among the ten sample sites in two dimensional space. Mean estimated abundance of each species at each site was subjected to a fourth root transformation to balance contributions of abundant versus rare species (Clarke and Warwick 2001), and to generate the Bray-Curtis similarity matrix in program PRIMER used for MDS. Analysis of Similarity (ANOSIM) was used to analyze variation in community structure. Temporal variability (year, season) was tested as a two-way crossed ANOSIM and spatial variability (habitat, sites) was tested as a two-way nested design, with sites nested in habitats. Data used in ANOSIM were fourth root transformed abundance estimates for each species at each site-sampling occasion. Correlation between the pattern of site similarity based on site mammal communities and the pattern of site similarity based on site habitat variables was analyzed using the RELATE routine of program PRIMER.

Species Diversity

Species diversity by sampling sites and habitat types was evaluated using the Shannon diversity index (H'$\log_e$) and Pielou'evenness index (J'), based on estimated numbers of individuals of each species present.

Habitat Variables

Habitat structure and composition for each of the 10 sampling sites was quantified based on data collected in September 2000. Five 10 m x 10 m vegetation plots within each site's trapping grid were randomly selected to provide an overall characterization. Each plot was sampled in three layers. "Overstory" (>5 m), and "understory" (1-5 m) were sampled from the entire 10m x 10 m plot, whereas "ground" (1m<) was sampled with a 1m x 1 m quadrat, placed in the southwest corner of the plot. At each plot, all trees (dbh>3 cm) were identified to species and dbh measured. In the overstory and understory layer, percent cover for individual species, and for structural/taxonomic groups (e.g. deciduous, coniferous) was assigned a cover class (i.e. 0-1%. 2-5%, 6-25%, 26-50, 51-75%, 76-95%, or 96-100%). In the ground layer, cover classes were assigned to individual species occurring within the quadrat, but not to structural/taxonomic groups (e.g. herbaceous, graminoid, forbs, woody shrub, etc.)

Based on further review of the literature (e.g. Adler et al. 1999, Bellows et al. 2001, Geier and Best 1980, Morrison and Anthony 1989) it was recognized that percent cover or cover classes should have been estimated for all ground layer structural/taxonomic groups. Since it is not feasible to derive this from individual species cover class data, a proxy value was used. This was obtained by counting, for each plot, the number of species of a particular structural/taxonomic group that occurred at a cover class of three (6-25%) or more. The sum of these counts over a site's five plots was used as an index of the abundance of each structural/taxomonic group at a given site.

Habitat Relationships

Principal components analysis (PCA) was performed to determine the relationship between sampling sites based on their habitat variables (Table 2), as well as reduce the number of habitat variables (Kelt et al. 1994, Morrison et. al 1998). Habitat variable data (Appendix Table 1) were tested for normality using the Shapiro-Wilks test of program STATISTICA (Statsoft 2000). Those not meeting assumptions of normality were square root transformed (Kelt et al.1994) following the transformation procedures detailed in Zar (1996), with the exception of percentages, which were arcsine tranformed. PCA was performed using program PRIMER with the data matrix normalized to provide a correlation-based PCA (Clarke and Gorley 2001). Principal components with eigenvalues greater than 1.0 were retained for use in stepwise multiple regression of the relationship between species abundance and habitat principal components. The mean of abundance estimates of a species at a given site was regressed against principal component factor scores for each of the 10 sites (Adler 1988, Stevens et al. 2002).

Table 2. Description of habitat variables measured at each small mammal sampling site.

Variable	Description
Ov Tree Stem	the total number of trees counted in the overstory of the plots sampled within a trapping grid
BA Total	the total basal area of trees in the overstory of the plots sampled within a trapping grid
MeanBA/Tree	the mean of the basal areas of all sampled trees (BA total/Ov Tree Stem)
BA Dead	the total basal area of dead trees in the overstory of the plots sampled within a trapping grid
BA Live	the total basal area of live trees in the overstory of the plots sampled within a trapping grid
BA Decid	the total basal area of deciduous trees in the overstory of the plots sampled within a trapping grid
BA Conif	the total basal area of coniferous trees in the overstory of the plots sampled within a trapping grid
UndTot%Cov	the mean of the midpoints of the categories for all 5 subplots within a plot for understory percent cover
Und#Woody>=3	within each site's understory layer, the number of times that woody vegetation was given a cover class value of a 3 or higher
Und#Conif>=3	within each site's understory layer, the number of times that a conifer was given a cover class value of a 3 or higher
Und#Herb>=3	within each site's understory layer, the number of times that a forb or graminiod was given a cover class value of a 3 or higher
Und#Gram>=3	within each site's understory layer, the number of times that a graminoid was given a cover class value of a 3 or higher
Und#Forb>=3	within each site's understory layer, the number of times that a forb was given a cover class value of a 3 or higher
GrndHerb>=3	within each site's ground layer, the number of times that a forb or a graminiod was given a cover class value of a 3 or higher
GrndGram>=3	within each site's ground layer, the number of times that a graminiod was given a cover class value of a 3 or higher
GrndForb>=3	within each site's ground layer, the number of times that a forb was given a cover class value of a 3 or higher
GrndWshrb>=3	within each site's ground layer, the number of times that a woody shrub was given a cover class value of a 3 or higher
GrndWgrnd>=3	within each site's ground layer, the number of times that woody ground cover was given a cover class value of a 3 or higher
GrndFern>=3	within each site's ground layer, the number of times that a woody shrub was given a cover class value of a 3 or higher
GrndMoss>=3	within each site's ground layer, the number of times that moss was given a cover class value of a 3 or higher
GrndLic>=3	within each site's ground layer, the number of times that a lichen was given a cover class value of a 3 or higher
GrndBare>=3	within each site's ground layer, the number of times that bare ground was given a cover class value of 3 or higher
GrndCWD>=3	within each site's ground layer, the number of times that coarse woody debris was given a cover class value of 3 or higher
GrndLLit>=3	within each site's ground layer, the number of times that leaf litter was given a cover class value of 3 or higher

Results

Raw Captures

In 2000, a total of 972 individuals representing nine species were captured. In 2001 there were 857 individuals representing 11 species. The white-footed mouse (*Peromyscus leucopus*, PELE) accounted for 46.6% and 36.6% of all individuals captured in 2000 and 2001 respectively, and rodents accounted for 88% and 78% of all individuals captured (Table 3).While PELE dominated the rodent catch, this dominance varied by habitat. The meadow vole (*Microtus pennsylvanicus*, MIPE) and meadow jumping mouse (*Zapus hudsonius*, ZAHU) dominated the catch in grassland and wetland habitats. Other species captured were masked shrew (*Sorex cinereus*, SOCI), southern red-backed vole (*Clethrionomys gapperi*, CLGA), short-tailed shrew (*Blarina brevicauda*, BLBR), chipmunk (*Tamias striatus*, TAST), southern flying squirrel (*Glaucomys volans*, GLVO), long-tailed weasel (*Mustela frenata*, MUFR), red squirrel (*Tamiasciurus hudsonicus*, TAHU) and eastern cottontail (*Sylvilagus floridanus*, SYFL) (Table 4).

Estimating Abundance

The ability of CAPTURE to generate abundance estimates for individual species based on weekly sampling sessions was constrained by small sample sizes. There were only 29 instances where 20 or more live individuals of a given species were captured during a weekly trapping session, and only 9 instances involving 30 or more (Figure 2). While there were 233 instances where a sampling session produced at least one live capture of a given species, CAPTURE was unable to produce an estimate of N in 100 (43%) of them. CAPTURE failed to produce an estimate of N in 10% of cases where there were 20 or more individuals of a species captured alive and in 8% of cases where there were 10 or more individuals captured alive. In contrast, for these cases, there were no instances where the modified Lincoln-Petersen estimator failed. Lincoln-Petersen estimates for all species and trapping sessions are presented in Appendix Table 2.

Table 3. Summary of raw captures (#inds, #capture events), percent composition of individuals captured, and catch per 100 trap nights, by species, for 2000 and 2001. All sites and sampling periods combined.

Species	Species Code	2000 Inds	2001 Inds	2000 Caps	2001 Caps	2000 % Inds	2001 % Inds	2000 Inds/100TN	2001 Inds/100TN	2000 Caps/100TN	2001 Caps/100TN
Total (All Species)	---	972	857	2106	1762	100%	100%	6.584	5.476	14.264	11.258
All Rodents	ROD	859	669	1990	1565	88%	78%	5.818	4.274	13.479	9.999
All Shrews	SHREW	110	186	113	195	11%	22%	0.745	1.188	0.765	1.246
Peromyscus leucopus	PELE	453	315	1229	936	47%	37%	3.068	2.013	8.324	5.980
Microtus pennsylvanicus	MIPE	212	138	422	301	22%	16%	1.436	0.882	2.858	1.923
Sorex cinereus	SOCI	97	155	98	162	10%	18%	0.657	0.990	0.664	1.035
Clethrionomys gapperi	CLGA	94	30	205	67	10%	4%	0.637	0.192	1.389	0.428
Zapus hudsonius	ZAHU	90	158	115	226	9%	18%	0.610	1.009	0.779	1.444
Blarina brevicauda	BLBR	13	31	15	33	1%	4%	0.088	0.198	0.102	0.211
Tamias striatus	TAST	5	8	11	10	1%	1%	0.034	0.051	0.075	0.064
Glaucomys volans	GLVO	5	19	8	24	1%	2%	0.034	0.121	0.054	0.153
Mustela frenata	MUFR	3	1	3	1	0%	0%	0.020	0.006	0.020	0.006
Tamiasciurus hudsonicus	TAHU	0	1	0	1	0%	0%	0.000	0.006	0.000	0.006
Sylvilagus floridanus	SYFL	0	1	0	1	0%	0%	0.000	0.006	0.000	0.006

Table 4. Species composition of individuals captured, by habitat, based on all individuals captured in 2000 and 2001.

Species	**Heath**	**Wetland**	**Oak**	**Pine**	**Grass**	**Total**
Total Inds (All Species)	224	615	486	224	280	1829
All Rodents	90.2%	71.4%	89.3%	90.6%	89.3%	83.5%
All Shrews	9.8%	28.1%	10.7%	9.4%	10.0%	16.2%
Peromyscus leucopus	64.7%	20.5%	52.1%	82.1%	21.4%	42.0%
Microtus pennsylvanicus	23.2%	28.5%	7.8%	2.7%	28.2%	19.1%
Sorex cinereus	7.6%	25.7%	8.2%	5.4%	8.9%	13.8%
Clethrionomys gapperi	0.0%	0.0%	24.9%	0.9%	0.4%	6.8%
Zapus hudsonius	0.4%	22.3%	0.0%	0.0%	39.3%	13.6%
Blarina brevicauda	2.2%	2.4%	2.5%	4.0%	1.1%	2.4%
Tamias striatus	1.8%	0.0%	0.0%	4.0%	0.0%	0.7%
Glaucomys volans	0.0%	0.0%	4.5%	0.9%	0.0%	1.3%
Mustela frenata	0.0%	0.5%	0.0%	0.0%	0.4%	0.2%
Tamiasciurus hudsonicus	0.0%	0.2%	0.0%	0.0%	0.0%	0.1%
Sylvilagus floridanus	0.0%	0.0%	0.0%	0.0%	0.4%	0.1%

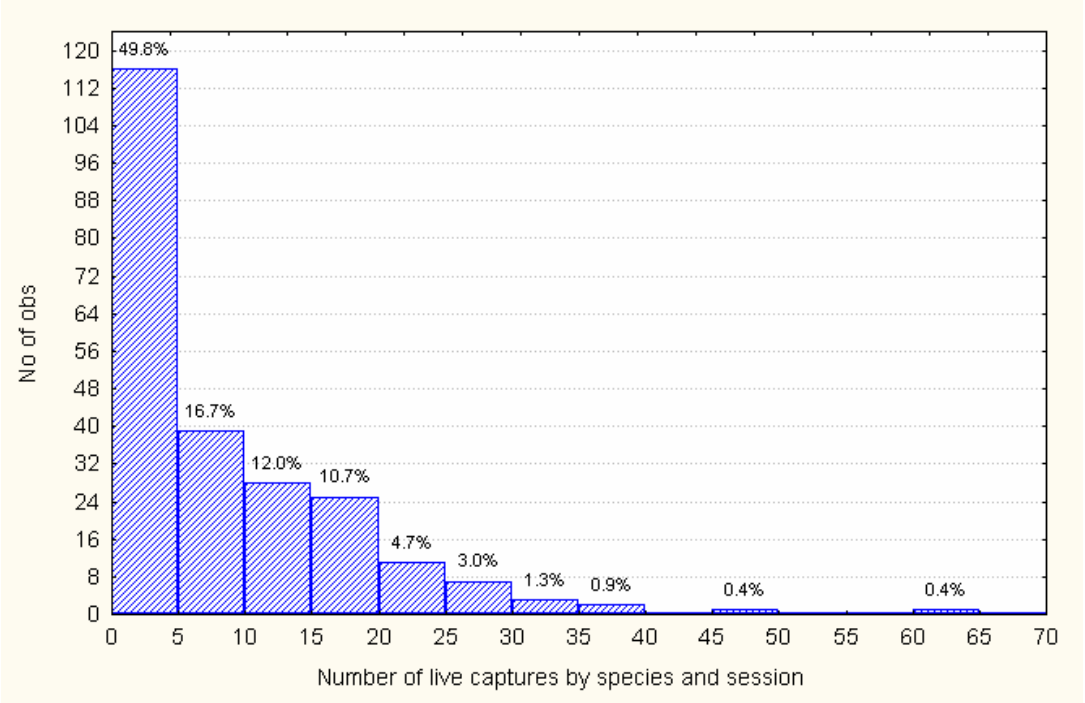


Figure 2. Frequency distribution of total number of individuals of a species captured alive during a weekly sampling session. Only instances where at least one live individual was captured are included.

There were 133 instances where capture data were able to generate estimates of N based on both CAPTURE and Lincoln-Petersen. Correlation between these two estimates of abundance was significant ($r=0.85$, $p<0.01$), as were correlations between CAPTURE and other potential measures of species abundance, i.e. total live individuals captured (INDS_LIV), total live capture events (CAPS_LIV), total individuals captured (INDS_TOT), total capture events (CAPS_TOT), and total individuals captured/100 trap nights (INDEX) (Table 5).

For shrew species (*Sorex cinereus* and *Blarina brevicauda*) there were 60 instances where trapping sessions produced at least one capture. CAPTURE (removal model) failed to produce an estimate of N for 45 (75%). For the 10 cases where total captures exceeded 10, CAPTURE failed 7 times (70%) and for the 21 cases where total captures exceeded 5, CAPTURE failed 13 times (62%). For the 15 cases where CAPTURE produced an estimate of N, there was highly significant correlation between the estimate of N and TOTAL CAPTURES ($r=0.89$, $p<0.000$).

Table 5. Correlation between abundance measures based on counts of individuals and capture events, an index (individuals/100 trap nights), and numerical estimators of population size derived from program CAPTURE (N_{CAP}) and the modified Lincoln-Petersen method (N_{LP}). Marked correlations (*) are significant at $p < .05$. n=133.

Variable	**INDS_LIVE**	**CAPS_LIV**	**INDS_TOT**	**CAPS_TOT**	**INDEX**	**N_{CAP}**	**N_{LP}**
INDS_LIV	1.00						
CAPS_LIV	.94*	1.00					
INDS_TOT	1.00*	.93*	1.00				
CAPS_TOT	.94*	1.00*	.94*	1.00			
INDEX	.98*	.90*	.99*	.91*	1.00		
N_{CAP}	.78*	.63*	.77*	.64*	.76*	1.00	
N_{LP}	.95*	.81*	.95*	.82*	.93*	.85*	1.00

Estimating Survival

Estimation of survival rates using MARK also appear to be constrained by small sample sizes. Two species (*P. leucopus* and *M. pennsylvanicus*) accounted for more than 60% of all individuals and, since species abundance often varied by habitat, even common species provided small sample sizes in some habitats. Consequently, there were few instances where, for a given species and site, a minimum sample size of 20 individuals for the entire sampling season was attained (Figure 3). Since estimation of survival in shrews is not feasible, we focused on the four dominant species of small rodents, i.e. PELE, MIPE, ZAHU, and CLGA. For these species, there were 33 instances with a minimum sample size of 20. MARK provided survival estimates for all 33 instances. However RELEASE Goodness of Fit tests indicated in all cases that data were insufficient to test model goodness of fit (Table 6).

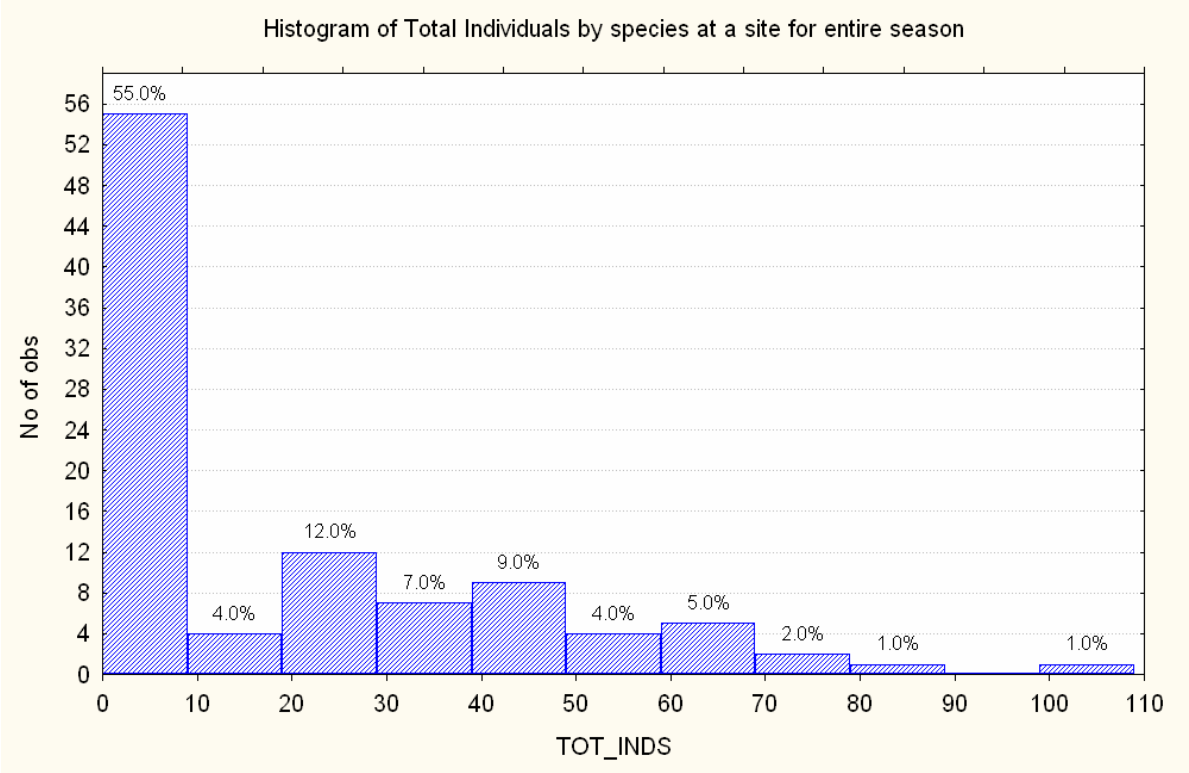


Figure 3. Frequency distribution of total number of individuals of a species captured at a site for an entire season.

Table 6. Estimated monthly survival of small mammals over the course of the trapping season, late spring through late summer. Program RELEASE goodness of fit tests indicate data were insufficient.

Year	Site	Species	#Inds	Model Selected	phi(.)	phi (1)	phi (2)	phi (3)	GOF (p)	Data Sufficient
2000	BBH	MIPE	31	Phi (.), p(.)	0.7883				0.25	no
2000	BBH	PELE	44	Phi (.), p(.)	0.4615				0.26	no
2000	BBP	PELE	45	Phi (.), p(.)	0.3636				0.63	no
2000	CMO	CLGA	80	Phi (.), p(.)	0.3766				0.007	no
2000	CMO	MIPE	29	Phi (.), p(.)	0.7712				1	no
2000	CMO	PELE	108	Phi (.), p(.)	0.4140				0.02	no
2000	DHW	MIPE	44	Phi (.), p(.)	0.4511				1	no
2000	DHW	PELE	41	Phi (.), p(.)	0.6487				0.05	no
2000	DHW	ZAHU	25	Phi (.), p(.)	0.0556				1	no
2000	FHG	ZAHU	37	Phi (.), p(.)	1.0000				1	no
2000	HTW	MIPE	57	Phi (t), p(t)		0.4046	0.6889	0.5388	1	no
2000	HTW	PELE	40	Phi (.), p(.)	0.3969				0.007	no
2000	HTW	ZAHU	27	Phi (.), p(t)	0.2490				0	no
2000	LNO	PELE	70	Phi (t), p(.)		0.5362	0.1958	0.8701	1	no
2000	LNP	PELE	63	Phi (t), p(t)		0.3750	0.9630	0.3536	0.317	no
2000	MCH	PELE	28	Phi (.), p(.)	0.4286				1	no
2000	PHG	MIPE	49	Phi (.), p(t)	0.5573				1	no
2000	PHG	PELE	20	Phi (.), p(.)	0.2222				1	no
2001	BBH	PELE	33	Phi (.), p(.)	0.6824				0.06	no
2001	BBP	PELE	34	Phi (t), p(.)		0.3636	0.5000	0.1176	1	no
2001	CMO	CLGA	25	Phi (.), p(.)	0.9652				1	no
2001	CMO	PELE	49	Phi (t), p(.)		0.7532	0.4316	0.8299	0.02	no
2001	DHW	MIPE	24	Phi (.), p(.)	0.7588				0.62	no
2001	DHW	ZAHU	24	Phi (t), p(.)		0.0000	0.0000	0.5814	0	no
2001	FHG	ZAHU	69	Phi (.), p(t)	0.5943				1	no
2001	HTW	MIPE	53	Phi (t), p(.)		0.5044	0.3680	1.0000	0.25	no
2001	HTW	PELE	31	Phi (t), p(.)		0.7446	0.0000	0.4964	0	no
2001	HTW	ZAHU	62	Phi (.), p(t)	0.2741				1	no
2001	LNO	PELE	36	Phi (.), p(.)	0.5128				0.15	no
2001	LNP	PELE	45	Phi (.), p(.)	0.5276				1	no
2001	MCH	PELE	41	Phi (.), p(.)	0.4995				0.95	no
2001	PHG	MIPE	30	Phi (.), p(.)	0.2917				1	no
2001	PHG	PELE	19	Phi (.), p(.)	0.3529				1	no

Variation in Abundance

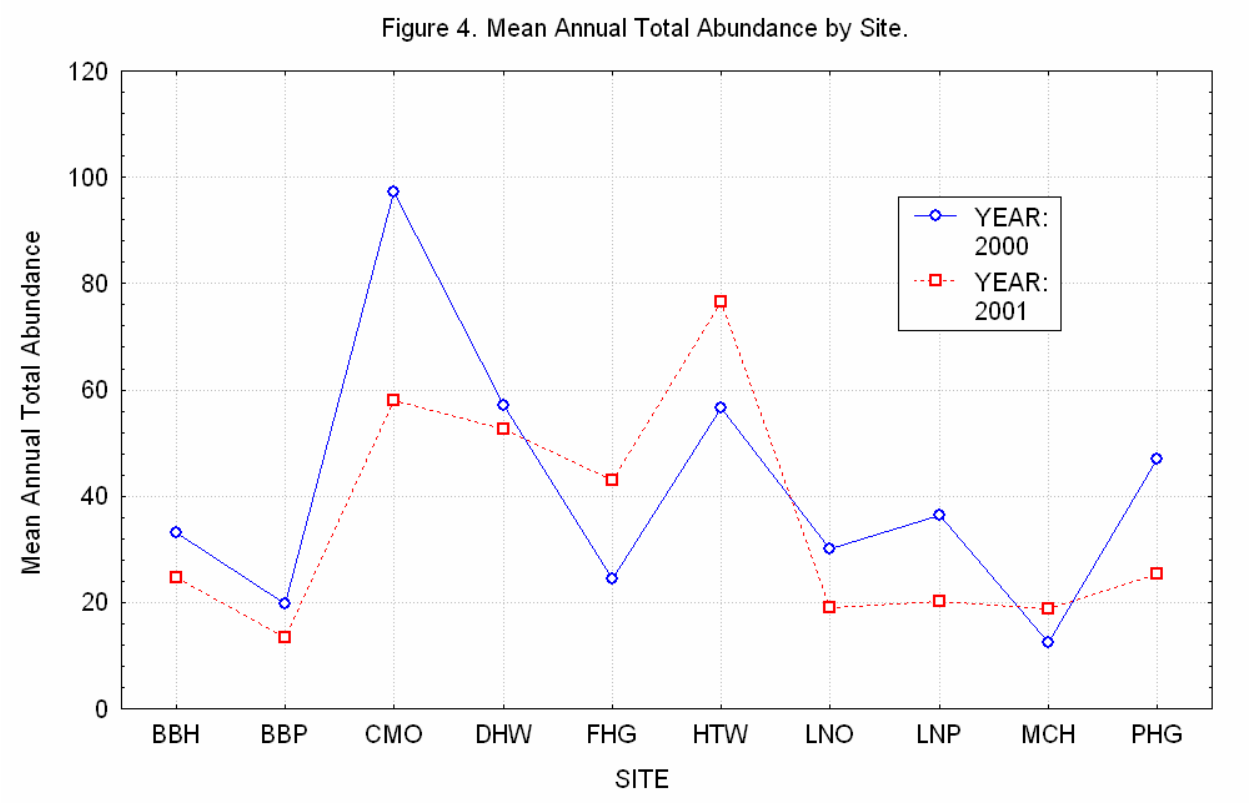
While there was an overall decline in mean annual total abundance and mean annual summed rodent abundance and an overall increase in mean annual summed shrew abundance from 2000 to 2001, none of these differences were statistically significant. For individual species, three of the five most abundant species (PELE, MIPE, and CLGA) showed a significant decrease in mean annual abundance, while the remaining two (SOCI and ZAHU) showed increases that were not statistically significant (Tables 7 and 8). Mean total abundance declined from 2000 to 2001 at seven sites, and increased at three (Figure 4).

Table 7. Variation in estimated abundance due to year, site, and habitat. Analysis of variance results (values of p) for three way ANOVA, with sites nested in habitat. Significant values $p < .05$ are in bold.

	Variation Due to		
Species/Group	**Year**	**Habitat**	**Site (Habitat)**
All Species-summed	0.343	**0.000**	**0.003**
Rodents-summed	0.199	**0.000**	**0.002**
Shrews-summed	0.180	**0.000**	0.157
Peromyscus leucopus	**0.005**	**0.000**	0.079
Microtus pennsylvanicus	**0.003**	**0.000**	**0.000**
Sorex cinereus	0.357	**0.000**	**0.030**
Clethrionomys gapperi	**0.005**	**0.000**	**0.000**
Zapus hudsonius	0.219	**0.000**	**0.000**
Blarina brevicauda	0.147	0.263	0.949
Tamias striatus	0.569	**0.000**	0.612
Glaucomys volans	0.521	**0.000**	**0.000**
Tamiasciurus hudsonicus	0.321	0.415	0.425
Sylvilagus floridanus	0.321	0.415	0.425
Mustela frenata	0.566	0.267	0.891

Table 8. Mean estimated abundance (N_{LP}) by year, for each of 11 species at Cape Cod National Seashore. Bold indicates species where means are significantly different ($p < .05$).

Species/Group	**2000**	**2001**	**% Change**
All Species-summed	41.43	35.16	-15%
Rodents-summed	38.52	29.95	-22%
Shrews-summed	2.83	5.20	84%
Peromyscus leucopus	**19.73**	**13.42**	-32%
Microtus pennsylvanicus	**10.17**	**5.30**	-48%
Sorex cinereus	2.45	4.38	79%
Clethrionomys gapperi	**3.82**	**1.18**	-69%
Zapus hudsonius	4.36	8.54	96%
Blarina brevicauda	0.38	0.83	118%
Tamias striatus	0.18	0.25	39%
Glaucomys volans	0.18	1.20	567%
Tamiasciurus hudsonicus	0.00	0.03	---
Sylvilagus floridanus	0.00	0.03	---
Mustela frenata	0.08	0.03	-63%



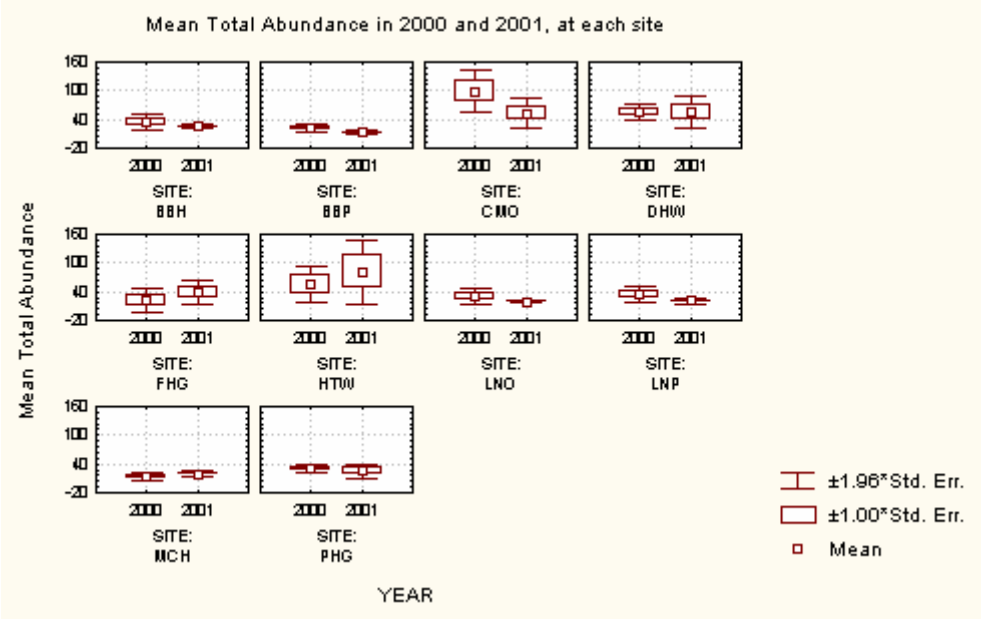


Figure 4. Mean annual estimated total abundance by site. There were no significant site-specific, inter-year differences.

There were significant seasonal differences in total abundance, summed rodent abundance, summed shrew abundance, and for four of the five most abundant species (PELE, MIPE, SOCI, and ZAHU) (Table 9). For most rodents, abundance was lowest in early spring, increasing by early summer and remaining relatively constant over the summer. However, ZAHU showed peaks of abundance in both late spring and late summer. Shrews, both as individual species (i.e. SOCI and BLBR) and summed, increased in abundance over the course of the summer (Table 10, Figure 5). Whereas the overall trend in mean abundance is one of increase from late spring to late summer (Figure 5), the pattern of increase differed in 2000 and 2001. In 2000, mean total abundance increased from late spring to early summer and then remained relatively constant. In 2001, mean total abundance remained relatively constant through mid-summer, and then increased in late summer (Figure 6).

Table 9. Variation in estimated abundance (ANOVA) for each species/group due to season, habitat and site (nested in habitat). Significant values (p<.05) are in bold.

	Variation Due to		
Species/Group	**Season**	**Habitat**	**Site(Habitat)**
All Species-summed	**0.000**	**0.004**	**0.035**
Rodents-summed	**0.002**	**0.029**	0.073
Shrews-summed	**0.000**	**0.002**	0.236
Peromyscus leucopus	**0.001**	**0.001**	0.203
Microtus pennsylvanicus	**0.000**	**0.000**	**0.000**
Sorex cinereus	**0.001**	**0.002**	0.175
Clethrionomys gapperi	0.954	**0.000**	**0.004**
Zapus hudsonius	**0.003**	**0.000**	**0.004**
Blarina brevicauda	**0.003**	0.247	0.918
Tamias striatus	0.170	0.085	0.952
Glaucomys volans	0.328	**0.014**	**0.009**
Tamiasciurus hudsonicus	0.406	0.452	0.465
Sylvilagus floridanus	0.406	0.452	0.465
Mustela frenata	0.319	0.323	0.882

Table 10. Mean estimated abundance, by habitat type, for each of 11 species at Cape Cod National Seashore. Bold indicates species where habitat means are significantly different at p<0.05. Means with same subscript belong to same homogeneous group.

	Heath	**Wet**	**Oak**	**Pine**	**Grass**	**Mean**
All Species-summed	22.25_{1}	60.71_{2}	$51.11_{1,2}$	22.46_{1}	34.96_{1}	38.30
Rodents-summed	20.75_{1}	48.52_{3}	$47.79_{2,3}$	21.09_{1}	$33.02_{1,2}$	34.23
Shrews-summed	1.50_{1}	12.00_{2}	3.31_{1}	1.38_{1}	1.88_{1}	4.01
Peromyscus leucopus	$15.83_{2,3}$	$12.22_{1,2}$	28.09_{3}	$19.46_{2,3}$	7.30_{1}	16.58
Microtus pennsylvanicus	$4.60_{1,2}$	18.35_{3}	$4.06_{1,2}$	0.50_{1}	11.16_{2}	7.74
Sorex cinereus	1.13_{1}	11.00_{2}	2.56_{1}	0.69_{1}	1.69_{1}	3.41
Clethrionomys gapperi	0.00_{1}	0.00_{1}	12.33_{2}	0.06_{1}	0.13_{1}	2.50
Zapus hudsonius	0.06_{1}	17.75_{2}	0.00_{1}	0.00_{1}	14.43_{2}	6.45
Blarina brevicauda	0.38	1.00	0.75	0.69	0.19	0.60
Tamias striatus	0.25	0.00	0.00	0.81	0.00	0.21
Glaucomys volans	0.00_{1}	0.00_{1}	3.31_{2}	$0.13_{1,2}$	0.00_{1}	0.69
Tamiasciurus hudsonicus	0.00	0.06	0.00	0.00	0.00	0.01
Sylvilagus floridanus	0.00	0.00	0.00	0.00	0.06	0.01
Mustela frenata	0.00	0.13	0.00	0.00	0.06	0.04

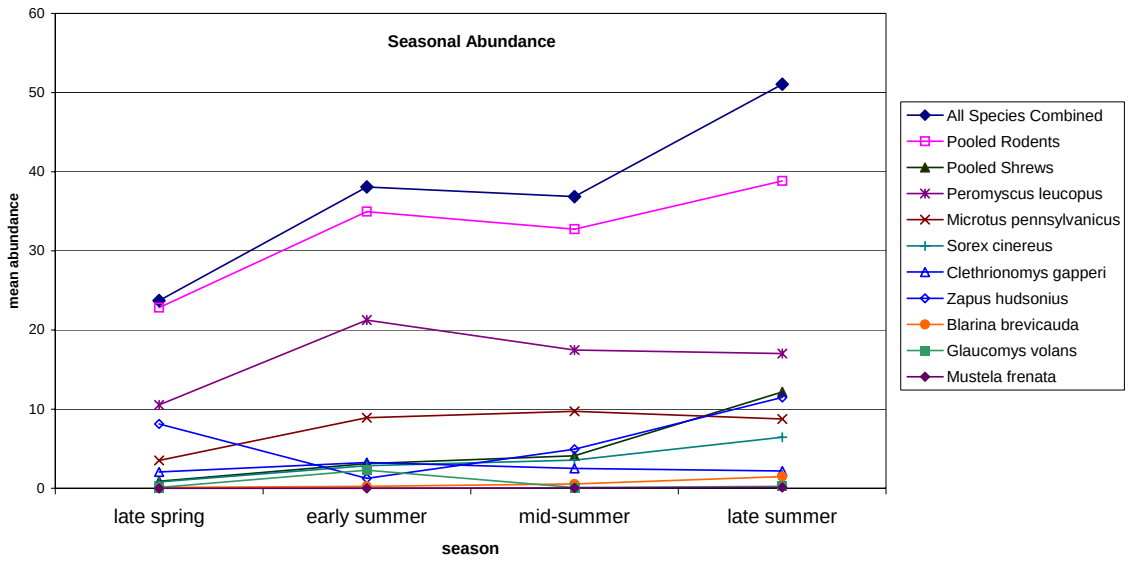


Figure 5. Variation in overall mean estimated abundance by season. For each species or species group, estimated abundance is mean of all sites for 2000 and 2001.

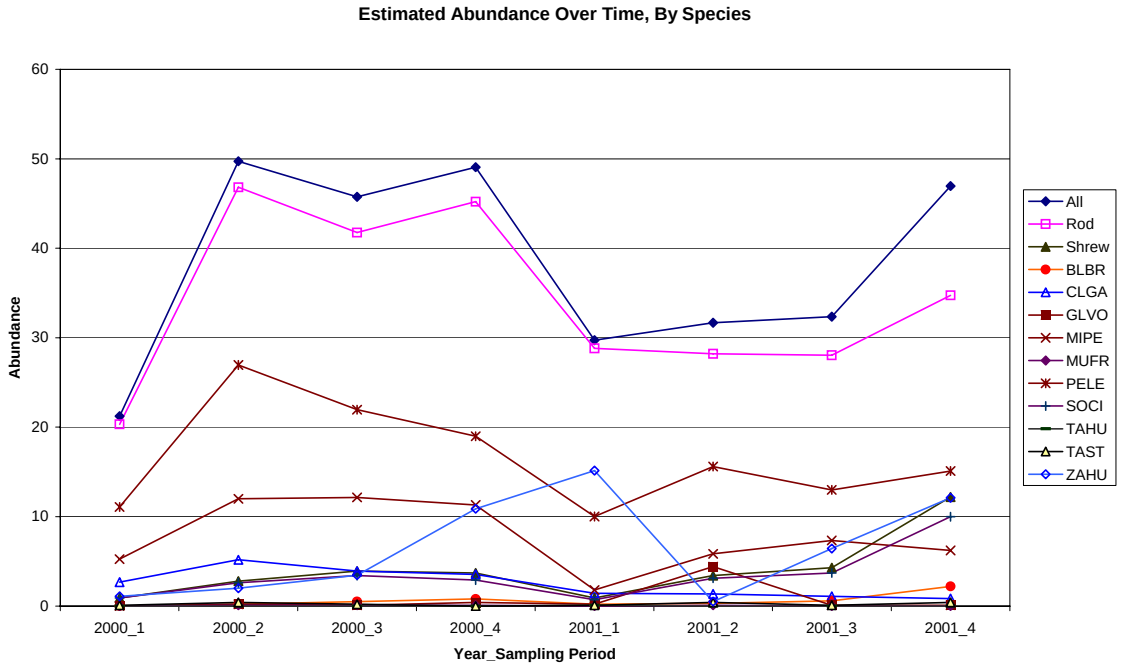


Figure 6. Variation in park-wide estimated abundance of each species or species group over the course of sampling. Data are means of all sites.

Total abundance varied significantly between habitats (Table 10), with wetlands and oak forest having greatest abundance (Table 11). This was also true for summed rodents, whereas shrews were significantly greatest in wetlands. Of the species showing significant habitat differences, PELE were widespread but most abundant in oak forest, pine forest, and heathland. MIPE and SOCI were also widespread, but most abundant in wetlands. CLGA occurred overwhelmingly in oak forest, ZAHU in wetland and grassland, and GLVO in oak forest (Table 11). In addition to differences in abundance due to habitats, there were significant differences between sites nested in the same habitat type for all species summed (total abundance), and MIPE, CLGA, ZAHU, and GLVO (Table 9, 12).

Table 11. Mean estimated abundance by sampling period/season, for each of 11 species at Cape Cod National Seashore. Bold indicates species where means are significantly different. Means with same subscript belong to same homogeneous group.

Species/group	late spring	early summer	mid-summer	late summer
All Species-summed	25.46 $_{1}$	40.69 $_{2}$	39.04 $_{2}$	48.00 $_{2}$
Rodents-summed	24.56 $_{1}$	37.51 $_{2}$	34.89 $_{2}$	39.97 $_{2}$
Shrews-summed	0.90 $_{1}$	3.10 $_{1,2}$	4.10 $_{2,3}$	7.95 $_{3}$
Peromyscus leucopus	10.55 $_{1}$	21.27 $_{2}$	17.46 $_{2}$	17.03 $_{2}$
Microtus pennsylvanicus	3.53 $_{1}$	8.92 $_{2}$	9.74 $_{2}$	8.76 $_{2}$
Sorex cinereus	0.80 $_{1}$	2.85 $_{1,2}$	3.55 $_{2,3}$	6.45 $_{3}$
Clethrionomys gapperi	2.06	3.27	2.50	2.19
Zapus hudsonius	8.13 $_{1,2}$	1.25 $_{1}$	4.94 $_{1,2}$	11.49 $_{2}$
Blarina brevicauda	0.10 $_{1}$	0.25 $_{1}$	0.55 $_{1,2}$	1.50 $_{2}$
Tamias striatus	0.10	0.40	0.15	0.20
Glaucomys volans	0.10	2.30	0.10	0.25
Tamiasciurus hudsonicus	0.05	0.00	0.00	0.00
Sylvilagus floridanus	0.00	0.05	0.00	0.00
Mustela frenata	0.00	0.00	0.05	0.10

Table 12. Mean estimated abundance, by site, for each of 11 species at Cape Cod National Seashore. Means with same subscript belong to same homogeneous group.

	Site									
Species/Group	**BBH**	**MCH**	**BBP**	**LNP**	**CMO**	**LNO**	**DHW**	**HTW**	**FHG**	**PHG**
All Species_summed	28.91 $_{1,2,3}$	15.58 $_{1}$	16.50 $_{1,2}$	28.30 $_{1,2,3}$	77.62 $_{3}$	24.60 $_{1,2,3}$	54.63 $_{2,3}$	66.40 $_{2,3}$	33.59 $_{1,2,3}$	36.19 $_{1,2,3}$
Rodents_summed	26.41 $_{1,2}$	15.08 $_{1}$	16.00 $_{1}$	26.05$_{1,2}$	72.24 $_{2}$	23.35 $_{1,2}$	42.25 $_{1,2}$	54.53 $_{1,2}$	33.09 $_{1,2}$	32.82 $_{1,2}$
Shrews_summed	2.50 $_{1,2}$	0.50 $_{1}$	0.50 $_{1}$	2.25 $_{1,2}$	5.38 $_{1,2}$	1.25 $_{1}$	12.25 $_{2}$	11.75 $_{1,2}$	0.38 $_{1}$	3.38 $_{1}$
Peromyscus leucopus	17.21 $_{1,2,3}$	14.46 $_{1,2,3}$	15.13 $_{1,2,3}$	23.80 $_{2,3}$	35.89 $_{3}$	20.28 $_{2,3}$	10.73 $_{1,2,3}$	13.72 $_{1,2,3}$	4.60 $_{1}$	9.99 $_{1,2}$
Microtus pennsylvanicus	8.70 $_{2,3}$	0.50 $_{1}$	0.00 $_{1}$	1.00 $_{1,2}$	8.13 $_{2,3}$	0.00 $_{1}$	14.46 $_{3}$	22.24 $_{3}$	0.13 $_{1}$	22.20 $_{3}$
Sorex cinereus	1.88 $_{1,2}$	0.38 $_{1}$	0.00 $_{1}$	1.38 $_{1,2}$	4.63 $_{1,2}$	0.50 $_{1}$	11.50 $_{2}$	10.50 $_{1,2}$	0.25 $_{1}$	3.13 $_{1}$
Clethrionomys gapperi	0.00 $_{1}$	0.00 $_{1}$	0.00 $_{1}$	0.25 $_{1}$	21.60 $_{2}$	3.06 $_{1}$	0.00 $_{1}$	0.00 $_{1}$	0.00 $_{1}$	0.13 $_{1}$
Zapus hudsonius	0.13 $_{1}$	0.00 $_{1}$	0.00 $_{1}$	0.00 $_{1}$	0.00 $_{1}$	0.00 $_{1}$	17.06 $_{2}$	18.44 $_{2}$	28.36 $_{2}$	0.50 $_{1}$
Blarina brevicauda	0.63	0.13	0.50	0.88	0.75	0.75	0.75	1.25	0.13	0.25
Tamias striatus	0.38	0.13	0.63	1.00	0.00	0.00	0.00	0.00	0.00	0.00
Glaucomys volans	0.00 $_{1}$	0.00 $_{1}$	0.25 $_{1}$	0.00 $_{1}$	6.63 $_{2}$	0.00 $_{1}$	0.00 $_{1}$	0.00 $_{1}$	0.00 $_{1}$	0.00 $_{1}$
Tamiasciurus hudsonicus	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.00	0.00
Sylvilagus floridanus	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.00	0.00
Mustela frenata	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.13	0.13	0.00

Power Analysis

Power to detect changes in population size depended on the magnitude of the annual changes, as well as the frequency of sampling, the number of sample replicates in a given year, number of sample sites, variability of abundance at sites, and species in question. For total abundance (all species summed), a monitoring program consisting of 10 sites and four seasonal replicates, repeated at 1, 2, 3, or 5 year intervals, was powerful enough to detect changes as low as ± 2% annually with a minimum of 99% power. A monitoring program consisting of a single, late summer sample (i.e. replicate 4), was nearly as powerful (Table 13). Power to detect trends in total abundance at individual sites was less. For a four replicate/year monitoring program, power to detect a 2% annual decline in total abundance at the least variable site (DHW) ranged from75% to 52%, depending on annual frequency (annual, biennial, etc.) and for the most variable site (FHG) it ranged from 43% to 28% (Table 14). For a program monitoring total abundance based on a single, late summer sample (i.e. replicate 4), power to detect a 2% annual decline at the least variable site (FHG) ranged was 100% for all sampling frequencies, and for the most variable site (MCH), it ranged from 18% to 10% (Table 15).

Table 13. Power to detect park-wide changes in estimated total abundance.

A. Four replicates/season. Based on estimates of total abundance at 10 sites, 4 replicates during survey season [a=0.05, trend=exponential, 2 tails, trend CV=0, constant=1, rounding=integers] Initial input data are site mean and SD based on eight sampling events, for each of 10 sites.

			Power to Detect Annual Trends of									
Interval	**Duration**	**TotalSurveys**	**-10%**	**-5%**	**-3%**	**-2%**	**-1%**	**1%**	**2%**	**3%**	**5%**	**10%**
1 year	19	20	100%	100%	100%	100%	90%	93%	100%	100%	100%	100%
2 year	20	11	100%	100%	100%	100%	74%	86%	100%	100%	100%	100%
3 year	21	8	100%	100%	100%	99%	75%	78%	100%	100%	100%	100%
5 year	25	6	100%	100%	100%	97%	74%	87%	100%	100%	100%	100%

B. Replicate four only. Based on estimates of total abundance at 10 sites, only one replicate (#4) during survey season [a=0.05, trend=exponential, 2 tails, trend CV=0, constant=1, rounding=integers] Initial input data are site mean and SD based on two sampling events, for each of 10 sites.

			Power to Detect Annual Trends of									
Interval	**Duration**	**TotalSurveys**	**-10%**	**-5%**	**-3%**	**-2%**	**-1%**	**1%**	**2%**	**3%**	**5%**	**10%**
1 year	19	20	100%	100%	100%	100%	76%	83%	100%	100%	100%	100%
2 year	20	11	100%	100%	100%	96%	63%	69%	99%	100%	100%	100%
3 year	21	8	100%	100%	99%	91%	55%	61%	99%	100%	100%	100%
5 year	25	6	100%	100%	100%	94%	53%	68%	99%	100%	100%	100%

Table 14. Power to detect change in estimated total abundance at least and most variable sites based on four temporal replicates per sampling season.

A. Least variable site. Based on estimates of total abundance at DHW, 4 replicates during survey season [a=0.05, trend=exponential, 2 tails, trend CV=0, constant=1, rounding=integers] Initial input data are site mean (54.84) and SD (20.70) based on eight sampling events at DHW based on one sampling grid at site

			Power to Detect Annual Trends of									
Interval	**Duration**	**TotalSurveys**	**-10%**	**-5%**	**-3%**	**-2%**	**-1%**	**1%**	**2%**	**3%**	**5%**	**10%**
1 year	19	20	100%	100%	97%	75%	29%	38%	93%	100%	100%	100%
2 year	20	11	100%	99%	86%	60%	23%	31%	82%	99%	100%	100%
3 year	21	8	100%	97%	82%	53%	17%	21%	72%	99%	100%	100%
5 year	25	6	100%	98%	78%	52%	23%	28%	82%	99%	100%	100%

B. Most variable site. Based on estimates of total abundance at FHG, 4 replicates during survey sesason [a=0.05, trend=exponential, 2 tails, trend CV=0, constant=1, rounding=integers] Initial input data are site mean (33.72) and SD (25.86) based on eight sampling events at FHG based on one sampling grid at a site

			Power to Detect Annual Trends of									
Interval	**Duration**	**TotalSurveys**	**-10%**	**-5%**	**-3%**	**-2%**	**-1%**	**1%**	**2%**	**3%**	**5%**	**10%**
1 year	19	20	100%	98%	79%	43%	18%	20%	62%	92%	100%	100%
2 year	20	11	100%	92%	59%	34%	15%	12%	41%	78%	100%	100%
3 year	21	8	100%	86%	48%	31%	9%	12%	36%	72%	100%	100%
5 year	25	6	100%	86%	52%	28%	11%	12%	42%	82%	99%	100%

Table 15. Power to detect change in estimated total abundance at least and most variable sites, based on replicate four only.

A. Least variable site. Based on total abundance at PHG, only one replicate (#4) during survey season. [a=0.05, trend=exponential, 2 tails, trend CV=0, constant=1, rounding=integers] Initial input data are site mean (44.0) and SD (1.47) based on two sampling events at PHG.

			Power to Detect Annual Trends of									
Interval	**Duration**	**TotalSurveys**	**-10%**	**-5%**	**-3%**	**-2%**	**-1%**	**1%**	**2%**	**3%**	**5%**	**10%**
1 year	19	20	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%
2 year	20	11	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%
3 year	21	8	100%	100%	100%	100%	98%	100%	100%	100%	100%	100%
5 year	25	6	100%	100%	100%	100%	98%	100%	100%	100%	100%	100%

B. Most variable site. Based on total abundance at MCH, only one replicate (#4) during survey season [a=0.05, trend=exponential, 2 tails, trend CV=0, constant=1 rounding=integers] Initial input data are site mean (15.0) and SD (8.93) based on two sampling events at MCH.

			Power to Detect Annual Trends of									
Interval	**Duration**	**TotalSurveys**	**-10%**	**-5%**	**-3%**	**-2%**	**-1%**	**1%**	**2%**	**3%**	**5%**	**10%**
1 year	19	20	90%	58%	30%	18%	9%	6%	21%	49%	92%	100%
2 year	20	11	77%	38%	20%	11%	6%	9%	16%	35%	76%	100%
3 year	21	8	63%	34%	19%	13%	7%	7%	12%	28%	65%	99%
5 year	25	6	62%	30%	18%	10%	7%	7%	12%	26%	71%	100%

For the most abundant species, PELE, power to detect a 2% annual decline based on a program consisting of 10 sites and four seasonal replicates, ranged from 98% to 100%. For a program consisting of a single, late summer sample (i.e. replicate 4), power to detect a 2% decline ranged from 92% to 79% (Table 16). For individual sites, power varied depending on whether the site exhibited low or high variability. For a four replicate/year monitoring program, power to detect a 2% annual decline at the least variable site (FHG) ranged from 90% to 70%, depending on annual frequency (annual, biennial, etc.) and for the most variable site (HTW) it ranged from 46% to 31% (Table 17). For a program monitoring PELE abundance consisting of a single, late summer sample (i.e. replicate 4), power to detect a 2% annual decline at the least variable site (HTW) ranged from 100 to 94%, depending on annual frequency, and for the most variable site (DHW), it ranged from 14% to 7% (Table 18). For ZAHU, a species that was both a habitat specialist and highly variable in its abundance, power to detect changes in abundance, even in preferred habitat, grassland and wetland habitat, was low (Table 19).

Table 16. Power to detect park-wide changes in estimated PELE abundance.

A. Four replicates/season. Based on estimates of PELE abundance at 10 sites, 4 replicates during survey season [a=0.05, trend=exponential, 2 tails, trend CV=0, constant=1, rounding=integers]. Initial input data are site mean and SD based on eight sampling events, for each of 10 sites.

			Power to Detect Annual Trends of									
Interval	**Duration**	**TotalSurveys**	**-10%**	**-5%**	**-3%**	**-2%**	**-1%**	**1%**	**2%**	**3%**	**5%**	**10%**
1 year	19	20	100%	100%	100%	100%	87%	96%	100%	100%	100%	100%
2 year	20	11	100%	100%	100%	100%	75%	80%	100%	100%	100%	100%
3 year	21	8	100%	100%	100%	98%	68%	77%	100%	100%	100%	100%
5 year	25	6	100%	100%	100%	100%	72%	84%	100%	100%	100%	100%

B. Replicate 4 only. Based on estimates of PELE abundance at 10 sites, only one replicate (#4) during survey season [a=0.05, trend=exponential, 2 tails, trend CV=0, constant=1, rounding=integers]. Initial input data are site mean and SD based on two sampling events, for each of 10 sites.

			Power to Detect Annual Trends of									
Interval	**Duration**	**TotalSurveys**	**-10%**	**-5%**	**-3%**	**-2%**	**-1%**	**1%**	**2%**	**3%**	**5%**	**10%**
1 year	19	20	100%	100%	100%	92%	53%	65%	100%	100%	100%	100%
2 year	20	11	100%	100%	97%	83%	41%	49%	100%	100%	100%	100%
3 year	21	8	100%	100%	95%	81%	33%	44%	99%	100%	100%	100%
5 year	25	6	100%	100%	96%	79%	42%	48%	100%	100%	100%	100%

Table 17. Power to detect change in estimated PELE abundance least and most variable sites, based on four temporal replicates per sampling season.

A. Least variable site. Based on estimates of PELE abundance at FHG, 4 replicates during survey season [a=0.05, trend=exponential, 2 tails, trend CV=0 constant=1, rounding=integers]. Initial input data are site mean (4.6) and SD (1.27) based on eight sampling events.

			Power to Detect Annual Trends of									
Interval	**Duration**	**TotalSurveys**	**-10%**	**-5%**	**-3%**	**-2%**	**-1%**	**1%**	**2%**	**3%**	**5%**	**10%**
1 year	19	20	100%	100%	99%	90%	41%	58%	100%	100%	100%	100%
2 year	20	11	100%	100%	95%	76%	31%	37%	95%	100%	100%	100%
3 year	21	8	100%	98%	89%	68%	28%	33%	91%	100%	100%	100%
5 year	25	6	100%	98%	91%	70%	30%	41%	95%	100%	100%	100%

B. Most variable site. Based on estimates of PELE abundance at site HTW, 4 replicates during survey season [a=0.05, trend=exponential, 2 tails, trend CV=0, constant=1, rounding=integers] Initial input data are site mean (13.72) and SD (10.52) based on eight sampling events.

			Power to Detect Annual Trends of									
Interval	**Duration**	**TotalSurveys**	**-10%**	**-5%**	**-3%**	**-2%**	**-1%**	**1%**	**2%**	**3%**	**5%**	**10%**
1 year	19	20	100%	98%	73%	46%	17%	16%	60%	91%	100%	100%
2 year	20	11	100%	89%	53%	30%	13%	13%	46%	78%	100%	100%
3 year	21	8	99%	83%	45%	25%	12%	14%	39%	74%	99%	100%
5 year	25	6	99%	87%	52%	31%	15%	12%	41%	76%	99%	100%

Table 18. Power to detect change in estimated PELE abundance at least and most variable site, based on replicate four only.

A. Least variable site. Based on estimates of PELE abundance at HTW, only one replicate (#4) during survey season [a=0.05, trend=exponential, 2 tails, trend CV=0, constant=1, rounding=integers]. Initial input data are site mean=27.75 and SD=1.77 based on two sampling events.

			Power to Detect Annual Trends of									
Interval	**Duration**	**TotalSurveys**	**-10%**	**-5%**	**-3%**	**-2%**	**-1%**	**1%**	**2%**	**3%**	**5%**	**10%**
1 year	19	20	100%	100%	100%	100%	93%	99%	100%	100%	100%	100%
2 year	20	11	100%	100%	100%	100%	74%	87%	100%	100%	100%	100%
3 year	21	8	100%	100%	100%	96%	65%	76%	100%	100%	100%	100%
5 year	25	6	99%	99%	99%	94%	59%	76%	100%	100%	100%	100%

B. Most variable site. Based on estimates of PELE abundance at DHW, only one replicate (#4) during survey season [a=0.05, trend=exponential, 2 tails, trend CV=0, constant=1, rounding=integers]. Initial input data are site mean=14.87 and SD=14.90 based on two sampling events.

			Power to Detect Annual Trends of									
Interval	**Duration**	**TotalSurveys**	**-10%**	**-5%**	**-3%**	**-2%**	**-1%**	**1%**	**2%**	**3%**	**5%**	**10%**
1 year	19	20	84%	43%	21%	14%	6%	6%	17%	33%	72%	100%
2 year	20	11	65%	30%	15%	8%	5%	7%	13%	20%	53%	98%
3 year	21	8	53%	25%	11%	9%	5%	6%	10%	15%	44%	94%
5 year	25	6	42%	21%	12%	7%	4%	6%	11%	19%	46%	93%

Table 19. Power to detect change in estimated ZAHU abundance, at 6 sites for both all four replicates and only replicate four.

A. Four replicates/season. Based on estimates of ZAHU abundance at 6 sites, 4 replicates during survey season [a=0.05, trend=exponential, 2 tails, trend CV=0, constant=1, rounding=integers]. Initial input data are site mean and SD based on eight sampling events, for each of 6 sites.

			Power to Detect Annual Trends of									
Interval	**Duration**	**TotalSurveys**	**-10%**	**-5%**	**-3%**	**-2%**	**-1%**	**1%**	**2%**	**3%**	**5%**	**10%**
1 year	19	20	64%	53%	39%	30%	14%	17%	51%	87%	100%	100%
2 year	20	11	54%	46%	34%	23%	11%	10%	43%	74%	99%	100%
3 year	21	8	50%	35%	31%	22%	9%	13%	33%	67%	98%	100%
5 year	25	6	44%	42%	29%	22%	12%	14%	41%	77%	99%	100%

B. Replicate 4 only. Based on estimates of ZAHU abundance at 6 sites, only one replicate (#4) during survey season [a=0.05, trend=exponential, 2 tails, trend CV=0, constant=1, rounding=integers]. Initial input data are site mean and SD based on two sampling events, for each of 6 sites.

			Power to Detect Annual Trends of									
Interval	**Duration**	**TotalSurveys**	**-10%**	**-5%**	**-3%**	**-2%**	**-1%**	**1%**	**2%**	**3%**	**5%**	**10%**
1 year	19	20	63%	69%	70%	64%	52%	49%	76%	85%	97%	100%
2 year	20	11	56%	59%	62%	57%	41%	42%	69%	83%	96%	100%
3 year	21	8	47%	54%	58%	52%	32%	38%	69%	79%	94%	100%
5 year	25	6	40%	47%	52%	46%	32%	41%	67%	83%	95%	100%

Community Structure

Multi-dimensional scaling based on mammal species composition shows that sites are generally ordered along a gradient from woodland to heathland to grassland/wetland. However, except for the two wetland sites, a site's "nearest neighbor" is generally a different habitat "type" (Figure 7). Bray–Curtis similarity values (Table 20) similarly show that, except for the two wetland sites, the site most similar to a given site was of a different habitat type. When summarized by habitat types, community composition showed the same gradient from grassland/wetland to woodland habitats (Figure 8). Two way crossed ANOSIM found no significant variation in species composition due to year (R=-0.036, p=0.863) or seasons (R=0.02, p=0.243). Two way nested ANOSIM of sites within habitat found significant variation in species composition among sites within each habitat (R=0.454, p=0.001) but none between habitats (R=-0.25, p=0.891). In addition, when sites were analyzed by two way ANOSIM (sites x seasons) without *a priori* habitat categories, there were significant differences between sites (R=0.642, p=0.001). Similarly, two way ANOSIM based on habitat categories (habitats x seasons) found significant between-habitat differences in community composition (R=0.424, p=0.001). Based on RELATE, the similarity matrix of site species composition was highly correlated with the similarity matrix of habitat variables (Spearman Rho=0.360, p=0.012), indicating a correlation between site mammal community structure and habitat structure.

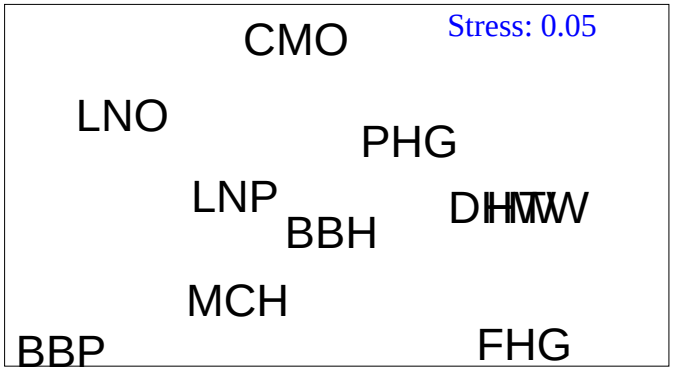


Figure 7. Ordination of small mammal communities at each sampling site. Based on mean estimated abundance of each species at each site.

Table 20. Similarity of small mammal community at sample sites. Bray-Curtis similarity matrix based on mean estimated abundance of each species at each site. Bold indicates sites with similar mammal communities, determined by two-way ANOSIM (sites x season).

	BBH	**BBP**	**CMO**	**DHW**	**FHG**	**HTW**	**LNO**	**LNP**	**MCH**	**PHG**
BBH										
BBP	61.97									
CMO	66.16	47.88								
DHW	75.64	39.05	60.60							
FHG	58.94	38.63	40.69	77.76						
HTW	**72.35**	37.79	58.58	**94.10**	73.24					
LNO	60.70	**58.44**	67.26	49.81	48.22	47.87				
LNP	**81.82**	**65.10**	68.71	59.81	50.91	57.89	**75.52**			
MCH	**79.76**	**68.59**	55.42	57.90	61.00	55.33	**66.76**	**81.34**		
PHG	**81.67**	41.98	68.89	79.63	61.48	77.34	62.11	71.82	65.62	

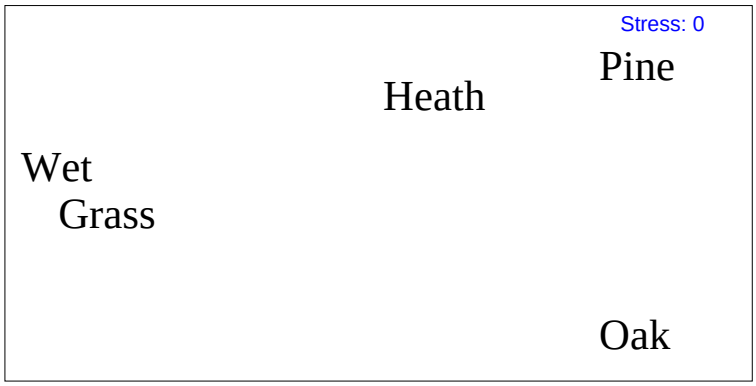


Figure 8. Ordination of small mammal communities by habitat type. Based on mean estimated abundance of each species in each site habitat type.

By site, species diversity (H') was greatest at High Toss Wetland and Duck Harbor Wetland and lowest at Marconi Heath and Bound Brook Pine. By habitat, H' was greatest in wetlands and lowest in pine forest. Evenness generally followed diversity (Table 21). For individual sites, species diversity was highly correlated with mean total abundance ($r=0.89$, $p<0.001$), whereas when sites were pooled together by habitats, correlation between diversity and abundance was not significant ($r=0.83$, $p=0.08$). Multiple regression of species diversity (H') on habitat PCA scores was non significant ($R^2=0.311$, $F(2,7)=1.5847$, $p=0.27058$). Variables included in the model were PCA3 (beta= -396, p=0.26) and PCA 2 (beta= -393, p= 0.25).

Table 21. Measures of species diversity (H'(log e)), evenness (J'), number of species recorded (S), and mean estimated total abundance for sample sites and by habitat. H' and J' were calculated using estimated abundance.

Sample Site/Habitat	S	J'	H' (log e)	Mean Total Abundance
BBH	6	0.5651	1.012	29.12
BBP	4	0.2698	.374	16.69
CMO	6	0.7657	1.372	76.04
DHW	6	0.8016	1.436	55.41
FHG	6	0.2881	.5163	31.27
HTW	7	0.7394	1.439	68.40
LNO	4	0.4356	.6039	24.68
LNP	6	0.379	.679	28.04
MCH	5	0.218	.3509	16.05
PHG	6	0.5473	.9806	28.54
Heath	6	0.4772	.855	22.58
Pine	7	0.3079	.5991	22.36
Oak	6	0.7047	1.263	50.36
Wet	7	.7412	1.442	61.90
Grass	7	.6446	1.254	29.91

Habitat Relationships

Principal components analysis of habitat variables identified four components with eigenvalues >1.0, accounting for 86.7% of the total variation (Table 22). Principal component 1 relates primarily to the overstory, with high negative loadings on overstory variables and coarse woody debris, defining a gradient from forested to non-forested habitat. Principal component 2 relates to the understory and ground layers, with high positive loadings representing heath-like conditions, e.g. short conifers and ground level lichens, woody ground cover (*Arctostaphlos* or *Corema*), and bare ground. Principal component 3 also relates to the understory and ground layers, where high negative loadings on understory cover and ground level ferns in conjunction with high positive loading on ground level graminoids, represent a gradient from moist, open habitat to dry grassland. Principal component 4, with high negative loadings on ground level graminoids and conifer basal area, and high positive loadings on deciduous basal area and ground level forbs describes a gradient from an open, grassy pine forest to a deciduous forest with a forb-dominated ground layer.

Ordination of sites based on principal components analysis of habitat parameters also shows sites organized along a gradient from forested to non-forested, with sites belonging to each of the five habitat types forming clusters (Figure 9). Multiple regression did not find significant relationships between total abundance and habitat principal components, nor for most individual species (Table 23). *Peromyscus leucopus* abundance showed a strong negative relation to non-forested habitats or, more straightforwardly, a strong positive relationship to forested habitats. *Zapus hudsonius* showed a strong positive relationship to non-forested, non-heath habitats, and *Sorex cinereus* showed a negative relationship with dry, grassy ground cover. In addition, there was no significant relationship between site species diversity values, and site habitat PCA scores (multiple $R=0.558$, $R^2=0.312$, $F(2,7)=1.5847$, $p<0.27058$).

Table 22. Eigenvalues and eigenvectors of significant habitat principal components.

Variable	**PC1**	**PC2**	**PC3**	**PC4**
Eigenvalue	9.25	5.23	1.90	1.82
%Variation	44	24.9	9.10	1.82
Cum%Variation	44	69	78	86.7
OvTreeStem	-0.312	0.095	0.059	.123
BA Total	-0.321	0.045	0.111	-0.016
MeanBA/Tree	-0.304	0.135	0.112	-0.041
BA Dead	-0.316	-0.038	0.041	-0.008
BA Live	-0.320	0.049	0.117	-0.020
BA Decid	-0.224	-0.075	-0.049	0.442
BA Conif	-0.240	0.105	0.212	-0.300
UndTot%Cov	-0.216	0.064	-0.434	-0.162
Und%Decid	-0.235	-0.140	-0.340	-0.252
Und%Conif	0.128	0.382	0.010	0.068
GrndHerb>3	0.208	-0.272	0.244	0.017
GrndGram>3	0.149	-0.064	0.441	-0.331
GrndForb>3	0.141	-0.294	-0.072	0.398
GrndWshrb>3	0.015	-0.306	0.056	0.294
GrndWgrnd>3	0.105	0.391	-0.097	0.153
GrndFern>3	0.070	-0.165	-0.555	-0.127
GrndMoss>3	0.071	0.272	0.004	0.271
GrndLic>3	0.105	0.391	-0.097	0.153
GrndBare>3	0.157	0.323	-0.003	-0.012
GrndCWD>3	-0.242	-0.085	0.111	0.281
GrndLLit>3	-0.279	0.071	0.061	0.183

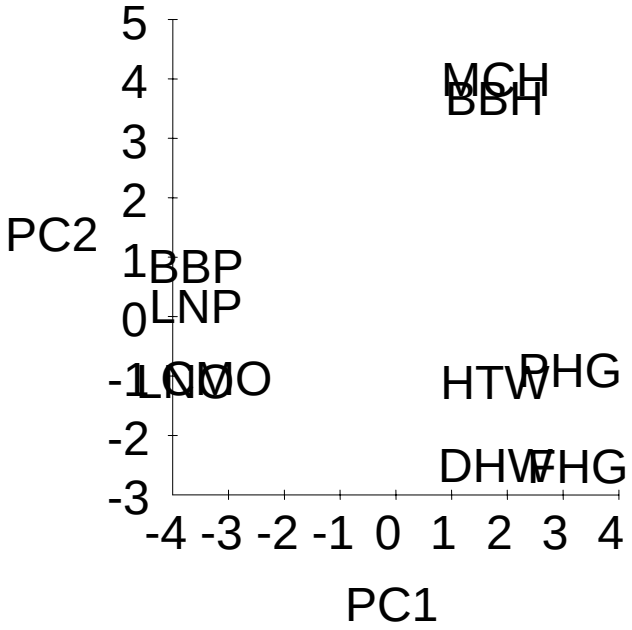


Figure 9. Ordination of sites by Principal Components Analysis based on habitat data.

Table 23. Results of forward stepwise linear regression of small mammal estimated abundance on habitat principal component scores. Significant relationships (P<0.05) are shown in bold.

	Overall Model					Variables included in Model		
Species	**Multiple R**	**R^2**	**df**	**F**	**p**	**Variable**	**Beta**	**p**
All	0.405	0.164	1,8	1.567	0.246	PCA2	-0.405	0.246
PELE	**0.739**	**0.546**	**1,8**	**9.640**	**0.015**	**PCA1**	**-0.739**	**0.015**
MIPE	0.497	0.247	1,8	2.621	0.144	PCA1	0.497	0.144
ZAHU	**0.825**	**0.680**	**2,7**	**7.445**	**0.018**	**PCA2**	**-0.609**	**0.025**
						PCA1	**0.556**	**0.035**
CLGA	0.687	0.471	2,7	3.121	0.107	PCA4	0.533	0.093
						PCA1	-0.432	0.160
BLBR	0.699	0.488	2,7	3.337	0.096	PCA1	-0.608	0.059
						PCA4	-0.344	0.244
SOCI	**0.896**	**0.804**	**4,5**	**5.114**	**0.051**	**PCA3**	**-0.784**	**0.011**
						PCA2	-0.308	0.181
						PCA1	0.230	0.298
						PCA4	-0.202	0.356
TAST	0.717	0.513	2,7	3.693	0.080	PCA4	-0.559	0.072
						PCA1	-0.448	0.133
MUFR	0.680	0.462	3,6	1.717	0.262	PCA1	0.448	0.185
						PCA2	-0.407	0.223
						PCA3	0.309	0.341
GLVO	0.510	0.260	1,8	2.809	0.132	PCA1	-0.510	0.132

Discussion

Estimating Abundance

Considering that, for an individual species at a given site, most sampling sessions captured relatively few individuals, the multiple mark and recapture methods of Program CAPTURE do not appear appropriate for CACO. CAPTURE's inability to produce population estimates in many instances, and the fact that there were very few instances where sample sizes approached the minimum needed for model selection to work ("several times larger than 10 to 20"; White et al. 1982) indicate that small mammals at CACO are not abundant enough for this method of population estimation to work here. When compared to small mammal live trapping studies elsewhere, capture rates at CACO are at the low end of the range (Table 24). Monitoring programs using CAPTURE, such as at Channel Islands N.P and Denali N.P. have larger sample sizes.

Menkens and Andersen (1988) noted that CAPTURE often produces poor estimates of population size when sample size is small and McKelvey and Pearson (2001) reported that 98% of published studies they reviewed had samples too small (<100) for effective model selection by CAPTURE. Menkens and Andersen (1988) recommend using Chapman's version of the Lincoln-Petersen estimator, and found that its superior performance was due to the pooling of multiple trapping periods into single mark and recapture periods. This approach was used by Ellison and van Riper (1998), whose sample sizes were very similar to those at CACO, and was one of the methods for estimating abundance tested in this study.

McKelvey and Pearson (2001) suggest that the widespread use of indices (e.g. number of unique individuals captured) rather than population estimators is due to the constraints of small sample size. Moreover, both Slade and Blair (2000) and McKelvey and Pearson (2001) found high correlation between indices (counts) and numeric estimators, indicating that indices closely track numeric estimations of abundance and generally lead to the same conclusions regarding trends and population comparisons. The results of this study (Table 5), with highly significant correlations between different indices and numeric estimators are consistent with these findings.

While the high correlation between indices and numeric estimators suggest that abundance could be monitored using indices, this approach has draw backs and is not recommended. Indices are not direct estimates of a population size and the relationship between an index and the actual population is generally not known (Conroy 1996, Thomson et al. 1998). In addition, since indices do not estimate capture probabilities, it is not possible to know if differences in index values across time, space, or between species reflect differences in abundance or differences in capture probabilities (Nichols 1986, Williams et al. 2001). For this reason, even though indices are correlated with numerical estimators, both Slade and Blair (2000) and McKelvey and Pearson (2001) concur with Nichols (1986) in recommending estimators over indices, particularly when making inter-specific comparisons.

Table 24. Comparison of small mammal capture rates at CACO with other, similar studies. Abundance method refers to manner in which abundance was expressed or calculated. CHIS=Channel Islands National Park, CA. DENA=Denali National Park, AK.

Location	**Caps/100TN**	**Inds/100TN**	**Abundance Method**	**Source**
CACO	12.82	6.06	various	this study
Eastern MA	21.44		#inds	Adler 1988
Upper Cape	17.99	11.26	#inds	Stevens and Cavanaugh 1997
Arizona	12.12	5.38	LP	Ellison and Van Riper 1998
CHIS		26.45	CAPTURE	Schwemm and Coonan 2001
Washington		12.71	inds/100TN	Taylor 1999
DENA2001	11.1	6.95	CAPTURE	Rexstad and Debevec 2001b
DENA2000	16.45		CAPTURE	Rexstad and Debevec 2001a
DENA 1999	30.79		CAPTURE	Rexstad and Debevec 1999
Tennesse	21.54	4.83	caps/100TN	Kitchings and Levy 1981
Illinois	29.49	23.65	caps/100TN	Hoffman et al. 2001

Given the strong arguments for numerical estimators and the constraints of the multiple mark and recapture method, the most satisfactory solution is to use the modified Lincoln-Petersen estimator. It was capable of providing population estimates when CAPTURE was not, and, in instances where both estimators worked, it produced results highly correlated with those of CAPTURE (Table 5). In addition to its broader utility, the use of a modified Lincoln-Petersen estimator offers other practical advantages. Lincoln-Petersen requires that animals be marked so that counts of individuals captured during the mark and recapture sessions can be made, which can be done with colored marking pens (Petryszyn undated). CAPTURE requires marking for individual recognition, so that capture histories of every unique individual can be constructed. In the protocol tested, this was done with PIT tags. While PIT Tags offer efficient and positive identification with low incidence of tag loss, their cost ($4.35 each when purchased in lots of 500-999 in 2002) is high. Based on the ca. 900 individuals captured annually in 2000 and 2001, PIT tag costs were nearly $4000/year. While ear tags are a far less expensive alternative for individual marking (ca. $0.10/tag), marking animals for individual identification requires more time in the field. Use of CAPTURE also requires more time in terms of preparing individual capture histories and running the analyses. Thus, in addition to being more appropriate for the sample sizes obtained at CACO, the use of the modified Lincoln-Petersen estimator is also less costly in terms of staff time (in and out of the field) and materials than CAPTURE.

Estimating Survival

Similar to estimates of abundance, sample size appears to be a serious constraint to survival estimation. Sample size is insufficient to allow for goodness of fit testing of models for even the largest samples obtained for a given species at a site, over the course of a sampling season (Table 6). This situation means the protocol tested has extremely limited ability to provide comparisons of survival rates between species, sites, or years. While increased sample size could be obtained by expanding the size of the trapping grid or doubling up the number of traps set at each grid point, this additional effort may not be feasible for a general monitoring program. Unless there is a specific question or hypothesis that requires survival estimation, it is probably not worth the additional effort and expense to increase sample size for survival estimation.

Variation in Abundance

Introduction

Variation in small mammal abundance, often quite extreme, is well documented and has been extensively studied. There are over 20 hypotheses to explain population cycles, and, as of 1992 over 1000 publications on the subject (Oli and Dobson 2001). Yet there is more disagreement than agreement among ecologists regarding causes of population cycles, and many hypotheses are controversial (Oli and Dobson 1999). Many small mammal population ecologists have focused on defining and understanding cyclic v. non-cyclic variation. Taitt and Krebs (1985) consider non-cyclic populations to have long term multi-annual fluctuations in density less than 5-fold and cyclic populations to have greater than 10-fold fluctuations. Testing small mammal abundance data to determine if a population is cyclic or non-cyclic requires at least 4-5 consecutive years of data, collected at the same time/season of year, such as autumn (Henttonen et al. 1985) or spring (Tamarin et al. 1987). Yet, others argue that defining populations as cyclic

or non-cyclic is artificial and that population fluctuations show a gradient from nearly stable to cyclic to chaotic (Sandell et al. 1991).

All models of small mammal population variation recognize that variation is product of the interaction of intrinsic factors, such as rates of survival, reproduction, dispersal, age at maturity, etc. and extrinsic factors, primarily food and predators (Krebs 1996, Oli and Dobson 2001), as well as weather (Tamarin et al. 1987). Even among proponents of cyclicity, there is recognition of "considerable temporal and spatial variation in the existence of these multi-annual cycles, as well as in the shape or morphology of the cycles that do occur. Some of this variation is inter-specific, that is some species are cyclic and others are not" (Lidicker 1988). In addition, there is intra-specific variation, with cyclicity believed to be most pronounced in northern or boreal populations and less well defined in temperate ones (Grant 1976, Lidicker 1988, Sandell et al. 1991).

Interestingly, much of the work in North America involving cyclic variation is based on studies of the herbivorous *Microtus* and lemmings (reviewed in Krebs 1996, Oli and Dobson 1999, 2001). In contrast, studies of non-cyclic, "irruptions" seem to involve the granivorous-omnivorous *Peromyscus*, whose population fluctuations are driven by mast production (Bowman et al. 2001, Elkinton et al. 1996. Ostfeld 1996, Wolff 1996). While population variation is undoubtedly the product of interacting intrinsic and extrinsic factors (Krebs 1996, Hansson 1998), it would appear that the relative importance of each is, at least partly, a function of herbivory versus granivory. High densities of the herbivorous *Microtus* are capable of having far more immediate and profound impact on the habitat that provides food and cover than the granivorous *Peromyscus*. This would promote a tighter system of cyclic feedback between *Microtus* and its habitat, whereas when *Peromyscus* responds to increases in mast production, the resultant high density of *Peromyscus* does not directly impact the stand of adult trees or determine subsequent mast production. Thus, while there are numerous examples of cyclic variation involving *Microtus* in grasslands, analysis of 43 years of data from an Ontario forest failed to detect evidence of population cycles in *Peromyscus* and six of seven other species (Fryxell et al. 1998).

Clearly, variation in small mammal abundance is not explained by any one variable, and multi-factor models, recognizing the many intrinsic and extrinsic factors that act synergistically and sequentially to produce density changes (Lidicker 1998) are now proposed (Hansson 1998). Grant (1976) pointed out that differences in the pattern of abundance between *Clethrionomys* and *Peromyscus* populations in boreal and deciduous forests reflected both differences in the extrinsic factors as well as differences in each species' response to them. In addition, temporal variation in extrinsic factors will add another layer of complexity by causing the relative importance of both extrinsic and intrinsic factors to vary over time as well.

Given the many different factors responsible for variation in small mammal abundance, and their varying importance, understanding the reasons for trends and variation in small mammal abundance is challenging, and the numbers of studies with seemingly incompatible findings understandable.

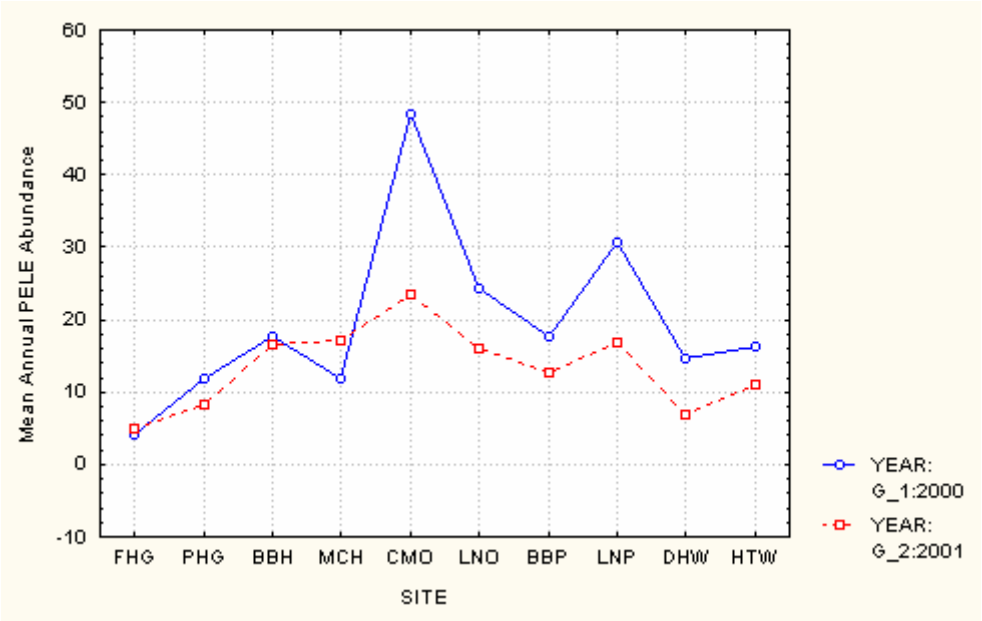
Annual Variation

While there is a large body of literature on annual variation in individual species abundance, generally at single sites, there are comparatively few published studies of annual variation at the community level, i.e. of the total assemblage of species. In one of the longer term data sets, spanning 43 years in forested habitats (ranging from pure deciduous to pure coniferous) at Alongonquin Provincial Park, Ontario (Fryxell et al. 1998), maximum abundance was ca. 1600% greater than minimum, and consecutive year change ranged from 0% to +321%. In the majority of instances, consecutive year change was less than 50%, and only 6 of 42 inter-year periods experienced change exceeding 100%. In an 11 year study in deciduous forest in Quebec (Grant 1976), maximum June abundance was 1000% greater than minimum, and consecutive year change ranged from 1-226%, with a frequency distribution similar to that of Fryxell (1998). Over a three year period, in New Brunswick forest, total abundance increased 208% over the first two years, and 3% over the last two (Bowman et al 2000). For a community at Point Pelee National Park in Ontario, with species composition similar to CACO, sampled across a variety of habitat types for two consecutive years, inter-year change in abundance was +56% (Morris 1984). Similarly, for a time sequence of successional old fields in Minnesota, also of species composition similar to CACO, inter-year change was +42% (Huntly and Inouye 1987).

In comparison, the -15% annual change in park-wide abundance from 2000 to 2001 at CACO is small, but due to the short term nature of the data it is impossible to know how variable overall park-wide abundance here is over the long term. While it is likely that over a longer term, we would observe greater change between consecutive years, the coefficient of variation (CV) for our data (0.57) is intermediate to those for longer term data sets, 0.44 over 43 years (Fryxell et al. 1998) and 0.82 over 11 years (Grant 1976).

While the relatively low amount of inter-year variability in total abundance park-wide suggests a certain degree of stability, the extent of inter-year variation also depends on the level of resolution looked at, e.g. individual or combined sites, individual or combined species. The lack of a significant park-wide difference in total abundance between 2000 and 2001 (Table 7,8) is partly a mean effect, the result of some individual species and/or sites remaining fairly constant, while others went either up or down, in a compensatory fashion (Figure 4, Table 8). While none of the individual sites differed significantly in total abundance from 2000 to 2001, at the species level, PELE, MIPE, and CLGA showed significant park-wide declines from 2000 to 2001. However, these declines were compensated for by non-significant park-wide increases in SOCI, ZAHU, BLBR, TAST, and GLVO, such that the between year decline in total abundance (from 41.43 to 35.16 individuals/sampling site/sampling period) was not significant (Table 8). A similar pattern was found in New Brunswick, where there was considerable species level variation from year to year, yet between year total abundance was far less variable (Bowman et all. 2000).

Annual variation may also be manifested at the individual site level. While there were no significant site-specific inter-year differences in total abundance, there were for some species (Table 8, Figures 10 -16). Moreover, while significant park-wide inter-year differences for a species were generally associated with significant site-specific inter-year differences, this was not always the case. For MIPE, the significant park-wide decline in total abundance from 2000 to



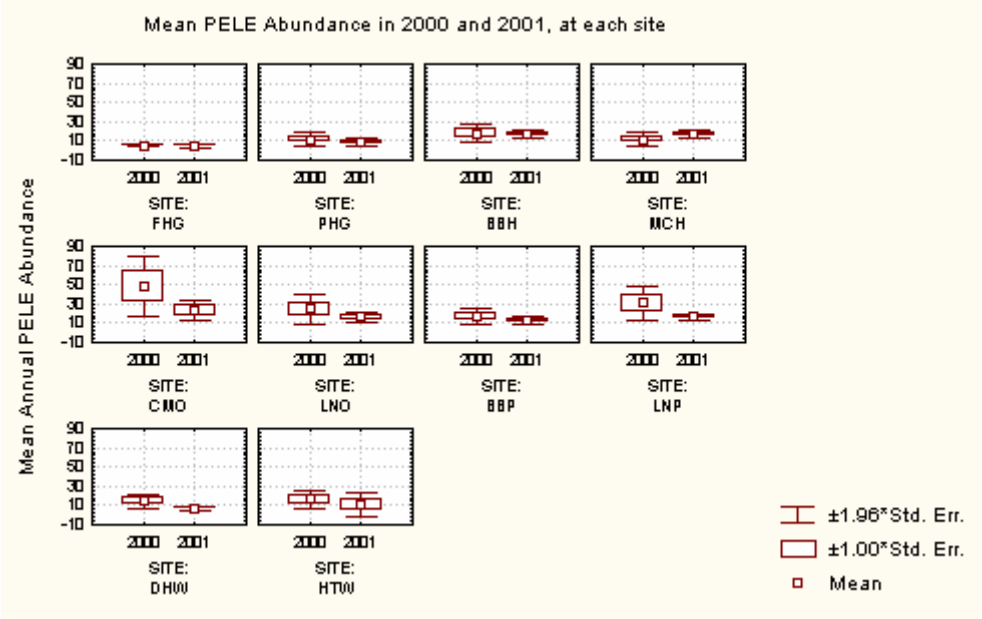
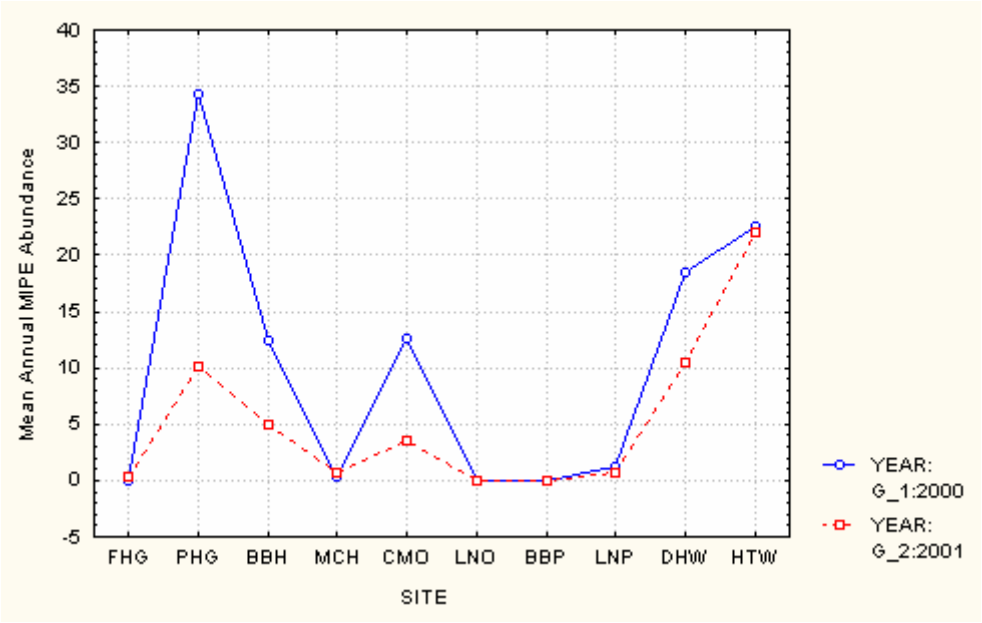


Figure 10. Mean annual estimated abundance of PELE by site. Site-specific, inter-year comparisons found no significant differences.



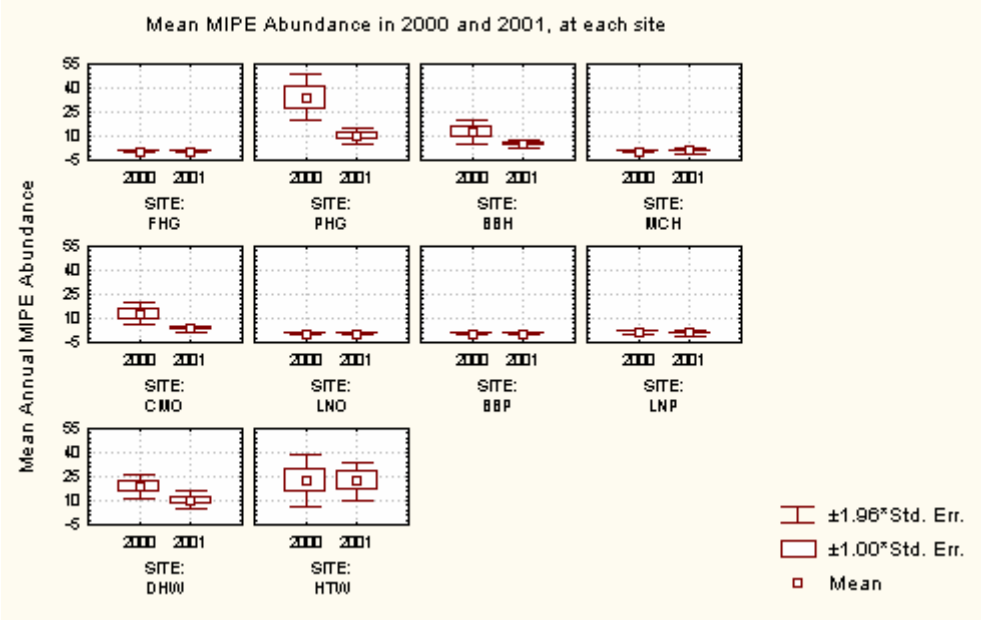
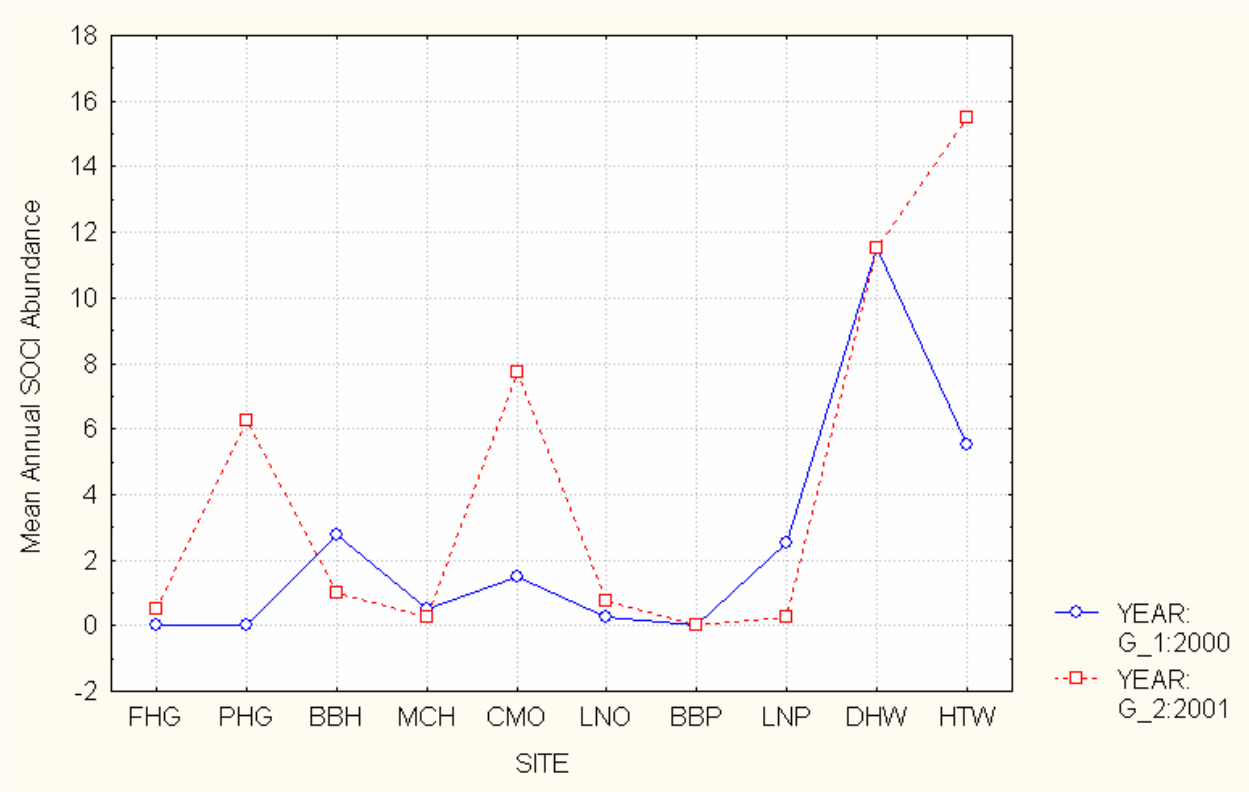


Figure 11. Mean annual estimated abundance of MIPE by site. There were significant inter-year differences at sites Cemetary Oak (CMO) (p=0.01) and Pilgrim Heights Grassland (PHG) (p=0.02).



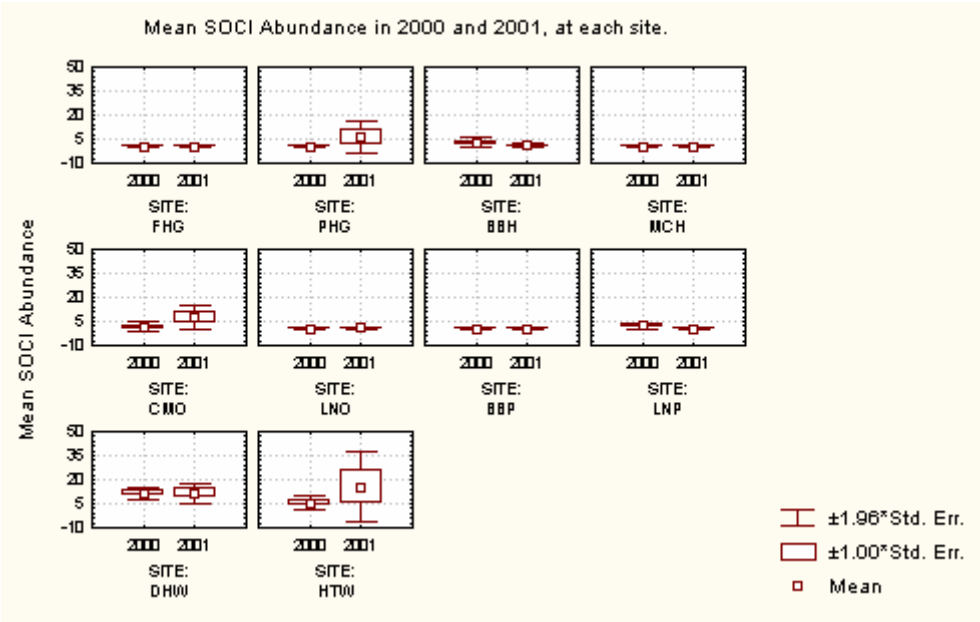
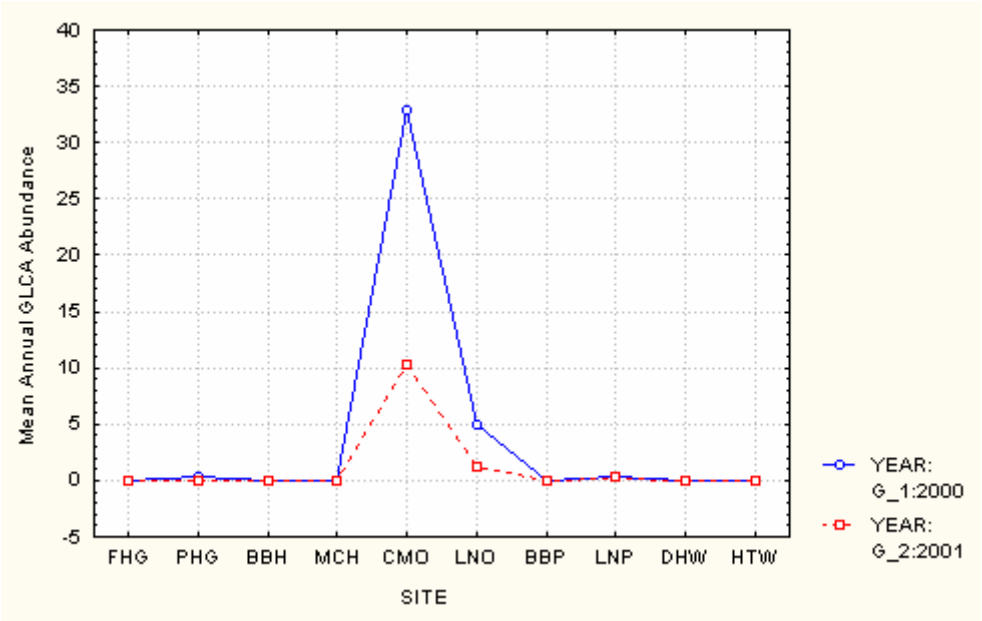


Figure 12. Mean annual estimated abundance of SOCI by site. There were no site-specific, inter-year differences.



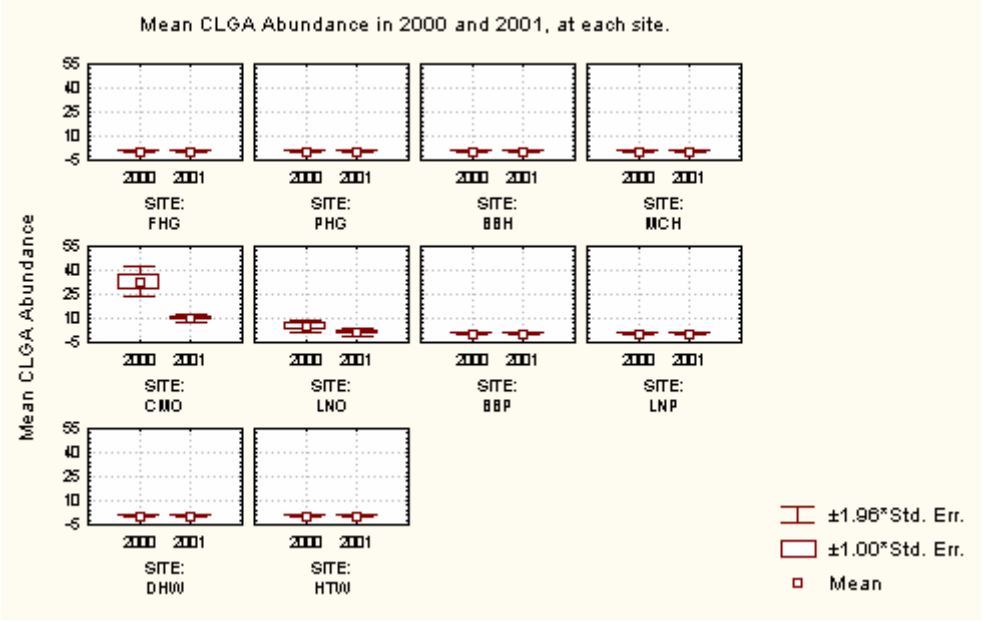
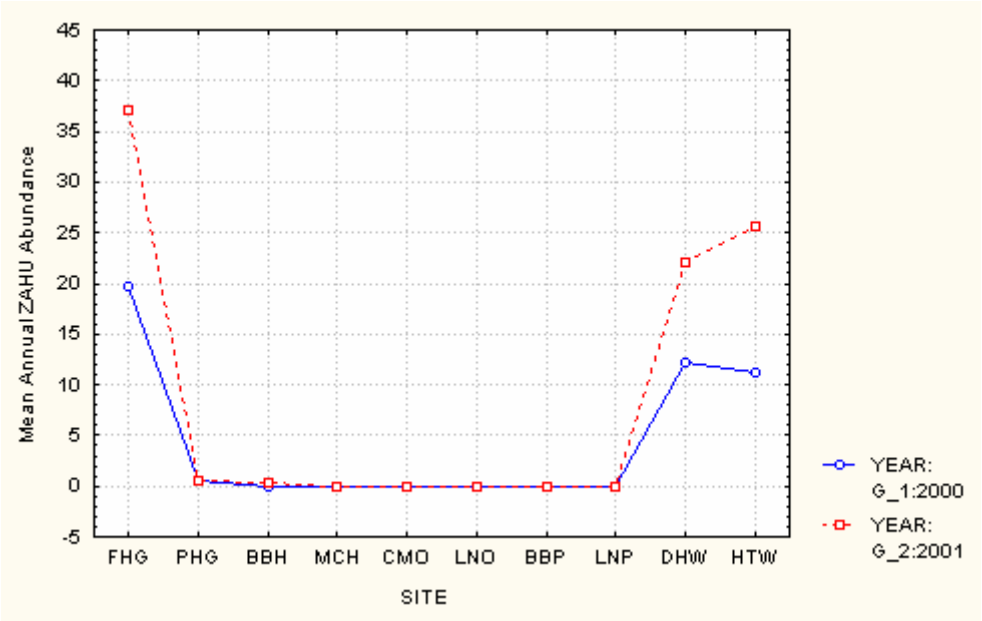


Figure 13. Mean annual estimated abundance of CLGA by site. There were significant inter-year differences at Cemetary Oak (CMO) (p=0.0007).



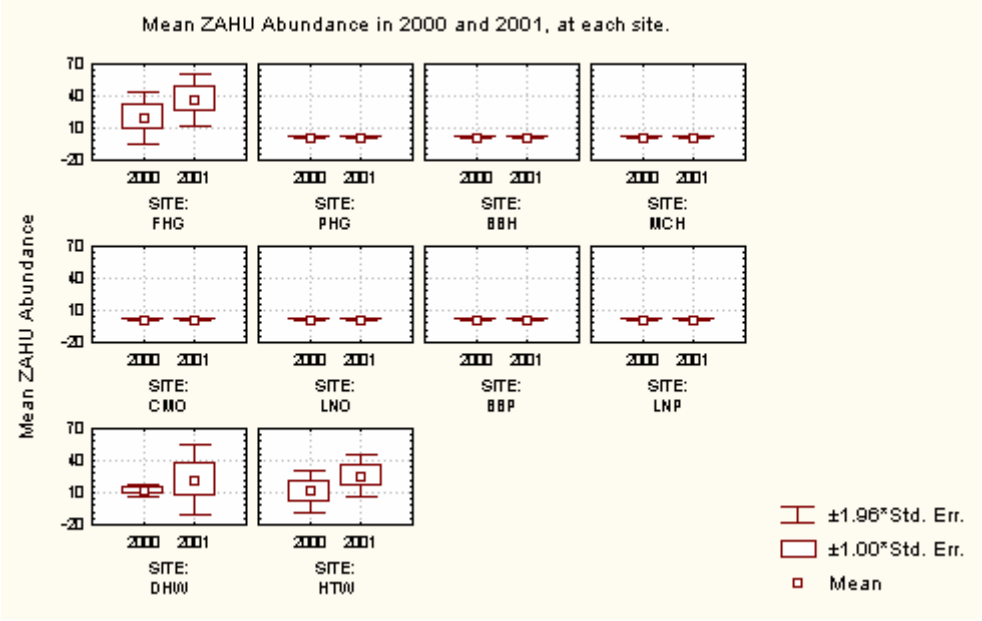
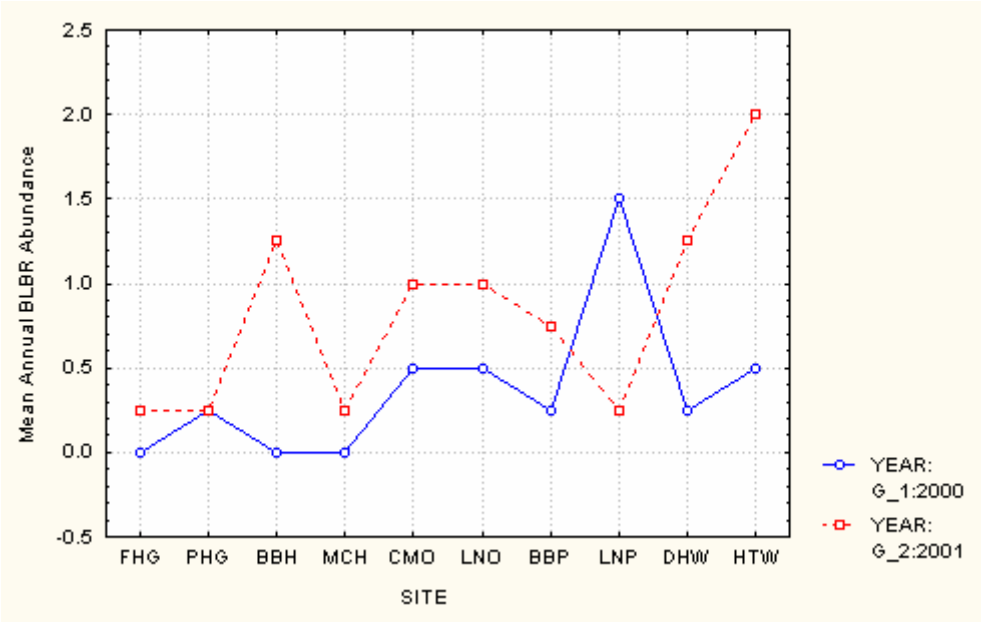


Figure 14. Mean annual estimated abundance of ZAHU by site. There were no significant site-specific inter-year differences.



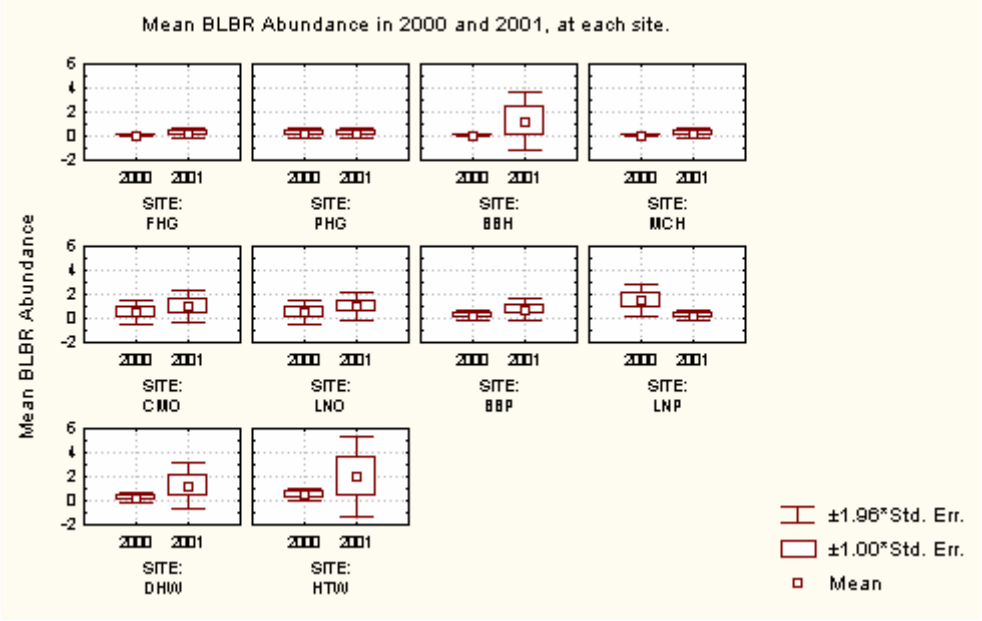
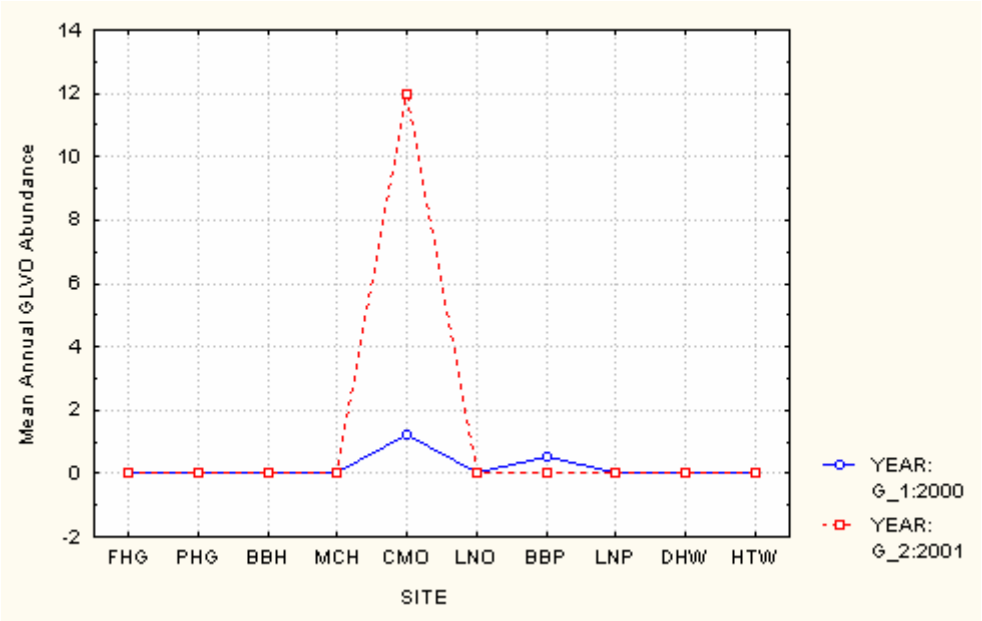


Figure 15. Mean annual estimated abundance of BLBR by site. There were no significant site-specific inter-year differences.



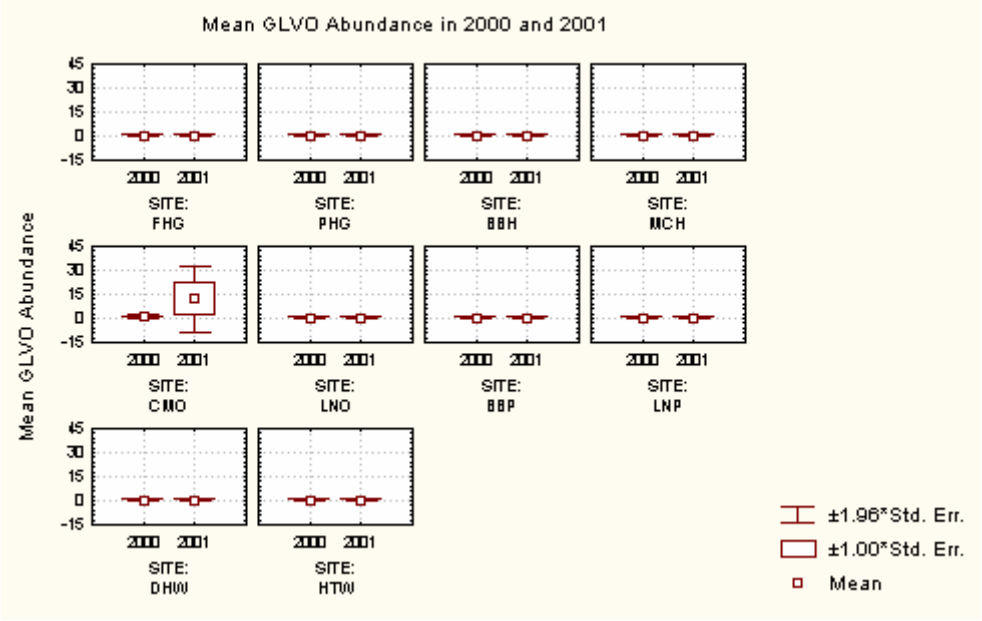


Figure 16. Mean annual estimated abundance of GLVO by site. There were no significant site-specific inter-year differences.

2001 was primarily due to significant declines at two sites, Pilgrim Heights Grassland and Cemetery Oak (Figure 11), whereas CLGA, largely restricted to oak forest, declined significantly at Cemetery Oak (Figure 13). However, though park-wide PELE abundance was significantly less in 2001 (Table 8), no individual sites showed statistically significant differences in mean abundance from 2000 to 2001 (Figure 10).

Underlying annual variability at the species and site level are a great variety of potential causes and explanations. As noted above, the primary factor driving variation in PELE abundance is variation in mast production, which is itself a complex process involving weather and genetic factors (Elkinton et al. 1996). Even within a relatively small area, there can be considerable variation in acorn production, and while declines of mast crops may be synchronized locally, at a finer scale the extent may vary over a couple of orders of magnitude (Elkinton 1996). Moreover, there is often considerable fine scale temporal and spatial variation in mammal abundance, even within what appears to be homogeneous habitat (Bowman et al. 2000, Krohne and Burgin 1990) or different sites supposedly sampling the same habitat type (Morris 1984).

Inter-specific interactions can also influence annual variation at the site and species level. Habitat generalist PELE (Adler and Wilson 1989) interacts antagonistically with MIPE, a grassland specialist, such that variation of MIPE displaces PELE in grassland habitats (Bellows et al. 2001). However, while Ostfeld (1997, 2002) found that PELE density was greater in fields with low MIPE density, Adler (1984) found that PELE abundance in eastern Massachusetts grasslands was not related to the density of grassland specialists MIPE and ZAHU. Rather, the PELE occurring in grasslands were individuals dispersing from adjacent woodlands, and it was PELE density in the woodlands that drove PELE density in the grasslands (Adler 1984). Thus, if competitive interactions between MIPE and PELE occur, we would expect to see their abundance move in opposition. While we are not aware of any long term data collected in grasslands/old fields that might help resolve this question, long term data from forested habitats, where MIPE did not occur, found that population fluctuations of species were synchronized (Fryxell 1998). Annual variation in abundance is also affected by winter weather, with extremes of severity or mildness correlated with ups and downs of spring density (Tamarin et al 1987), which frequently sets the stage for abundance over the course of the spring and summer.

Given the complexity of these processes and their inter-relations, and the limited data for CACO, it is difficult to make much of the between year variation observed here. The data, when examined from a park-wide perspective (Figure 6) or by individual sites (Figures 17-26) show that variation between species was generally synchronized, and there was little evidence of negative relationships between PELE and MIPE. When viewed from the perspective of a single species across multiple sampling sites (Figures 27-33), between site variation is also generally synchronized, though there is some variation in seasonal patterns that is discussed below. Overall, the amount of variation in abundance of individual species from 2000 to 2001 at the park-wide level (Table 8) is relatively small compared to the magnitude of inter-year variation recorded at other sites in New England and eastern Canada (Adler 1985, Bowman et al. 2000, Fryxell et al. 1998, Grant 1976, Huntly and Inouye 1987, Morris 1984, Seamon and Adler 1996, Tamarin et al. 1987).

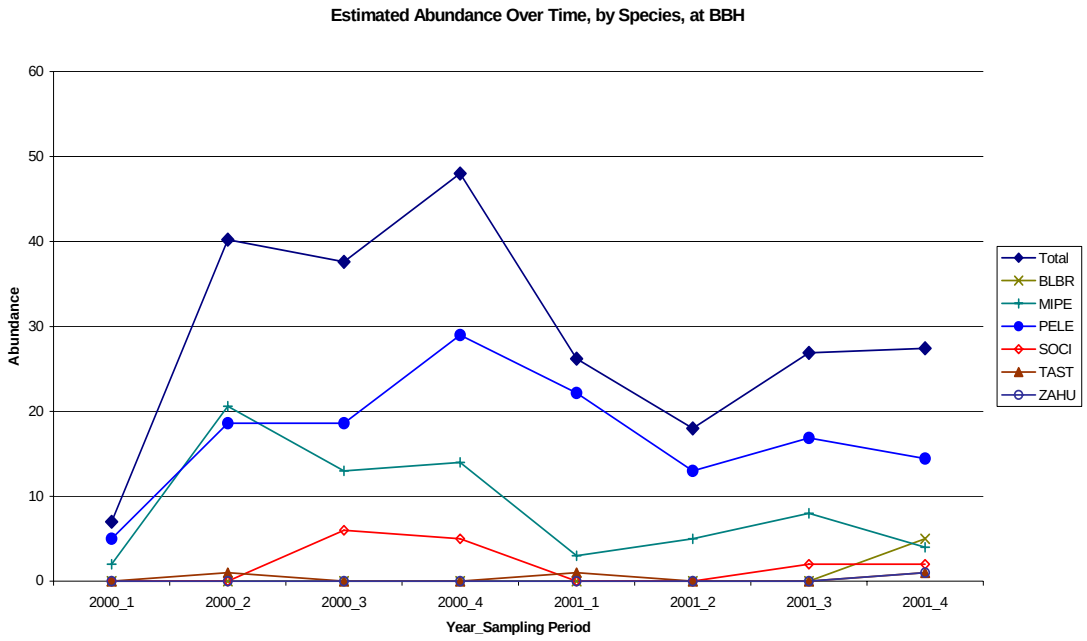


Figure 17. Temporal variation in estimated abundance of each species at Bound Brook Heathland over the course of sampling.

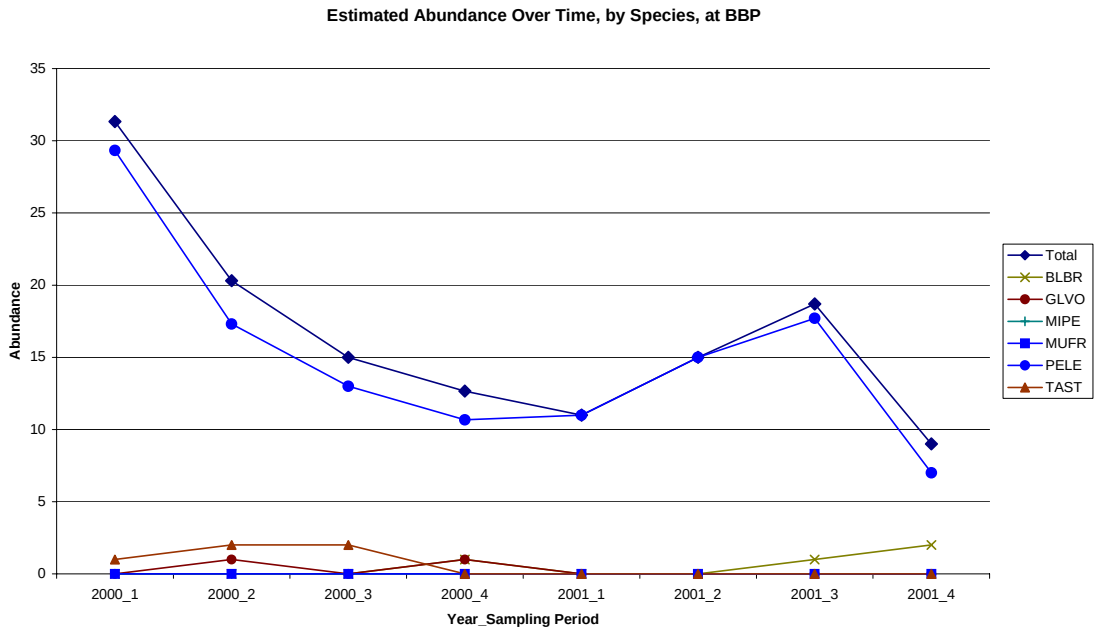


Figure 18. Temporal variation in estimated abundance of each species at Bound Brook Pine Forest over the course of sampling.

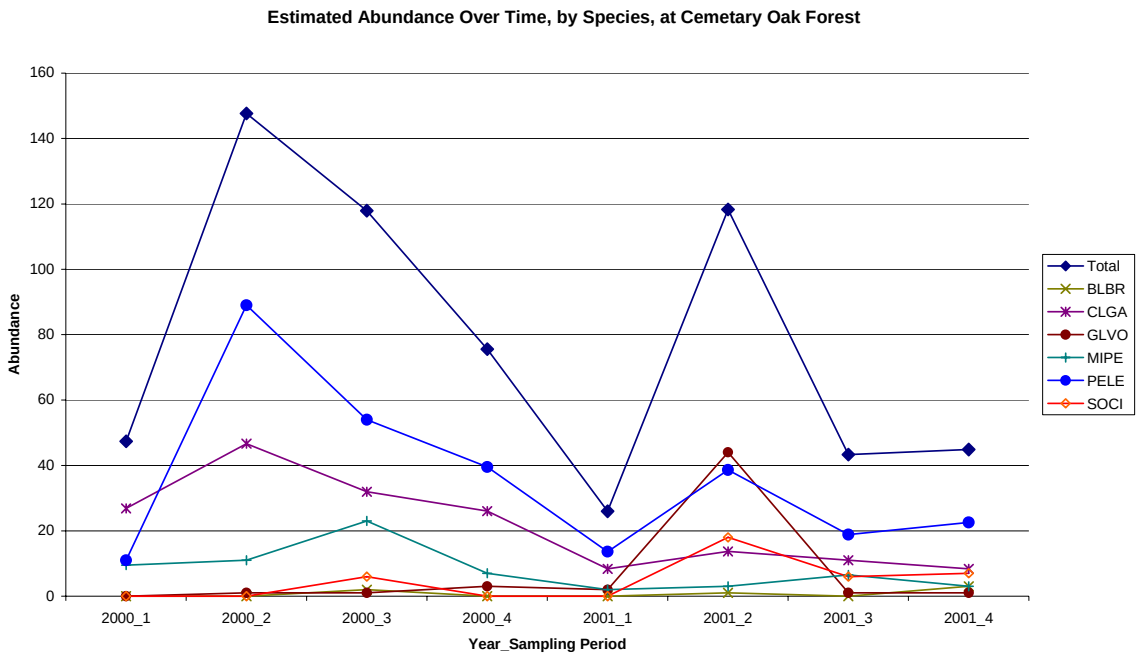


Figure 19. Temporal variation in estimated abundance of each species at Cemetery Oak Forest over the course of sampling.

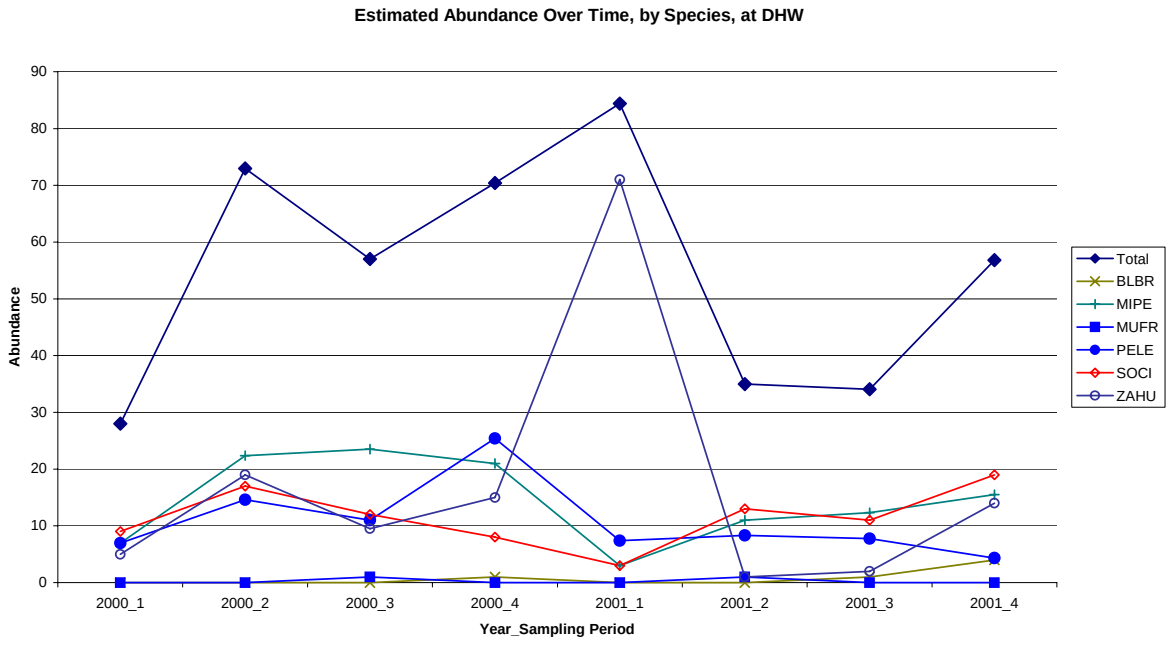


Figure 20. Temporal variation in estimated abundance of each species at Duck Harbor Wetland over the course of sampling.

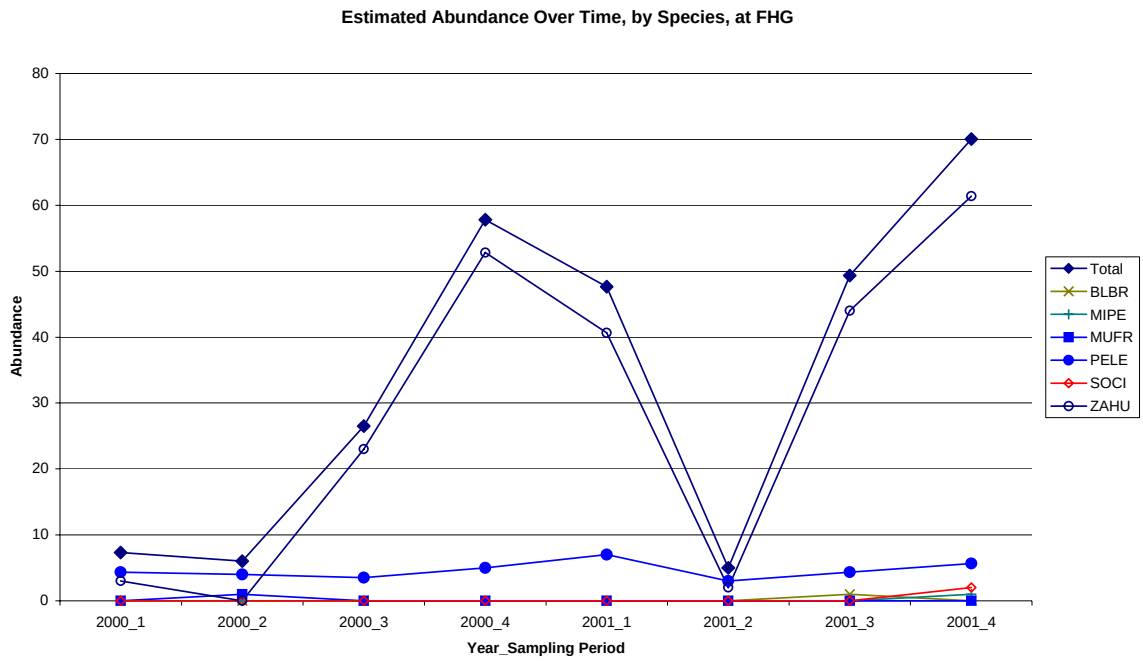


Figure 21. Temporal variation in estimated abundance of each species at Fort Hill Grassland over the course of sampling.

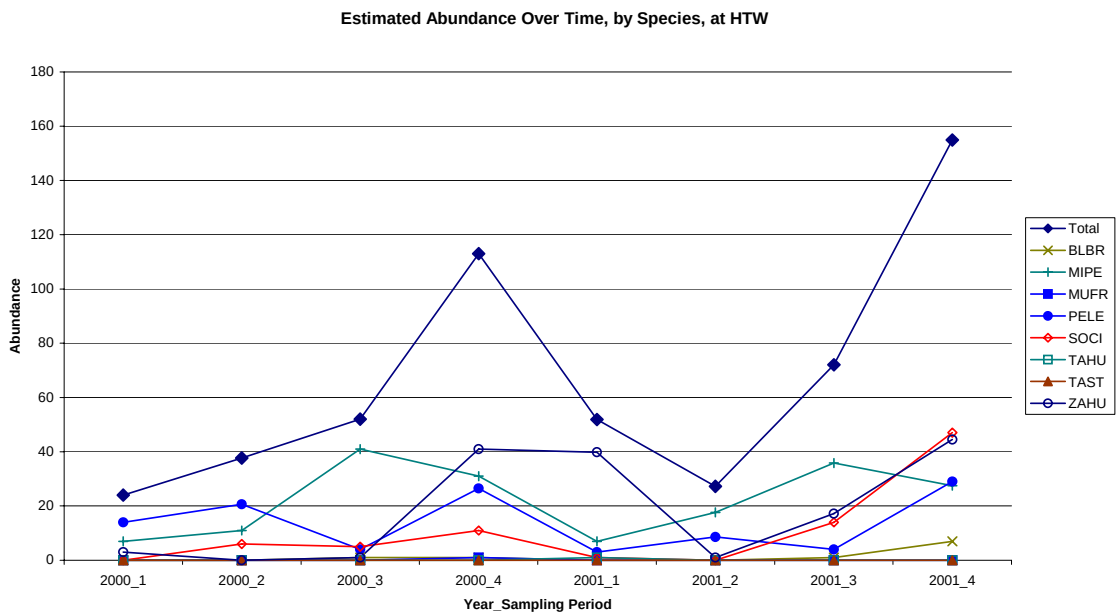


Figure 22. Temporal variation in estimated abundance of each species at High Toss Wetland over the course of sampling.

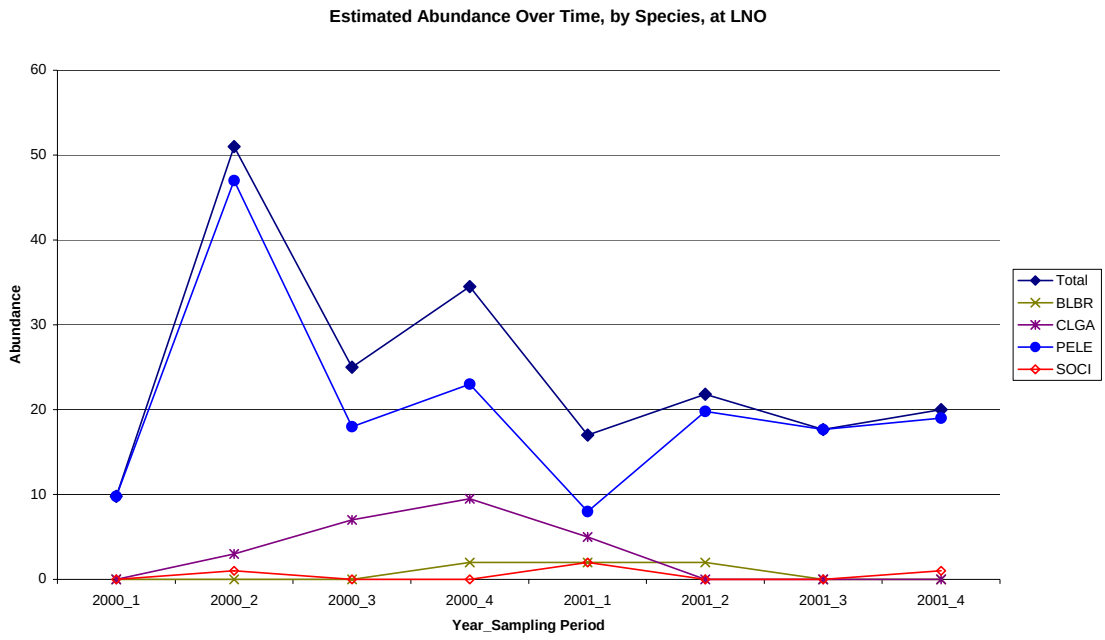


Figure 23. Temporal variation in estimated abundance of each species group at Longnook Oak Forest over the course of sampling.

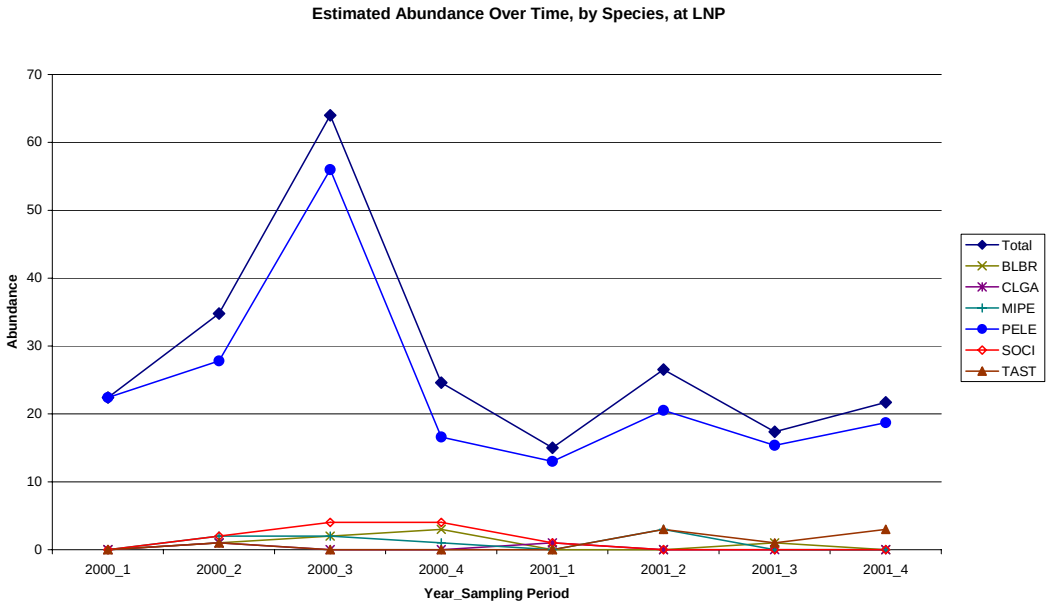


Figure 24. Temporal variation in estimated abundance of each species at Longnook Pine Forest over the course of sampling.

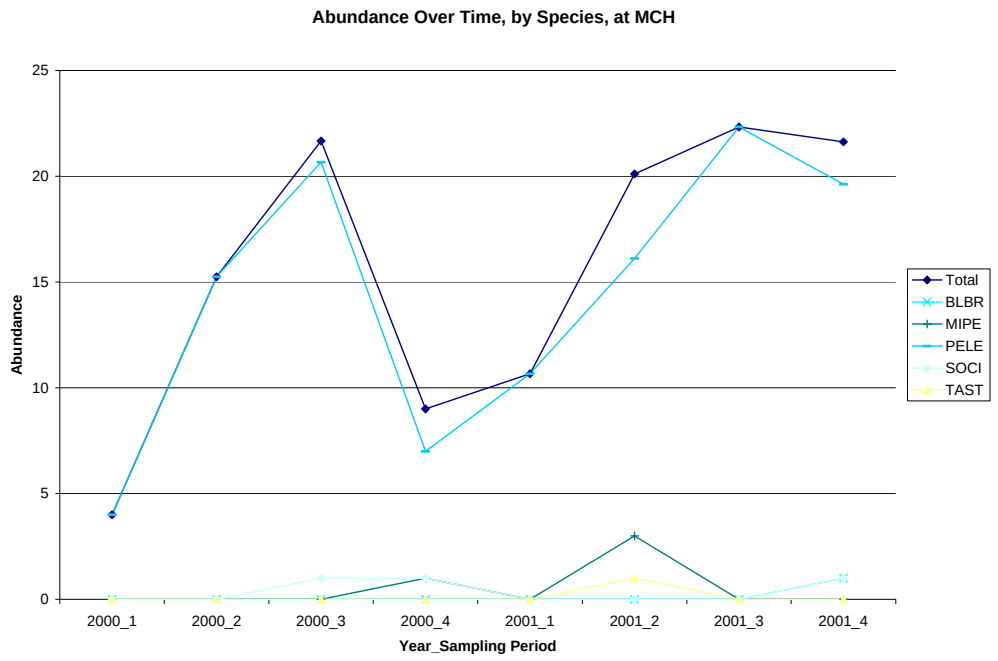


Figure 25. Temporal variation in estimated abundance of each species at Marconi Heathland over the course of sampling.

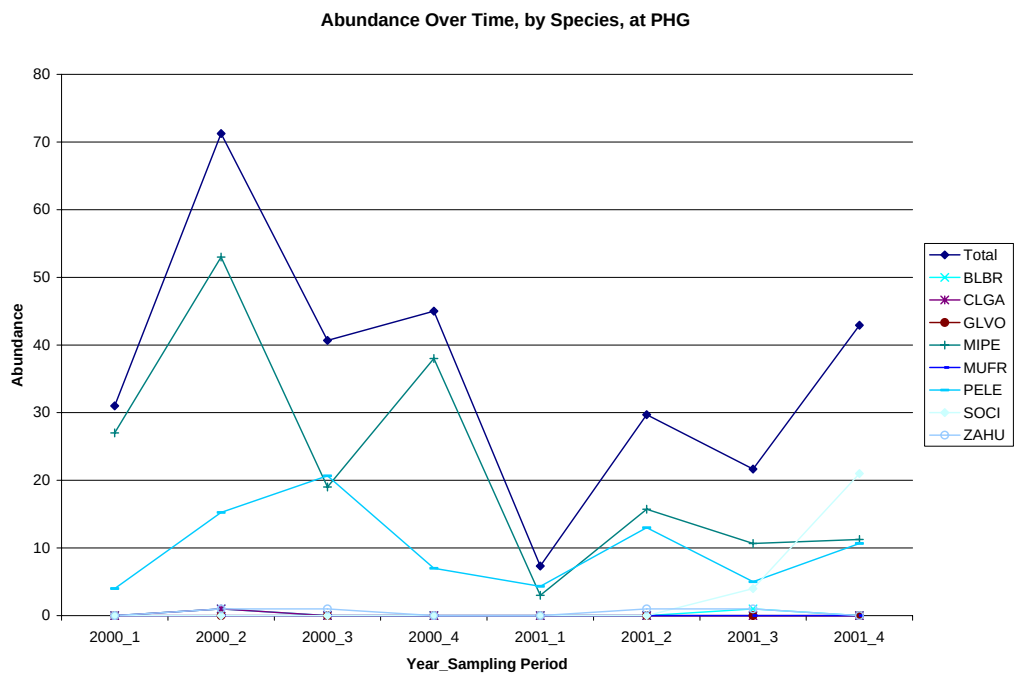


Figure 26. Temporal variation in estimated abundance of each species at Pilgrim Heights Grassland over the course of sampling.

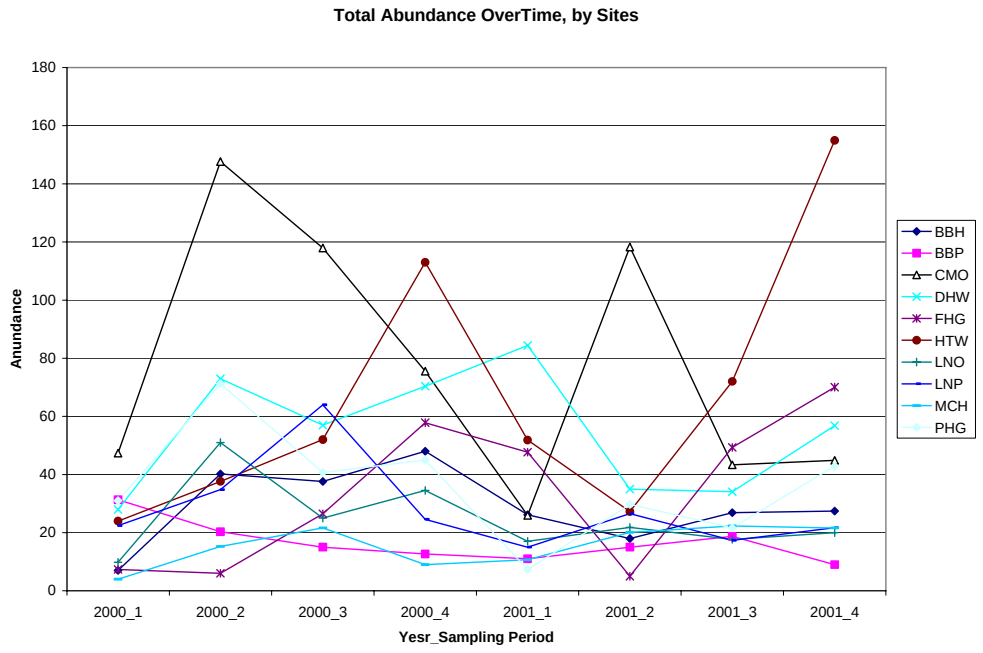


Figure 27. Estimated total abundance over time, by site, over all sampling periods.

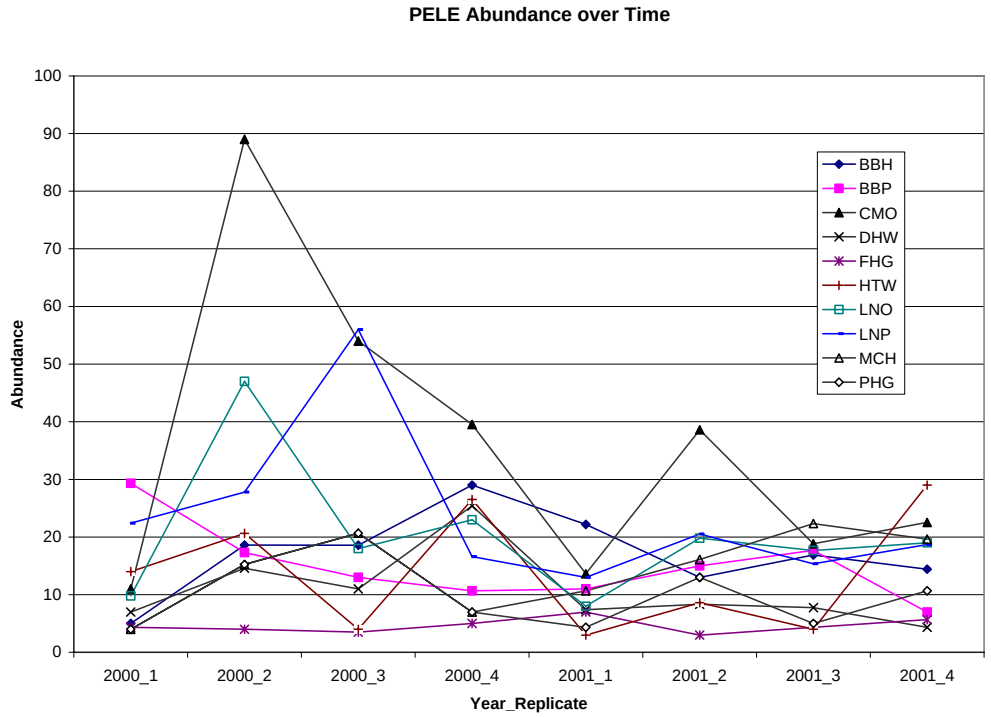


Figure 28. Estimated abundance over time, by site, over all sampling periods for *Peromyscus leucopus*.

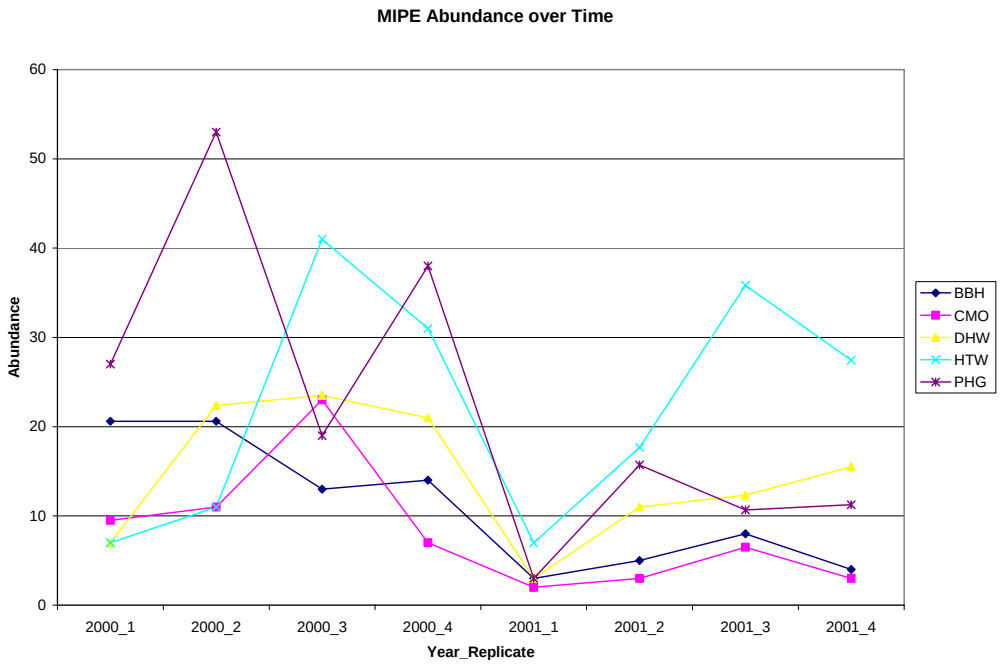


Figure 29. Estimated abundance over time, for all sites, over all sampling periods for *Microtus pennsylvanicus*.

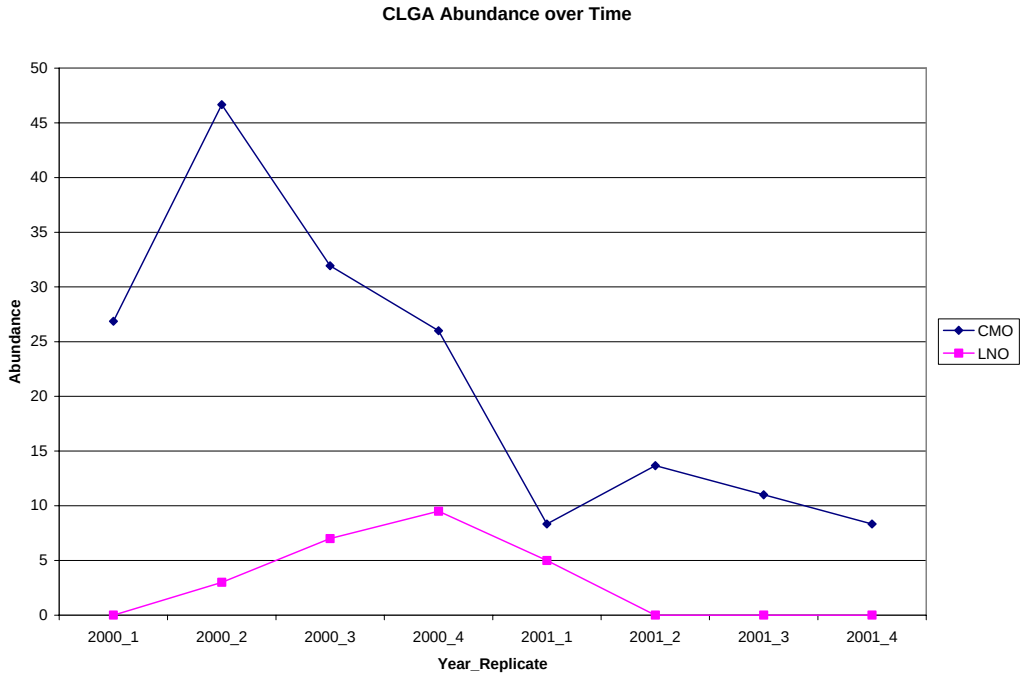


Figure 30. Estimated abundance over time, by site, over all sampling periods for *Clethrionomys gapperi*.

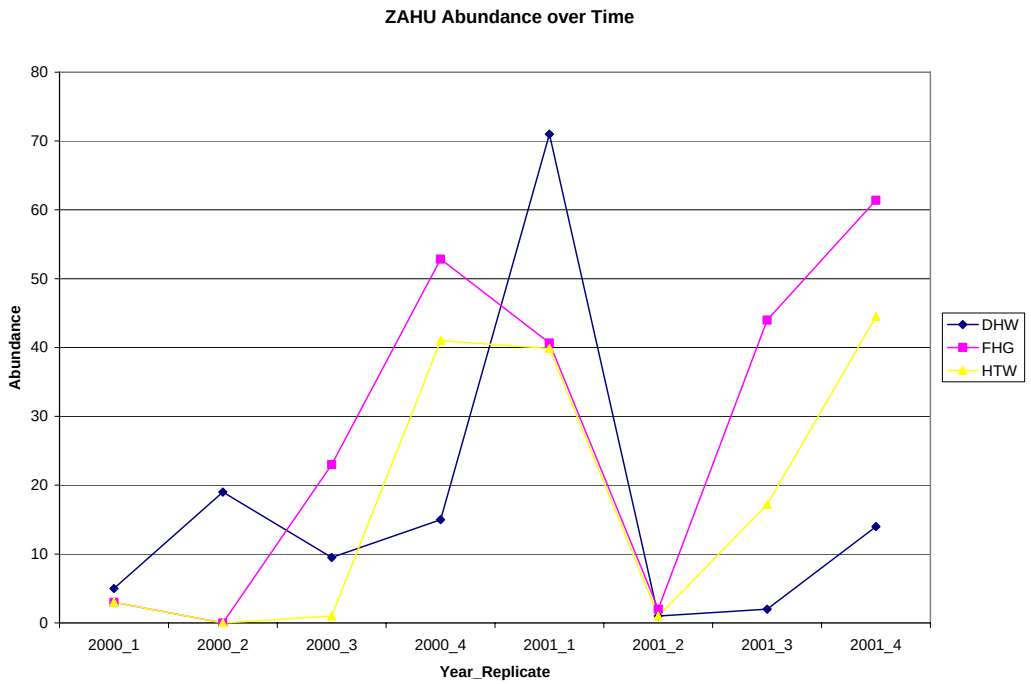


Figure 31. Estimated abundance over time, by site, over all sampling periods for *Zapus hudsonius*.

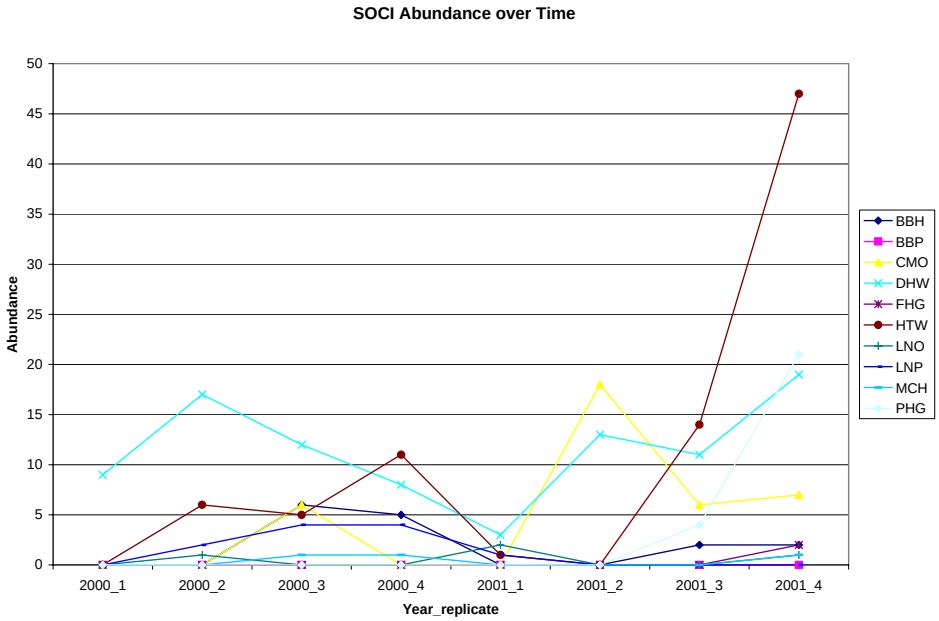


Figure 32. Estimated abundance over time, for all sites, over all sampling periods for *Sorex cinereus*.

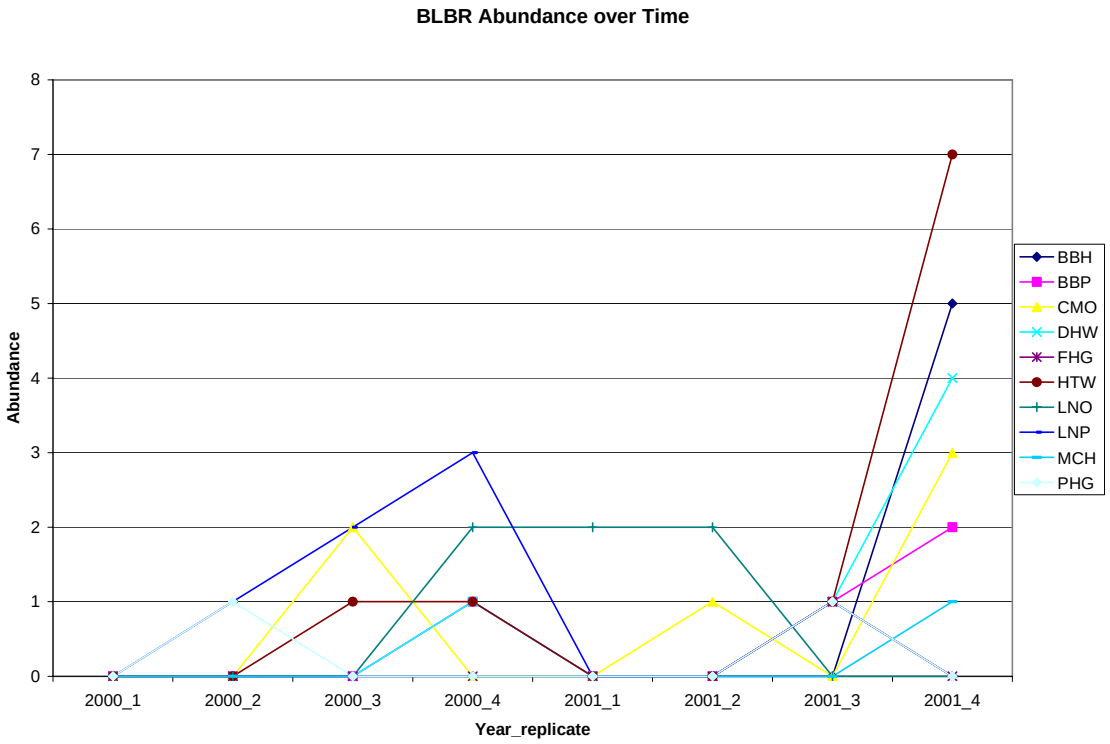


Figure 33. Estimated abundance over time, by site, over all sampling periods for *Blarina brevicauda*.

Seasonal Variation

Seasonal patterns in small mammal abundance have also been documented in many species. In the northern United States, the general pattern is one of population decline over the winter, with numbers building back up over the course of the summer and into autumn. For example, in southeast Massachusetts, MIPE numbers generally decline in late winter-early spring and increase through the summer. The related, insular species, *Microtus breweri* also shows a similar pattern (Tamarin et al. 1987). Massachusetts PELE also decline over the winter, with lows in the spring and peaks in autumn-early winter (Adler and Tamarin 1984, Adler et al. 1984). However, some Massachusetts populations of PELE and MIPE show this pattern more distinctly and regularly than others (Adler 1985). Moreover, differences in seasonal patterns between sites or over time sometimes reflect the influence of more random factors, such as severity of winter weather (Tamarin et al. 1987) or population irruptions due to mast crops (Wolff 1996), and the fact that these factors do not affect all sites or species equally. In the years following an irruption, a crashing population may instead show a decline from spring to fall (Wolff 1996).

The significant seasonal variation shown by most small mammals at CACO (Table 9, 11) follows to the general pattern of spring lows and late summer peaks (Figure 5). However, when viewed across two year's worth of seasons, the pattern is more variable, with the increase in 2000 occurring between late spring and early summer, whereas in 2001, the seasonal increase occurred between mid and late summer (Figure 6). Fluctuations in PELE, ZAHU, and SOCI appear to be driving most of this pattern (Figures 28, 31, 32).

While PELE followed the general increase from spring through late summer and decline over the winter, on a park-wide scale (Table 11, Figure 5, 6) as well as for most individual sites (Figure 28), their numbers peaked in early summer to mid summer. Krohne and Burgin (1990) also recorded peak numbers of Indiana PELE in early to mid summer. When viewed in detail (Figure 28), it is apparent that PELE at most sites fluctuates within a relatively small range. However, three sites, Cemetery Oak, Long Nook Oak, and Long Nook Pine, show much greater peaks in early to mid-summer, and appear to drive the park-wide seasonal pattern for PELE. Moreover, since PELE is the single most abundant small mammal species park-wide, the fluctuations of PELE at these three sites influences park-wide patterns of total abundance.

Interestingly, the three sites where PELE increased dramatically in early to mid summer and then declined by late summer are sites with a lot of oak, which is the preferred host plant of gypsy moth. PELE is a generalized insect predator (Ostfeld et al. 1996) that preys on gypsy moth larvae and pupae when available in spring and early summer (Elkinton 1996). Though Elkinton et al. (1996) speculate that gypsy moth numbers should have little or no impact on PELE abundance, the seasonal pattern observed at CMO, LNO, and LNP suggests that the PELE are responding to a seasonally available resource, which may be gypsy moth.

ZAHU showed significant seasonal variation, with peaks in abundance in late spring and late summer and lows in early and mid summer (Table 11, Figure 5). Yet while both 2000 and 2001 show the late summer peak, there is considerable difference between their abundance in spring 2000 v. 2001 (Figures 6, 31), and the spring peak shown in Figure 5 is the result of a peak in spring 2001. The seasonal variation in ZAHU abundance observed here contrasts with the

relatively stable numbers found by Nichols and Conley (1986) across an activity season in Michigan, but is generally consistent with variation observed in eastern Massachusetts ZAHU (Adler et al. 1986).

Variation in ZAHU abundance is complex, and involves both seasonal and spatial variability, and their interaction. ZAHU hibernate from ca. mid-autumn until mid-spring, with exact dates varying geographically (Godin 1977) and from year to year (Quimby 1951). This variation is presumably temperature driven and thus variation in winter severity, by influencing emergence, could influence abundance estimates in the spring. Mortality during hibernation occurs, and may be as high as 67% (Whitaker 1963). In addition, there is also post- and pre-hibernation movement. Nichols and Conley (1982) suspected that low spring survival may be due to ZAHU emigrating after emergence from hibernation. Quimby's (1951) belief that ZAHU migrated to drier habitats prior to hibernation is consistent with this. Quimby (1951) also attributed seasonal variation to moisture, with ZAHU avoiding flooded wetlands in the spring and moving into them as they dried out. Whereas for most small mammal species, seasonal trends primarily reflect population level at a site, ZAHU seasonal trends in abundance at a site appear to be heavily influenced by the movements of individuals in response to seasonal variation in wetland moisture (Quimby 1951, Townsend 1935, cited in Whitaker 1963).

Interpreting the seasonal variation in ZAHU abundance at CACO in light of these processes is difficult and explanations only partially satisfactory. Were the low numbers in spring 2000 relative to 2001 the result of delayed emergence due to severe winter weather in 2000? If so, since males emerge a couple of weeks ahead of females (Whitaker 1963, Quimby 1951) we might expect sex ratio differences between years. There were none (male:female =1.33 in 2000, 1.44 in 2001) nor were mean winter temperatures in 2000 lower than 2001 (mean 2000=38.53^{o} F, 2001=35.95^{o} F). Seasonal patterns at individual sites are also inconsistent. The pattern at Fort Hill Grassland, where the trapping grid was upslope from a wetland, suggests a process where animals emerge from hibernation, disperse down into the wetland, and then return prior to hibernation. The late summer increase at Duck Harbor Wetland and High Toss Wetland is consistent with the late summer movement of individuals into wetlands observed by Quimby (1951). Yet, it is hard to reconcile the abundance in these wetlands in the spring 2001 with this idea. If ZAHU move into these wetlands as they dry out in mid to late summer, wouldn't they be too wet in the spring?

While we are aware of the factors that contribute to seasonal variation in ZAHU abundance, there is more going on than our data can tell us. Understanding seasonal variation in ZAHU abundance at a given site would require data that are both more intensively collected over time, and over a more extensive area, so that seasonal shifts by animals across the landscape can be determined.

Seasonal variation in SOCI shows the general park-wide trend of increasing from a spring low through the course of the summer, and declining over the winter (Table 11, Figure 5). However, this trend is more pronounced in 2001 than 2000 (Figure 6), and at the site-specific level the pattern is far more variable (Figure 32). SOCI is a habitat generalist whose principal habitat requirement is moisture and varies greatly in abundance spatially and temporally (Whitaker and Hamilton 1998). The dramatic late summer 2001 increase in abundance at High Toss Wetland

could be the result of animals moving into the wetland as soil in the adjacent uplands dried out, making their invertebrate prey more difficult to obtain in the uplands. The spike in abundance at Pilgrim Heights Grassland in late summer 2001 is inexplicable, since this is a dry grassland, habitat SOCI generally avoids (Godin 1977).

Habitat Variation and Associations

The significantly greater total abundance of small mammals in wetland and oak forest (Table 9, 10) is the sum of individual species abundances. These reflect well established patterns of habitat preference. In addition, while no specific data on site moisture were collected, the ranked order of total abundance reflects a qualitative sense of relative site moisture. Site moisture may be acting on both specialist and generalist species to influence total abundance. In wetland, for example, the open wetlands supported an abundance of MIPE and ZAHU, both grassland specialists (Bellows et al. 2001), with ZAHU in particular being abundant in moist open habitats (Quimby 1951). In conjunction with SOCI, a widespread species also most abundant in moist sites where invertebrate prey are abundant (Whitaker and Hamilton 1998) and the ubiquitous PELE, these all add up to wetland having the greatest total abundance. In oak forest, the high abundance is primarily due to the abundance of a mast eating, woodland specialist CLGA (Whitaker and Hamilton 1998) and PELE. While PELE is considered a generalist, its greatest abundance is reached in woodlands (Adler et al 1984, Morris 1984) and the association between its abundance and oak mast is well established (Elkington 1996, Wolff 1996). Conversely, the habitats with lowest total abundance, heath and pine forest, are dry, lack oak mast, and have sparsely vegetated ground layers (Appendix 1).

In the analysis of variation in total abundance due to habitat variables, only principal component two, representing a gradient from open heath-like conditions to a denser ground cover of forbs and shrubs (Table 22) was included in model, though it was not significant (Table 23). While the habitat data collected are inconclusive statistically, the pattern of variation in total abundance (Figure 34) suggests that a dense ground cover of woody shrubs and forbs, in conjunction with moist soil and/or a mast producing species are important factors influencing total abundance. The significant differences in total abundance between habitats here at CACO contrasts with the findings of Bellows et al. (1999) on the coastal plain of Virginia. In addition, these patterns of abundance contrast with the postulation of Bennett (1998) that small mammal abundance at CACO is highest in grasslands and will decline as grasslands succeed into woodlands.

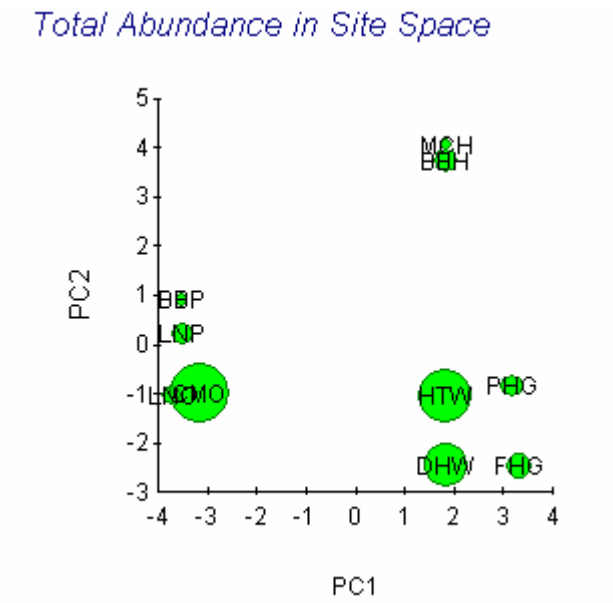


Figure 34. Total abundance of sites relative to their position in habitat space, as determined by principal components analysis of site habitat data. Site total abundance is expressed by size of circle.

On the species level, patterns of abundance (Table 10) reflect well known habitat affinities and are consistent with other work on Cape Cod small mammals (Spitzer 1977). PELE, though widespread, was most abundant in oak forest and least abundant in grassland. PELE is widely regarded as a habitat generalist (Bellows et al. 2001, Hoffman et al. 2001), that reaches it highest abundance in woody habitats (Adler 1984, 1988), particularly in deciduous forest (Dueser and Shugart 1978, Ostfeld et al. 1996). The significant negative association of PELE abundance with habitat principal component 1 (Table 23) reflects the positive association of PELE with woody dominated habitats (Adler 1987) and its lesser abundance in habitats lacking woody vegetation (Schweiger et al. 2000). The lower PELE abundance in habitats with highest MIPE abundance (wetland and grassland) is consistent with patterns observed in Virginia (Bellows et al. 2001), New York (Ostfeld et al. 1996) attributed to antagonistic displacement of PELE by MIPE, although work in Massachusetts (Adler 1984, 1987) found PELE abundance in adjacent woodlands to be most responsible for PELE abundance in grassland.

MIPE was also widespread, though significantly greater in wetland and grassland, with small numbers occurring outside these two habitats (Table 10). This is consistent with its known affinity for herbaceous-dominated habitats (Adler 1987, Bellows et al. 2001, Morris 1984) and consistent with the rapid decline in abundance of this habitat specialist outside its preferred habitat (Adler and Wilson 1989). While not statistically significant (Table 23), the positive relation to habitat principal component 1 (Table 22), reflects the increase in MIPE abundance along a grassland habitat gradient found by Adler and Wilson (1989) in eastern Massachusetts.

SOCI is a habitat generalist (Whitaker and Hamilton 1998) and its abundance does not appear related to specific habitat structure (Adler 1985). Its significantly greater abundance in wetlands at CACO (Table 10) is consistent with its preference for moist habitats (Whitaker and Hamilton 1998) and the findings of Spitzer (1977) on Outer Cape Cod. The significant negative relationship between SOCI abundance and habitat principal component 3 (Table 23), which represents a gradient from moist site to dry site vegetation (Table 22) is consistent with the avoidance of dry habitats by SOCI.

CLGA is predominantly a mast feeder of cool, moist woodlands (Whitaker and Hamilton 1998). Its preference for oak forest (Table 10) is consistent with this. Spitzer (1977) found CLGA in moist, deciduous woodlands of CACO and at Camp Edwards on Cape Cod, CLGA dominated pine-oak forest (Stevens and Cavanaugh 1997). The absence of CLGA in pine habitats in this study, we believe, reflects the dryness of the pine forest habitats sampled at CACO and is consistent with the strong association between CLGA abundance and moist sites with well developed leaf litter (Miller and Getz 1977). The near significant positive relationship between CLGA abundance and habitat principal component 4 (Table 23) reflects avoidance of open, grassy pine forest and preference for deciduous forest with well developed ground layer of forbs (Table 22).

ZAHU is a specialist of grassland, old field, and early successional stage habitats (Bellows et al. 2001a) with an affinity for moist sites (Bellows et al 2001b, Quimby 1951). Their near exclusive occurrence in wetlands and grasslands (Table 10) reflects this. While ZAHU abundance was comparable in the two wetland sites, in grasslands it was very rare in Pilgrim Heights Grassland (Table 12). While Whitaker (1963) found little correlation between ZAHU abundance and either

soil moisture or proximity to water, water was widely distributed across his study site. In contrast, Quimby (1951) found that ZAHU preferred moist, open habitats, and that proximity of water was important in habitat selection. The dryness of Pilgrim Heights Grassland, situated on the highest, flattest ground in the vicinity, and distance to nearest wetland, ca. 300 m, contrasts with the mid-slope location, with nearby wetlands at Fort Hill Grasslands. Thus, the rarity of ZAHU at Pilgrim Heights Grasslands is likely due to a lack of moisture and distance from water. The significant negative relationship with habitat principal component two, and positive relationship with component one (Table 23), reflects its avoidance of dry, heath-like sites and sites with well developed forest (Table 22), i.e. a preference for moist, open sites. Previous studies of outer Cape Cod small mammals also found ZAHU to specialize in moist, open habitats (Spitzer 1977).

GLVO is a mast eating, cavity nester whose distribution is determined by that of mast producing trees (Whitaker and Hamilton 1998). Its near exclusive occurrence in oak forest is consistent with this, though it was also recorded in pine forest as well (Table 10, 12). Spitzer (1977) and Stevens and Cavanaugh (1997) also found this species in deciduous and mixed forest on Cape Cod, as did Connor (1971) on Long Island, NY. However, Connor (1971) also considered them fairly common in pine barren too, in stands of large pitch pine. In this study, GLVO predominantly occurred at Cemetery Oak, not Long Nook Oak. Reasons for this are unclear, since the two sites appear to be very similar in habitat structure (Figure 8).

Conclusion Regarding Variation

The general patterns of variation in CACO small mammal abundance observed conform reasonably well to known patterns of seasonal variation and habitat preference. There are clearly many different factors influencing overall abundance, individual species abundance, and their patterns of variation. While our data and analysis do a reasonably good job of explaining variation at the general or coarse scale, it is also clear that many site and species specific questions regarding factors driving variation are unanswerable without more detailed and focused study.

Power Analysis

The ability of a monitoring program to detect trends within a statistically defined framework is known as its power (Gibbs 1995). In analyzing trends in abundance, the null hypothesis is that there is no trend. Power is the probability of correctly rejecting the null hypothesis, i.e. of concluding that there is a trend when one actually exists. In a program intended to monitor abundance, power is determined by inherent variability in abundance, sample size, number of spatial and temporal replicates within a year, annual frequency and duration of monitoring, as well as the statistical significance, and magnitude of the trend to be detected (Eagle et al. undated).

In conducting power analysis our goal was to determine the power of different monitoring scenarios built upon the protocol being tested (Bennett 1998) to detect trends of varying magnitude. While many of the factors that are incorporated into program MONITOR (Gibbs 1995) are determined in the course of running the simulations and are under our control, one

critical one is not. That is the inherent variability of the parameter being monitored, i.e. annual abundance. In program MONITOR, inherent variability is input as plot means and standard deviations to provide each plot's coefficient of variation (CV). Given that our data are relatively short term we were concerned that our estimations of variability (Table 25) might be low. Coefficients of variation over time for total abundance at CACO sampling sites ranged from a low of 0.42 at BBP to a high of 0.77 at FHG. These compare to CV in total abundance for long term data sets, 0.44 over 43 years (Fryxell et al. 1998) and 0.82 over an 11 year period (Grant et al. 1976). Similarly, our CV for PELE at each site ranged from 0.28 at FHG to 0.77 at HTW. Data from Fryxell et al. (1998) provided a 43 year CV for PELE abundance of 0.49 and data from Grant et al. (1976) for 11 years produced a CV of 1.01. Thus the pilot data used for our power analysis simulations appear reasonable.

There are no universal standards regarding acceptable power. This depends on the goals of a monitoring program and the nature of what is being monitored. However, both Eagle et al. (undated) and Gibbs (1995) suggest that 90% power to detect 3% annual change is a desirable standard of power in wildlife monitoring. The results of the power analyses we conducted (Tables 13-19), using the abundance estimates collected in two years of protocol testing indicate that the full protocol (10 sites x four temporal replicates) is suitable for monitoring total small mammal abundance at the park-wide level (Table 13a). Over the long term (20-25 years), the power to detect a 2% annual decline is the same for a sampling interval of 5 years as it is for annual sampling. Thus, while annual monitoring would be expected to provide greater power in the short term, for a long term monitoring program, monitoring every 3-5 years will still provide data with the power to detect trends as small as ± 2%/year. As a reference, a decline of 2%/year adds up to a decline of 32% over a 20 year period. Moreover, even a park-wide monitoring program based solely on a late summer replicate is suitable in terms of power to detect annual trends of ±2%/year or greater (Table 13b).

While the protocol tested appears very powerful for monitoring total abundance at the park-wide scale (10 sample plots), it is, predictably, less powerful for monitoring trends at a single site with only one plot. Annual monitoring at the least variable sites provided acceptable power to detect trends of ± 2-3%/year, based on either four replicates or a single late summer replicate (Table 14a, 15a). However, the high power indicated for less frequent sampling in late summer (replicate 4) at the least variable site (Table 15a) should be viewed with caution, since it is based on such a small CV. At the most variable individual sites, the tested protocol lacks acceptable power (Table 14b, 15b). While we did not conduct analysis of all possible scenarios, a monitoring program intended to detect changes at a single specific site in the park would need additional spatial replication, i.e. additional sampling plots.

Table 25. Coefficients of variation in estimated abundance based on all 4 replicates and Replicate 4 only. ***denotes locations where species was not captured

a. CV based on all four replicates for 2000 and 2001.

	Species					
Sites	**Total Abundance**	**PELE**	**ZAHU**	**CLGA**	**MIPE**	**SOCI**
BBH	0.45	0.41	2.83	***	.75	1.29
MCH	0.45	0.46	***	***	2.14	13.80
BBP	0.42	0.45	***	***	***	***
LNP	0.55	0.58	***	1.85	1.20	1.29
LNO	0.52	0.59	***	1.23	***	1.51
CMO	0.58	0.73	***	0.63	0.84	1.35
PHG	0.52	0.61	1.07	2.82	0.74	2.35
FHG	0.77	0.28	0.87	***	2.83	2.83
HTW	0.69	0.77	1.09	***	0.60	1.49
DHW	0.38	0.62	1.33	***	0.52	0.44

b. CV based on Replicate 4 only for 2000 and 2001.

	Species					
Sites	**Total Abundance**	**PELE**	**ZAHU**	**CLGA**	**MIPE**	**SOCI**
BBH	0.39	0.47	1.41	***	0.79	0.61
MCH	0.58	0.67	***	***	1.41	0.00
BBP	0.24	0.29	***	***	***	***
LNP	0.09	0.08	***	***	1.41	1.41
LNO	0.38	0.13	***	1.41	***	1.41
CMO	0.36	0.39	***	0.73	0.57	1.41
PHG	0.03	0.29	***	***	0.77	1.41
FHG	0.13	0.09	0.11	***	1.41	1.41
HTW	0.22	0.06	0.06	***	0.09	0.88
DHW	0.15	1.00	0.05	***	0.21	0.58

The power of this protocol to monitor changes in the abundance of particular species is variable. For our most widespread and abundant species, PELE, it has very good power to detect changes at the park-wide scale (Table 16). The results here are very similar to those of total abundance (Table 13). Sampling every 3-5 years, with four temporal replicates appears to have adequate power at the park-wide scale. For a single temporal sample, annual or biennial sampling is necessary (Table 16b). For single site monitoring of PELE, only the least variable site could be monitored adequately (Tables 17, 18). For species other than PELE however, abundance and corresponding CV's were more variable (Table 25). For ZAHU, for example, even when based only on the sites at which it was recorded, the power was inadequate (Table 19).

Based on this analysis, the protocol tested appears able to adequately monitor long term trends in total abundance on a park-wide scale, even with sampling every three to five years. However, the power analysis indicates its ability to make meaningful statements regarding trends in abundance at the site specific level would be limited to only those sites with low variability. Similarly, with the exception of PELE, this protocol in its present form would not be adequate for monitoring trends in individual species. Where trends analysis at individual sites or for the less common species is needed, a more customized monitoring program, with more spatial replication at each site would be necessary.

While this analysis, based on our pilot data, suggests that sampling park-wide, once very five years in late summer is adequate to detect long term declines of -2% (Table 13b), due to the short term nature of the pilot data, we would recommend a more conservative approach at the outset of any monitoring program. Either the number of temporal samples or the frequency of sampling should be increased. Two scenarios of essentially equal power seem feasible. The first is to sample four times over the course of the season, once every 3-5 years. The second is to sample every year in late summer. The first scenario, with intensive but infrequent sampling would work best in an operation where a sampling crew samples a different park each year, or shifts focus between taxa each year. This allows for monitoring of multiple parks or multiple taxa. The disadvantage of this approach however, is that there may be a loss of data regarding annual variation. In the second scenario, annual sampling in late summer allows for sampling at the time of year when small mammal density tends to be greatest, and will provide insight into annual variation. Operationally, this scenario would work well when piggy-backed onto the tail end of a spring-summer field season that samples other taxa earlier in the year (e.g breeding amphibians, breeding birds).

Community Analysis

Small mammal communities in the eastern United States consist of species that vary in terms of their degree of habitat specialization or selection. Habitat selection in temperate zone small mammals appears to be the result of innate species differences in habitat preference (Morris 1984). Habitats form gradients of suitability for each species, related primarily to habitat structure, but also reflecting availability of food, predation, competition, and parasitism (Adler 1988) and thus, habitat structure ultimately determines community composition by influencing the occurrence and abundance of particular species. Moreover, because the factors determining habitat suitability for a given species vary spatially, even within a seemingly homogeneous

habitat type (e.g. see Krohne and Burgin 1990), and temporally (Adler 1988), small mammal community composition will also be variable.

The species composition of CACO small mammal communities consist of species considered to be generalists, such as PELE, SOCI, BLBR, and more specialized species such as MIPE, ZAHU, CLGA, TAST, and GLVO (Adler 1988, Adler and Wilson, 1989, Bellows 2001, Dueser and Shugart 1978,1979, Kitchings and Levy 1981,Miller and Getz 1976, Morris 1984). While there are some apparent inconsistencies, and finer details vary due to site-specific differences, the general pattern from these studies is that woodland small mammal communities in the northeast U.S. are dominated by the generalist PELE, with lesser and varying numbers of woodland specialists CLGA, TAST, and GLVO, plus habitat generalists SOCI and BLBR. Grassland, old field, and meadows are dominated by specialists MIPE and ZAHU, with lesser numbers of generalists PELE, BLBR, and SOCI.

The composition of small mammal communities at CACO is generally consistent with these patterns, though we believe that site-specific moisture played an important role in influencing finer level details. Regardless of whether they are considered a generalist or a specialist with regard to habitat structure, the occurrence and abundance of ZAHU, CLGA, BLBR, and SOCI appears related to site moisture (Miller and Getz 1977, Whitaker and Hamilton 1998, Quimby 1951). Since PELE is relatively evenly distributed across habitats and sites (in spite of it being most abundant in oak forest), it is the distribution and abundance of these other species that drives most of the observed differences in species composition (Tables 4, 10, 11). The species composition in grassland and wetlands differs from the species composition in other habitats due to the abundance of MIPE, ZAHU, and SOCI and oak and pine forest differ primarily due to the influence of CLGA and CLVO. Bellow et al. (2001) found a similar situation in coastal plain Virginia, with community differences in species composition between forested and old field habitats due primarily to the addition of old field specialists rather than due to differences in PELE.

ANOSIM provides some seemingly contradictory results. In the nested analysis, which reflects the *a priori* habitat categories of each site, the differences between sites nested within habitats were sufficiently large that they drove the analysis. Yet, when sites are pooled into habitat types, or sites are analyzed without being assigned to a habitat type, there are significant differences in community similarity between habitats and sites. Whereas sites are real entities, habitat types are categories representing a range of conditions. Sites classified into the same habitat category are not necessarily identical. The location of habitats on Figure 8 is simply the average of the site-habitat pairs on Figure 7. Morris (1984) also found differences in abundance and species composition between site-within-habitat type replicates and concluded that sites, in reality, graded into each other, and that habitat boundaries were arbitrary and indistinct. Thus sites are more continuous in nature whereas habitat types categorical.

This appears to be the case at CACO as well. Essentially, community similarity at either site or habitat level is organized along a gradient from woody dominated to open herbaceous dominated sites or habitats (Figures 7, 8). This is seen in the decline in abundance/importance of PELE as you move along this gradient, and the corresponding increase in MIPE and ZAHU. This is also

seen in the significant relationship determined by RELATE between the spatial organization of sites based on habitat variables and on mammal species composition.

Patterns of species diversity (Figure 35) indicate that gross differences in habitat structure (e.g. forested vs grassland) had little impact on diversity. While regression of species diversity on habitat PCA scores was non significant, the inclusion of PCA3 and PCA2, each with a negative loading, suggests a negative relationship between species diversity and dry, bare, heathland habitats. The significant relationship between total abundance and species diversity also suggest that site moisture, by attracting species with higher inherent moisture requirements or attracted to the greater availability of invertebrate prey, plays a significant role in both abundance and species diversity.

Species composition of small mammal communities at Cape Cod National Seashore are essentially the same as those found elsewhere on Cape Cod (Adler 1988, Stevens and Cavanaugh 1997). However, relative abundance of species differs. Compared to other sites studied in the Cape Cod region, masked shrew and meadow jumping mouse were more abundant, and short-tailed shrew and red-backed vole were less abundant at Cape Cod National Seashore.

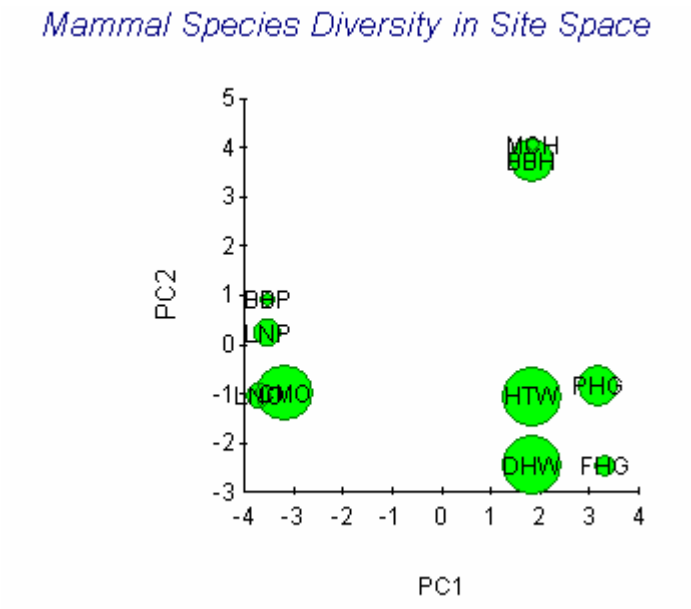


Figure 35. Species diversity at sites relative to their location in habitat space as determined by principal components analysis. Site diversity is expressed by size of circle.

Adequacy of Species Detection

Several different trap types may be used to capture small mammals and, by virtue of different spatial arrangements that may be employed, there are a multitude of specific ways to inventory and monitor small mammals. Each trap type and spatial arrangement has biases in terms of species detection and inherent suitability for detecting species presence versus estimation of abundance and density (Jones et al. 1996). Consequently, species detected and estimated relative abundance may reflect choice of trap type and their spatial arrangement rather than the true abundance of each species. While the use of population estimators rather than indices accounts for differences in detection probabilities, there are still issues associated with detecting presence, especially when a method is marginal in its ability to detect a particular species and/or a species is rare. For example, while small rodents and large shrews such as BLBR are readily caught in both the Sherman traps used in this work and pitfall traps, smaller shrews and moles are most effectively caught in pitfall traps (Jones et al. 1996, Gartshore 1988, Laakkonen et al. 2003, Williams and Braun 1983). Thus there is always uncertainty regarding failure to detect species that were actually present.

Though the Sherman traps used in this work may have a bias against capture of small shrews such as SOCI, and their high trap mortality compared to rodents may result in lower population estimates, SOCI comprised 13.8% (252/1829) of all individuals captured. This indicates that at least in terms of detecting presence of SOCI, trapping methods were adequate. In contrast, Eastern moles (*Scalopus aquaticus*) occur at CACO and, based on presence of mole burrows at the surface, appear to be widespread and common. They were not detected in this survey. Adler (1988) detected them in extremely small numbers (2 of 3333 captures) using live traps on Nantucket. Clearly, the methods used here at CACO provide no useful information on moles. Other information available on the small mammal fauna of Lower Cape Cod (Prescott 1994, Spitzer 1976) indicates that the only other potentially present but undetected species of small mammal at CACO is short-tailed weasel (*Mustela erminea*). This species' occurrence on Lower Cape Cod is based on only a few records (Spitzer 1976, Prescott, pers. comm). Recent work at CACO, targeting meso-mammals, also detected the long tailed weasel (*Mustela frenata*) but not the short-tailed weasel (O'Connell et al. in press).

Thus, with one known exception and one possible exception, all the species of small mammals known to occur on Lower Cape Cod were detected during this inventory.

Conclusions/Recommendations

CACO Mammals

Over the course of sampling at 10 sites representing five habitats, over two year growing seasons (May through September), 1829 individuals representing 11 species were captured. In order of relative abundance (based on overall mean estimated abundance) they were: white-footed mouse, *Peromyscus leucopus*, 43.36%; meadow vole, *Microtus pennsylvanicus*, 20.24%; meadow jumping mouse, *Zapus hudsonius*, 16.87%; masked shrew, *Sorex cinereus*, 8.92%; southern red-backed vole, *Clethrionomys gapperi*, 6.54%; southern flying squirrel, *Glaucomys volans*, 1.80%; short-tailed shrew, *Blarina brevicauda*, 1.57 %; chipmunk, *Tamias striatus*, 0.55%,; long-tailed weasel, *Mustela frenata*, 0.10%; red squirrel, *Tamiasciurus hudsonicus*, 0.03%; and eastern cottontail, *Sylvilagus floridanus*, 0.03%. Annual variation in abundance was not significant, except for *P. leucopus, M. pennsylvaticus, and C. gapperi*. Seasonal and habitat differences in abundance were mostly significant, except for uncommon species where sample sizes were small. Seasonal abundance generally followed the well documented pattern of decline over the winter, followed by increase over the course of spring to late summer. Based on our analysis of variation in abundance and species composition, and habitat variables we found:

Species composition of small mammal communities at Cape Cod National Seashore are essentially the same as those found elsewhere on Cape Cod. However, relative abundance of species differs. Compared to other sites studied in the Cape Cod region, masked shrew and meadow jumping mouse were more abundant, and short-tailed shrew and red-backed vole were less abundant at Cape Cod National Seashore.

Small mammal abundance is greatest in woodland and wetland habitats and lowest in grassland and heath. Similar patterns of abundance have been found elsewhere on Cape Cod.

Distribution of species among habitats is fairly consistent with known habitat affinities for these species in general, and on Cape Cod in particular. Certain species are widespread, but vary in abundance between habitats, probably as a result of food habits and site moisture. The granivorous white-footed mouse appears to be most abundant in woody-dominated habitats and least abundant in herbaceous-dominated ones. For the herbivorous meadow vole, the pattern is reversed. Other species appear to show stronger habitat affinities, such as masked shrew with wetlands, meadow jumping mouse with moist, herbaceous habitat, and red-backed vole with oak forest. This latter was surprising, given the red-backed vole's known association with coniferous habitats, and may be due to the xeric nature of most pine habitats on Cape Cod.

Estimation of Abundance

Considering the sample sizes obtained and the performance of multiple mark and recapture models for estimation of population size, we recommend use of Chapman's modification of the Lincoln-Petersen Index. This can be applied to a weekly sampling schedule such that trap checks on Tuesday and Wednesday constituted sampling period one (marking session) and Thursday and Friday constitute period two (recapture session). This method will provide valid estimates of abundance, and will be less time consuming both in the field and in the office. It will also be more economical in terms of materials.

Estimation of Survival

Considering the small sample sizes we obtained over the course of an entire sampling season, and the lack of model fit, we do not recommend survival estimation. While there are specific situations where estimation of survival is important to compare trends or differences between sites, or in response to a management action or other variables of interest, for a general long term monitoring program based on this protocol it is not practical.

Monitoring Program

For CACO, a program of annual park-wide sampling in late summer is recommended. This will provide the power to monitor trends in total abundance at the park-wide scale. While its ability to monitor specific sites or the less common species is less, there are no rare or endangered species here, nor is there currently a specific site where pre and post management- action monitoring is called for. In either of these cases, a different monitoring program would be called for. Thus the primary purpose of this monitoring program would be to track park-wide trends in abundance and occurrence, and relate them to long term trends in habitat succession. Through annual monitoring, greater insight into annual variation and its underlying causes will be possible. Not only is this desirable biologically, but it would also produce more credible estimates of annual variability and allow for a more robust power analysis after several years. From an operational perspective, annual sampling would also ensure a continuity of expertise, institutional commitment, and equipment-readiness. In addition to collecting data on small mammals, a small mammal monitoring program should collect data on habitat variables, or be linked to a vegetation monitoring program.

While these recommendations for CACO may be applicable to other sites, it should not be taken for granted that they will work for all parks under all circumstances. As previously discussed, the different types and sizes of mammal traps, the different trap layouts (e.g. various grid size and spacings v. trapping webs), different lengths of a sampling period, variation in the number of spatial and temporal replicates in a sampling period, and frequency of sampling over the long term present an almost infinite number of possibilities in the details of a monitoring program. Choice of a monitoring protocol will depend on the question(s) that monitoring is being used to answer and the parameters of interest. For example, where unbiased density estimation is critical, trapping webs appear to be superior to grids (Parmenter et al. 2003). Moreover, differences between sites in the abundance/density or inherent variability of small mammals may also mean that a protocol that provides adequate power for detecting trends or estimating survival at one park may be excessive or inadequate at another. Based on the experience gained in protocol testing at CACO, small mammal monitoring at other parks need to carefully consider the monitoring questions and design a program capable of answering them. The CACO protocol can serve as a starting point, but adjustments to fit the specifics of monitoring questions as well as biological differences will likely be needed.

Critical review of the protocol tested at CACO raised the following suggestions:

If captures during a Monday to Friday sampling period are too low, consider extending the sampling period for a longer period.
Similarly, a larger grid size/number of traps can also be used to increase number of captures, or two traps per station can also be used.
Ideally, monitoring questions should be stated as specifically as possible as working hypotheses, so that a protocol to answer the specific question can be developed.
While power analysis indicated that sampling every 3-5 years provided adequate power for trends detection, there was broad concern that sampling at this frequency will be challenging institutionally. Such infrequent sampling makes it difficult to maintain continuity of staff expertise, equipment readiness, and institutional commitment.

Finally, as with all monitoring protocols, critical review is necessary after an initial period of implementation. With the recommended program of annual sampling in late summer, the question of temporal variability and power analysis should be revisited after five years.

Part Two: Small Mammal Monitoring Protocol

Introduction

Based on field testing conducted in 2000 and 2001, and subsequent analysis of that data, small mammal monitoring will be conducted through a program of live trapping. As detailed below, 10 sites, representing five habitat types will be sampled once annually, in late summer. Data will be used to estimate abundance, community composition, and determine species distribution and habitat relations. Data will be used to track trends in these over time and interpret them relative to successional changes in vegetation.

Safety Considerations

There are four main areas of safety concern related to conducting the field work associated with this protocol. These are; sun and heat (heat stress), tick borne diseases, poison ivy, and hanta virus. Biting flies and mosquitoes may also be present at times and constitute both a nuisance and a possible disease vector.

All staff conducting small mammal monitoring will receive training and implement the measures established in the CACO Inventory and Monitoring Program's programmatic SOP# P01-Monitoring Project Safety. Special attention should be paid to the sections of this SOP relating to tick bites and Lyme disease, heat stress, and poison ivy. In addition, the following measures should be undertaken:

- Field staff should wear light-colored, long pants and shirts to protect against sun, poison ivy, biting insects, and to make it easier to spot ticks. The park will provide light colored coveralls if requested. Tuck pant legs into socks and wear sturdy boots.
- Staff are encouraged to apply insect repellent provided to their clothes/coveralls.
- Staff are encouraged to perform periodic tick checks while in the field, as well as thorough tick checks after returning from the field.
- Staff sensitive to poison ivy should make use of pre-contact solution (such as Ivy Block) and post-contact wash (such as Technu). The project manager will ensure that this is supplied.
- Staff must review NPS Hanta-Virus Worker Protection Recommendations (http://www.nps.gov/public_health/zed/hanta/hanta_worker_pro.htm) and CDC Guidelines for biologists trapping small mammals (http://www.cdc.gov/ncidod/dvrd/spb/mnpages/rodentmanual.htm).
- All workers checking, handling, or transporting traps that have been in use must be wearing half-face air-purifying respirators (N-100 Filter Type) and disposable rubber gloves. In addition, gardening gloves for small rodents and leather gloves for large rodents and carnivores should be worn.
- When checking or otherwise handling traps, hold trap(s) on downwind side.
- Transport traps in the back of an open vehicle.
- Disinfect traps at the end of each weekly sampling session using 5% Lysol solution.

Field Methods

Site Selection

As detailed in Part One, the five major upland and bottomland habitat types present at CACO, heathland, wetland, pine forest, oak forest, and grassland, will be monitored. Two 0.81 ha sites per habitat were randomly chosen for a total of ten sample sites (Figure 1, Table 1).

Sampling Season

Annual sampling will occur in late summer, from late-August through late-September. Sampling will begin the Monday of the last full week in August. This generally falls from 8/20 to 8/26 in any given year. Two sites will be sampled each week, with the complete round of sampling at the 10 sites taking a total of five weeks. The two sites sampled in any given weekly period will represent different habitat types. Pilgrim Heights Grassland and Cemetery Oak will be sampled in week one, Long Nook Pine and Long Nook Oak in week two, Fort Hill Grassland and Marconi Heath in week three, Bound Brook Pine and High Toss Wetland in week four, and Bound Brook Heath and Duck Harbor wetland in week five. This schedule and order of sampling should be consistent from year to year, with only slight variation in the dates.

Sampling Sessions

Each site's weekly sampling session will span a five day period, Monday to Friday. Traps will be set beginning Monday morning and left open for four consecutive nights. Traps will be checked each morning, starting Tuesday and removed on Friday.

Trapping Grids

Delineating and flagging the trapping grid at all sites must be done prior to the beginning of small mammal trapping season, i.e. prior to setting the traps. One hundred Sherman traps (model LFATGD) will be set on a 90m by 90m grid at each site with one trap placed every 10 meters. The grid will be oriented in a north-south direction with the first trap set in the southwest corner of the plot and labeled 1A. The traps (grid points) to the north of that point are sequentially numbered up to ten, (2A, 3A, 4A, et.) and the traps (grid points) to the east are alphabetically labeled up to trap 'J' making a number-letter combination for each trap station (Figure 36). Each trap station is marked with a 36" surveyor flag and labeled with a paint pen for ease of locating it

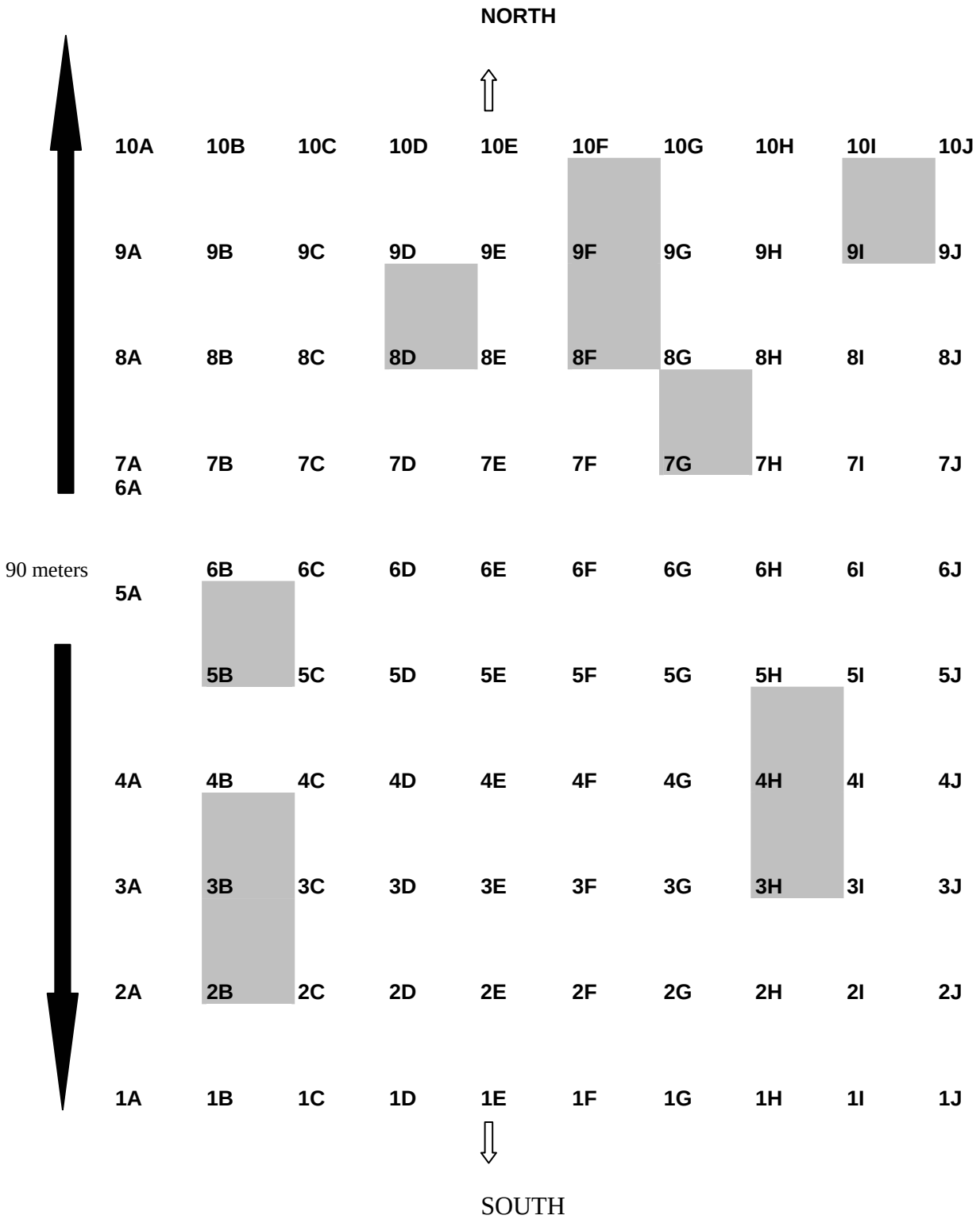


Figure 36. Mammal trapping grid layout. Shaded areas represent vegetation sampling plots.

once the vegetation has filled in. At sites with dense vegetation, additional flagging tape may be needed to mark trapping points. Trapping stations will be consistent for the duration of small mammal monitoring at Cape Cod National Seashore. Supplies and materials needed to re-establish and mark trapping grids are detailed in Appendix 3.

Setting Traps

Supplies and materials needed to trap, process and mark small mammals are detailed in Appendix 3. All these materials will need to be obtained prior to starting field work. Due to the high cost of traps and potential loss to vandals or predators, the 200 traps will be moved weekly between sites. They will not be left in place when not in use. Trapping will occur Monday through Friday, with traps placed and set on Monday and pulled after the Friday check.

One Sherman trap will be set at each station in the most optimal location for small mammal capture within one meter of the flag. Optimal trap sites are parallel to logs, in shaded areas, in runways, and along the edges of vegetation. The traps will be initially set on Monday morning. Set the trap on the ground, scraping an area free of vegetation if necessary to find a flat surface. Confirm that the entrance to the trap is clear and kept away from branches that may accidentally trigger the trap. In wet areas, make sure the trap is not set in an area that could be flooded by precipitation. To minimize overheating of the metal traps, cover the top of the trap with leaves or grass. While overheating was not a serious issue at CACO, parks in warmer climate should consider wooden covers to protect traps from overheating.

Bait should be prepared in large batches weekly prior to small mammal trapping. At least 200 bait packets should be brought into the field each day of trapping. Traps are baited with a mixture of one 4 pound jar of peanut butter, 1.5 pounds of oats, and one package chopped, cooked bacon with its grease. Mix ingredients together and make into bait packets by taking one teaspoon and twisting it into a small three-inch square piece of paper towel, like taffy. To reduce bait stealing or accidental triggering of the trap, hang one bait packet from the inside of the back of each trap by cinching the loose tip of the paper towel in the rear trap door.

Bedding material is supplied to keep captured animals warm and occupied while spending the night in a trap. Compressed cotton pads known as “nestlets” are used. Place one nestlet inside each trap. Reuse nestlets if they are in good condition and dry, even if used. Adjust the sensitivity of the trap door so that by tapping lightly on the treadle, the door springs shut. The door will not shut if the tension is too high and may not capture shrews. If the tension is too low, the door will be overly sensitive and too easily triggered.

Daily Trap Checks

All workers checking traps must be wearing half-face air-purifying respirators (N-100 Filter Type) and disposable rubber gloves. In addition, gardening gloves for small rodents and leather gloves for large rodents and carnivores should be worn.

Traps will be checked once per day in the morning from Tuesday through Friday. Trap checks should begin at 0600 hours to minimize convective heating of the traps. There should be

minimal trap mortality because most of the species are nocturnal and will enter the traps during the nighttime hours. Check sites with the least vegetative cover (such as grasslands and heathlands) first, because those traps are more susceptible to heating and trap mortality.
Upon arrival at site, record starting time, date, marking period, marking day, and collectors on the field data sheet (Table 26). Take the air temperature in the shade near the first trap station, and use a calendar to record the phase of the moon (full, first quarter, new, last quarter). Record the sky and wind codes (Table 27). Prior evening precipitation can be noted in the event notes box. If the site appears to have been visited by a predator, recognized by trap disturbance, write this in the event notes field of the data sheet.

Begin at one corner of the trapping grid and walk along trap lines, one researcher per line. Stay close to coworker in case of captures, so that captures can be processed as a team. Recheck bait and trap sensitivity every day in case of missing bait or dysfunctional trap. If a trap door is closed and trap is empty, missing or flipped over, reset the trap in the correct location and keep a running tally of disturbed traps. Record the total number of empty disturbed traps at that site at the top of the data sheet (Table 27). Replace the trap if it has been damaged past the point of functionality (always keep spare traps in vehicle used for field work).

Table 26. Small mammal field data sheet.

SITE_____ **Air Temp (°C)**_____ **Start Time**_____ **Event Notes:**

Sky Code_____ **End Time**_____

DATE_____ **Prev night precipitation Y N** **Moon Phase** full first last new **#Disturbed traps**_____

Soil Moisture ___ ___ ___ ___ ___ **Collectors**_____

Marking Period 1 (red) or 2 (blue)
Marking Day 1 2 3 4 (circle appropriate)

X	Y	Previous Mark (R,B,RB,none)*	New Mark (R,B,RB,none)*	Species	Sex	Reproductive Condition	Age	Weight (g)	Full Body (mm)	Tail (mm)	Rear Foot (mm)	Comments

* R=red mark, B=blue mark, RB=both red and blue mark, none=no new mark given. "Previous Mark" is mark that is on an animal at time of capture, "New Mark" is mark given to animal on current day of capture unless it already has mark from that *same marking period*, in that case write "none".

Table 27. Sky codes.

Code	Description
0	Few Clouds
1	Partly Cloudy/Variable
2	Cloudy or overcast
4	Fog or smoke
5	Drizzle or light rain
7	Snow
8	Showers

Marking Animals

As will be detailed in the section on Estimating Abundance, the four days that traps are checked and animals captured will be considered two separate sampling periods. Tuesday and Wednesday are sampling period one, and Thursday and Friday are sampling period two. The purpose of marking will be to mark the animals captured in a way that allows a count of the number of individuals captured in period 1 (n_1), the number of individuals captured in period 2 (n_2), and the number of individuals captured and marked in period 1 that are also captured in period 2 (m_2).

Animals captured in period 1 (Tuesday and Wednesday) will be given a red "x" with a red permanent marking pen on their ventral surface, just under the chin. Animals captured in period 2 (Thursday and Friday) will be marked with a blue "x" in the center of the abdomen. For either mark, put a small line of the same color on the top of the head as a backup.

Processing Captures

When an animal is captured, work in teams of two to expedite the process of handling and marking. On the field datasheet (Table 26), record the trap station (e.g. 2A, 6J, etc.). Identify the species in the trap and determine the proper handling bag to use. To identify species, first, go by weight. Weasels and squirrels are much heavier than mice and shrews. Also, weasels have a strong musk. If the trap is light, tip the trap vertically and carefully open the top door, looking down to view the animal. Be sure that your free hand is slightly covering the entrance in case the animal tries to escape, as is common. Record the four-letter species code in the "Species" column of the field datasheet. Captured animals need to be transferred to a handling bag for processing. For smaller species (everything but squirrels and weasels) use a clear plastic handing bag. The bags used by pet stores to transport fish work well. For squirrels and weasels, use a cloth mesh bag ("laundry bag") to avoid animals tearing through the bag.

To process small mammals: Place the handling bag over the mouth of the trap, push open door and tip the animal into the bag. Immediately close the bag and drop the trap.

Identify if the animal has been captured previously by checking for marks on the ventral surface of the animal. Record "none" if there is no previous mark and "R", "B" or "RB" if there is already a mark on the animal. "R"=red mark only, "B"=blue mark only, and "RB"=red and blue marks. Record this information in the "Previous Mark" field of the datasheet.

Weigh the animal in the bag using a Pesola scale. As a general rule, for weasels and squirrels use a 100g or 500g scale, for PELE, ZAHU, CLGA, BLBR and MIPE use a 30g or 60g scale, and for SOCI and juveniles use a 10g scale. Subtract the weight of the empty bag from the total weight of the bag plus animal to get the actual weight of the animal in grams. Never clip the animal directly to the scale unless it is dead.

To remove the animal from the bag, through the bag, hold onto the base of the tail with one hand, and peel the bag back around the animal. Immediately pinch the scruff of the animal with a free hand and release the tail and bag. Process weasels directly in the handling bag to avoid probable escape.

Once the animal is out of the bag, mark it immediately with the appropriate permanent-marking pen in case it manages to escape while taking morphological data. On Tuesday and Wednesday (period 1) use a red marker. Mark the animals on the ventral surface just under the chin. On Thursday and Friday (period 2) use a blue permanent marker and make an X in the center of the abdomen. Whenever using either mark, put a small line of the same color on the top of the head as a backup mark. Record the new mark given on the datasheet under "New Mark" field using the marking scheme in #2 above. If an animal has already been marked with the color for the current trapping period, record "none" in the "New Mark" field. Specifically, on day 2, an animal that already has a red mark will be recorded as "R" in the "Previous Mark" column and "none" in the "New Mark". On day 4, an animal captured that already has both red and blue is recorded as "RB" in the "Previously Marked" column and "none" in the "New Mark". An animal captured with only a blue mark, will be recorded as "B" in the "Previous Mark" column and "none" in the "New Mark".

After marking, record sex, age (adult, subadult, juvenile), and reproductive condition. For reproductive condition note if testes are scrotal or abdominal in males and note if females have swollen nipples or, if obvious, they are pregnant. Write "unknown" if no determination can be made.

The following guidelines for aging different species are meant as a guide. Large juveniles can sometimes weigh more than small adults and there is sexual dimorphism to consider as well. As the researcher encounters many animals, the difference of adult versus juvenile becomes clearer.

Aging of species is as follows:
PELE juveniles are completely grey, subadults still have some grey as well as some brown and a distinct stripe on their back, adults have no grey and have a distinct stripe on their back
MIPE adults are usually over 30g, juveniles are under 25g and there is some overlap
CLGA adults are over 16g, juveniles below 16g with some overlap
SOCI adults are around 2.5-3.0g, it is unlikely to capture a juvenile SOCI
ZAHU adults are vividly colored reddish brown adults weighing in the mid-teens and up, juveniles are paler colored and noticeably smaller under 13g
BLBR adults are usually over 10g, juveniles are rarely captured and will be noticeably smaller than adults
GLVO adults are usually over 45g, juveniles are below 45g with some overlap
TAST adults are over 66g, juveniles are under 66g with some overlap

Measure each animal captured with a flexible ruler. Measure the hind foot length from nail to heel, body length from tip of nose to tail tip, and tail length from anus to tip of tail. Make note in the "comments" field if the tail has been damaged.

In the comments field: If a female has a litter in the trap, make note of number of young, and place them next to the trap in nesting material. Often the mother will retrieve her young once left alone.

Record number and type of parasites including mites, botflies and ticks.

Record if animal escaped without receiving a mark for that day.

For trap mortalities, record all data as described above and place animal in a resealable bag with a tag that contains all of the biological data. Write "DEAD" in comments. Store these specimens in a freezer to be made into study skins. If the dead animal is not used by the next field season, discard it.

After double-checking that all data fields are complete, release the animal next to the trap station it was trapped.

Re-set trap. Replace bait and check condition of nestlet. Replace nestlet if wet or otherwise unusable.

If animal runs back into the trap following its release and the trap's resetting, release it before leaving the grid that morning.

Soil Moisture

Soil moisture will be recorded at each site, at each trapping session, at the ten grid points corresponding to those used for vegetation sampling (see below). These are 2B, 3B, 3H, 4H, 5B, 7G, 8D, 8F, 9F, and 9I. Record soil moisture on Monday, while setting traps, using a "Kelway Soil pH and Moisture Meter". Insert the "Kelway" meter into the ground to the point where the top of the metal electrodes are flush with the surface.

Data Entry

After returning from the field, paper data sheets will be filed in a "to be entered" file folder. Once this data is entered in the Access database on the computer (within 24 hours), it will be put in a file folder labeled "to be proofed". After the data has been entered and proofed, the original paper datasheets will be photocopied and stored in a separate location from the originals. The original, updated version of the Access database will be entered on the hard drive of the computer and on Friday of each week, upon completion of entry, the most up to date copy will be backed up on the Y:/I&M Projects/Small Mammals/Data/Current Data/File Name. The naming convention for the database will be 'Mammal Data Entry_YEAR_MONTH_DAY', with the month being spelled out (e.g. Sept instead of "9") to avoid any confusion. The date on the original entry database on the hard drive will be updated every day that it is modified or data is entered. At the end of the season, after all entry and proofing is done, the file name should be followed with the word "FINAL" and saved on both the hard drive and the Y-drive and copied to a compact disc which will be put in a folder with the original datasheets.

Data Analysis

Tabulating Raw Data

To estimate abundance of each species at each site, for a trapping session, the following must be derived from the data.

n1 = individuals marked in period 1 (sum of "R" in "New Mark" column)
n2 = individuals captured in period 2 (sum of "B" plus sum of RB in "New Mark" column)
m2 =individuals captured in period 2 that were marked in period 1 (sum of "RB" in "New Mark" column)

Estimating Abundance

Abundance will be estimated for each species at each site. To estimate population size (N), Chapman's modified Lincoln-Petersen will be used.

$$\boldsymbol{N} = \frac{((n_1 + 1)(n_2 + 1))}{(m_2 + 1)} - 1.$$

Variance and 95% confidence intervals can also be estimated. Formulas can be found in Nichols and Dickman (1996) or Thompson et al. (1998).

Estimate the population size for individual species sampled for each site. Since shrews are rarely recaptured, total number of individuals captured is calculated for their group. This is done by summing all values of "none" in the "Previously Marked" column of the data for a week long sample at a given site.

In addition to estimating abundance for each species, an estimate of total abundance (all species), all shrews, and all rodents should be determined. This should be done by summing the estimate for each individual species for the particular time and place.

Trends Analysis

Short term trends (i.e. between two points in time) or between two sites can be analyzed with a z test according to formulas presented in Thompson et al. (1998). First, for each population estimate, its variance must first be estimated:

$$\text{Variance}(\check{N}) = \frac{(n_1 + 1)(n_2 + 1)(n_1 - m_2)(n_2 - m_2)}{(m_2+1)^2 (m_2+2)}.$$

This is then log transformed as

$$\text{Var}[\ln(\check{N})] = \frac{\text{Variance}(\check{N})}{\check{N}^2}.$$

After the log transformed variance for both sites or points in time have been estimated, construct the *z* test to determine if there is significant difference between two populations estimates $\check{N}_1$ and $\check{N}_2$ as follows.

$$Z = \frac{\ln(\check{N}_1) - \ln(\check{N}_2)}{\sqrt{\mathrm{Var}[\ln(\check{N}_1)] + \mathrm{Var}[\ln(\check{N}_2)]}}.$$

The probability (*p*) of obtaining such a value when there is no difference in population estimates can be determined from a table of critical vales of the *t* distribution, with degrees of freedom = ∞ (Zar 1996). If *p* is less than 0.05, we would reject the null hypothesis that differences in the population estimates was due to sampling variation and conclude they are different.

Longer term trends in the abundance of a particular species or species group should be analyzed using regression techniques, described in further detail in Thompson et al. (1998). Essentially, abundance will be the dependant variable and time the independent variable. In regression analysis of trends, the null hypothesis is that there is no trend, and that, therefore, the slope of the line is zero. The testing is to determine if the actual slope deviates significantly from zero.

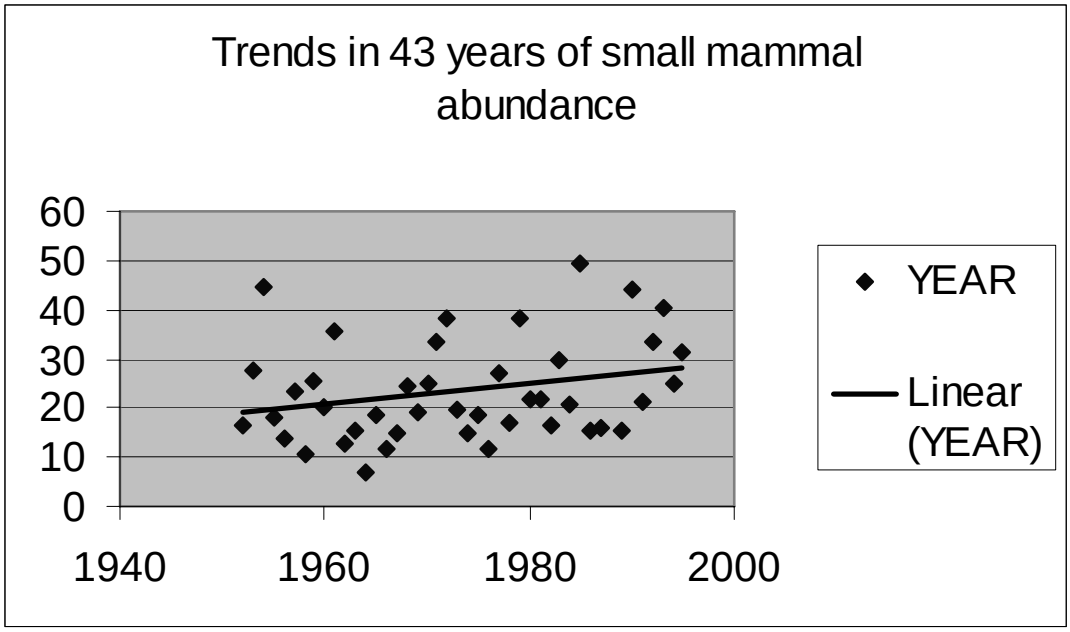
STATISTICA or other statistical packages can be used for regression analysis. Below is a regression analysis of the total abundance of small mammals reported by Fryxell (1998) from Ontario. This was done in Excel.

Regression Statistics	
Multiple R	0.265748374
R Square	0.070622198
Adjusted R Square	0.047954447
Standard Error	9.955870605
Observations	43

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regression	1	308.8099469	308.8099	3.115536	0.084996787
Residual	41	4063.89374	99.11936		
Total	42	4372.703687			

	Coefficients	*Standard Error*	*t Stat*	*P-value*
Intercept	-394.6837408	236.8433902	-1.66643	0.10325
X Variable	0.211863345	0.120029896	1.765088	0.084997



In this analysis, the slope of the line or regression coefficient is 0.211, indicating that the trend over the 43 year period is positive. However, the *p* value for the regression, 0.084 indicates that there is an 8.4% chance that this is due to sampling variation rather than a real trend. Thus, these data do not show a trend significant at the $p<0.05$ level.

Vegetation Sampling

To detect changes in the vegetation community due to annual variation or long term succession, as well as determine the relationship of the small mammal community to the vegetation, sampling of vegetation at each site will be done. Vegetation sampling will be conducted during the two weeks following the last week of small mammal trapping.

The existing trapping grid will be used for vegetation sampling, which will consist of ten 10 m x 10 m vegetation sampling plots. Grid points 2B, 3B, 3H, 4H, 5B, 7G, 8D, 8F, 9F, and 9I were randomly chosen and constitute the southwest corner each 10 m x 10 m plot (Figure 36). Within the 10 m x 10 m plot, basal area of trees, and percent cover of overstory (vegetation > 5 m) and understory (woody vegetation 1-5 m high) components will be estimated to the nearest 5% increments . Within each plot, groundcover will be sampled four times, once in each corner of the 10 by 10 meter vegetation plot, with a 1 x 1 meter square quadrat.

Many of the estimates will be of percent cover of a given species or vegetation type (Table 2). Prior to this work, practice estimating percent cover, referencing Figure 37. To determine percent cover, first locate the vegetation that you will be estimating in the plot, then determine how much of the entire plot is covered by that vegetation rounded to the nearest five percent. Diameter at breast height of trees will be taken by measuring the tree 1.5 meters above the ground by wrapping DBH tape around the tree at that height. Litter/duff layer will be measured with a metric ruler from the bottom of the duff layer to the top of the litter layer.

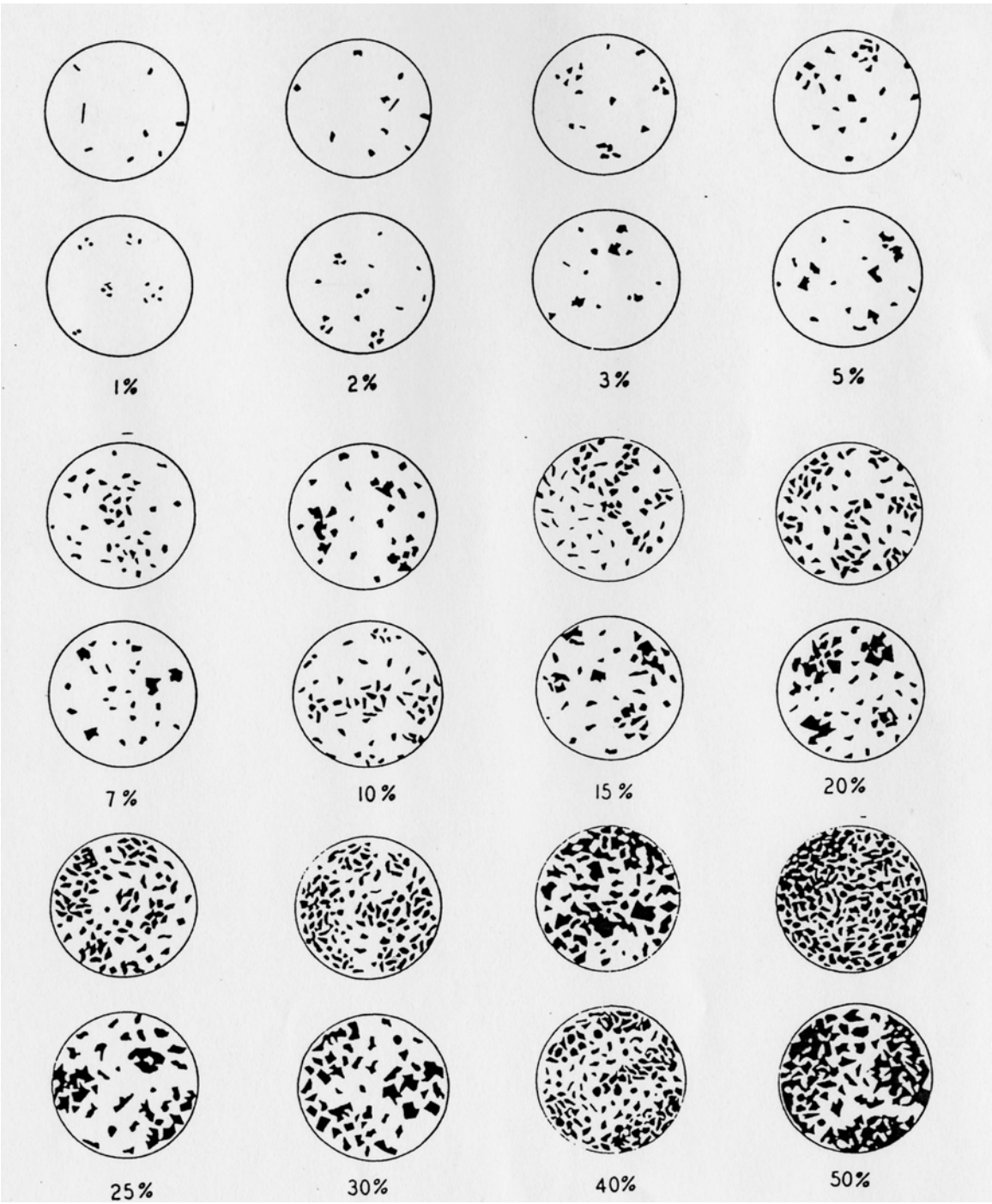


Figure 37. Percent cover visual aid.

One datasheet will be used for recording the overstory, understory, and DBH for a given plot (Table 28). Another datasheet will be used for recording data from the four groundcover subplots and duff layer measurements (Table 29). Hence each small mammal sampling site will have a total of 20 datasheets, two per vegetation sampling plot.

Basal Area Sampling

DBH will be recorded for all trees dead or alive with a minimum DBH of 3 centimeters. Each tree will have its own line on the datasheet. For each tree encountered record species and DBH. If tree is dead, record in column "Dead?"

Overstory Sampling

Sample all trees over five meters tall. For each plot, record total percent tree canopy cover, total conifer cover, total deciduous cover and percent cover of each individual species on the datasheet (Table 28).

Table 28. Overstory, understory, and tree sampling datasheet. In each plot, estimate cover clases to nearest 5%.

Date	Site Name	Plot Number	Vegetation Type

Overstory

Vegetation Type/Species	Percent Cover
Total overstory cover	
Total deciduous cover	
Total coniferous cover	

Understory

Vegetation Type/Species	Percent Cover
Total understory cover	
Total deciduous cover	
Total coniferous cover	

Tree DBH

Tree Species	DBH	Dead?

Table 29. Groundcover vegetation sampling datasheet.

Date	Site Name	Vegetation Plot Number	Vegetation Type

In subplots, estimate cover class to nearest 5%, *check (✓) if species is present in entire veg plot

genus/species	Subplot 1	Subplot 2	Subplot 3	Subplot 4	Plot presence*
Ground herbaceous					
Ground graminoid					
Ground forb					
Ground woody shrub					
Ground woody groundcover					
Ground fern					
Ground moss					
Ground lichen					
Ground bare ground					
Ground coarse woody debris					
Ground leaf litter					
Duff/litter depth					

Understory Sampling

Sample all woody vegetation between one to five meters high. Estimate percent cover of the entire shrub understory layer, not including species that are herbaceous even if over 1m tall (e.g. goldenrod, canary grass). Record the percent cover of all conifer species combined, all deciduous species combined, and the percent cover of each individual understory species.

Groundcover Sampling

Sample all vegetation that is less than one meter tall as well as all herbaceous vegetation, regardless of height. Record percent cover of graminoids, forbs, all herbs (graminoids *and* forbs), woody groundcover (e.g. *Arctostaphylos, Corema*), woody shrubs, ferns, mosses, lichens, bare ground, coarse woody debris, and leaf litter. In addition, record the presence of any additional species that occur outside the 1 x 1-meter subplots but within the 10 by 10-meter plot.

Litter/duff Sampling

The depth of the litter and duff will be taken in the same place and at the same time as the groundcover samples are taken. Take one measurement per subplot in the corner closest to the center of the larger 10 by 10 meter grid.

Use of Vegetation Data in Habitat Analysis

Habitat data for each of the ten mammal sampling sites will be summarized to provide site specific values for all of the variables detailed in Table 2. However, for understory and ground cover variables, percentage cover will be used rather than the proxy values used in initial protocol testing. Each variable should be tested for normality and square root transformed if raw data deviate from normal. Percentage data should be arc sine transformed (Zar 1996). Data may then be subjected to principal components analysis (PCA) to simplify the variables. The role of habitat Principal Components in abundance and diversity can then be analysed using multiple regression. Principal Component scores are the independent variables and abundance or diversity is treated as the dependant variable.

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Appendix 1. Values of mammal habitat variables used in Principal Components Analysis. Basal area is expressed as cm^2 and represents the sum of basal areas for the trees of interest recorded in the five vegetation sampling plots within a site's trapping grid.

plots	OvTreeStem	BA Total	MeanBA/Tree	BA Dead	BA Live	BA Decid	BA Conif	UndTot%Cov	Und%Decid	Und%Conif
BBH	10	402.7	40.3	0.0	402.7	58.9	343.8	15.2	4.4	10.7
BBP	42	13694.3	326.1	735.5	12958.8	631.1	13063.2	31.6	31.6	0.8
CMO	51	11354.2	222.6	403.5	10950.8	5495.8	5858.5	12.8	12.8	0
DHW	0	0.0	0.0	0.0	0.0	0.0	0.0	36.1	36.1	0
FHG	0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0	0
HTW	0	0.0	0.0	0.0	0.0	0.0	0.0	7.6	7.6	0
LNO	57	9711.2	170.4	1306.2	8405.0	9711.2	0.0	43.5	43.5	0
LNP	40	12686.4	317.2	849.4	11837.0	0.0	12686.4	38.5	38.5	0
MCH	11	880.0	80.0	0.0	880.0	0.0	880.0	19.1	0	19.1
PHG	0	0.0	0.0	0.0	0.0	0.0	0.0	10.7	8.3	3.8

plots	GrndHerb>3	GrndGram>3	GrndForb>3	GrndWshrb>3	GrndWgrnd>3	GrndFern>3	GrndMoss>3	GrndLic>3	GrndBare>3	GrndCWD>3	GrndLLit>3
BBH	2	2	0	0	6	0	0	1	2	0	0
BBP	3	3	0	1	0	0	0	0	0	0	5
CMO	6	1	5	10	0	0	0	0	0	2	4
DHW	9	2	7	10	0	2	0	0	0	0	0
FHG	14	7	7	14	0	0	0	0	0	0	0
HTW	8	1	7	0	0	0	0	0	0	0	2
LNO	5	0	5	8	0	0	0	0	0	1	5
LNP	5	5	0	3	0	0	0	0	0	1	3
MCH	6	3	3	2	6	0	1	1	1	0	3
PHG	14	9	5	4	0	0	0	0	1	0	0

Appendix 2. Population estimates for all species present (total abundance), total rodents, total shrews, and each individual species, for all sites and trapping sessions.

Year	Site	Season	All	Rodent	Shrew	BLBR	CLGA	GLVO	MIPE	MUFR	PELE	SOCI	TAHU	TAST	ZAHU
2000	BBH	1	7	7	0	0	0	0	2	0	5	0	0	0	0
2000	BBH	2	41	41	0	0	0	0	21	0	19	0	0	1	0
2000	BBH	3	38	32	6	0	0	0	13	0	19	6	0	0	0
2000	BBH	4	48	43	5	0	0	0	14	0	29	5	0	0	0
2000	BBP	1	30	30	0	0	0	0	0	0	29	0	0	1	0
2000	BBP	2	20	20	0	0	0	1	0	0	17	0	0	2	0
2000	BBP	3	15	15	0	0	0	0	0	0	13	0	0	2	0
2000	BBP	4	13	12	1	1	0	1	0	0	11	0	0	0	0
2000	CMO	1	48	48	0	0	27	0	10	0	11	0	0	0	0
2000	CMO	2	148	148	0	0	47	1	11	0	89	0	0	0	0
2000	CMO	3	118	110	8	2	32	1	23	0	54	6	0	0	0
2000	CMO	4	76	76	0	0	26	3	7	0	40	0	0	0	0
2000	DHW	1	28	19	9	0	0	0	7	0	7	9	0	0	5
2000	DHW	2	73	56	17	0	0	0	22	0	15	17	0	0	19
2000	DHW	3	58	45	12	0	0	0	24	1	11	12	0	0	10
2000	DHW	4	70	61	9	1	0	0	21	0	25	8	0	0	15
2000	FHG	1	7	7	0	0	0	0	0	0	4	0	0	0	3
2000	FHG	2	5	4	0	0	0	0	0	1	4	0	0	0	0
2000	FHG	3	27	27	0	0	0	0	0	0	4	0	0	0	23
2000	FHG	4	58	58	0	0	0	0	0	0	5	0	0	0	53
2000	HTW	1	24	24	0	0	0	0	7	0	14	0	0	0	3
2000	HTW	2	38	32	6	0	0	0	11	0	21	6	0	0	0
2000	HTW	3	52	46	6	1	0	0	41	0	4	5	0	0	1
2000	HTW	4	112	99	12	1	0	0	31	1	27	11	0	0	41
2000	LNO	1	10	10	0	0	0	0	0	0	10	0	0	0	0
2000	LNO	2	51	50	1	0	3	0	0	0	47	1	0	0	0
2000	LNO	3	25	25	0	0	7	0	0	0	18	0	0	0	0
2000	LNO	4	35	33	2	2	10	0	0	0	23	0	0	0	0
2000	LNP	1	22	22	0	0	0	0	0	0	22	0	0	0	0
2000	LNP	2	35	32	3	1	1	0	2	0	28	2	0	1	0
2000	LNP	3	64	58	6	2	0	0	2	0	56	4	0	0	0
2000	LNP	4	25	18	7	3	0	0	1	0	17	4	0	0	0
2000	MCH	1	4	4	0	0	0	0	0	0	4	0	0	0	0
2000	MCH	2	15	15	0	0	0	0	0	0	15	0	0	0	0
2000	MCH	3	22	21	1	0	0	0	0	0	21	1	0	0	0
2000	MCH	4	9	8	1	0	0	0	1	0	7	1	0	0	0
2000	PHG	1	31	31	0	0	0	0	27	0	4	0	0	0	0
2000	PHG	2	71	70	1	1	1	0	53	0	15	0	0	0	1
2000	PHG	3	41	41	0	0	0	0	19	0	21	0	0	0	1
2000	PHG	4	45	45	0	0	0	0	38	0	7	0	0	0	0

Appendix 2: Population estimates for all species present (total abundance), total rodents, total shrews, and each individual species, for all sites and trapping sessions (continued).

Year	Site	Season	All	Rodent	Shrew	BLBR	CLGA	GLVO	MIPE	MUFR	PELE	SOCI	TAHU	TAST	ZAHU
2001	BBH	1	26	26	0	0	0	0	3	0	22	0	0	1	0
2001	BBH	2	18	18	0	0	0	0	5	0	13	0	0	0	0
2001	BBH	3	27	25	2	0	0	0	8	0	17	2	0	0	0
2001	BBH	4	27	20	7	5	0	0	4	0	14	2	0	1	1
2001	BBP	1	11	11	0	0	0	0	0	0	11	0	0	0	0
2001	BBP	2	15	15	0	0	0	0	0	0	15	0	0	0	0
2001	BBP	3	19	18	1	1	0	0	0	0	18	0	0	0	0
2001	BBP	4	9	7	2	2	0	0	0	0	7	0	0	0	0
2001	CMO	1	26	26	0	0	8	2	2	0	14	0	0	0	0
2001	CMO	2	119	100	19	1	14	44	3	0	39	18	0	0	0
2001	CMO	3	44	38	6	0	11	1	7	0	19	6	0	0	0
2001	CMO	4	45	35	10	3	8	1	3	0	23	7	0	0	0
2001	DHW	1	84	81	3	0	0	0	3	0	7	3	0	0	71
2001	DHW	2	34	20	13	0	0	0	11	1	8	13	0	0	1
2001	DHW	3	34	22	12	1	0	0	12	0	8	11	0	0	2
2001	DHW	4	57	34	23	4	0	0	16	0	4	19	0	0	14
2001	FHG	1	48	48	0	0	0	0	0	0	7	0	0	0	41
2001	FHG	2	5	5	0	0	0	0	0	0	3	0	0	0	2
2001	FHG	3	49	48	1	1	0	0	0	0	4	0	0	0	44
2001	FHG	4	70	68	2	0	0	0	1	0	6	2	0	0	61
2001	HTW	1	52	51	1	0	0	0	7	0	3	1	1	0	40
2001	HTW	2	28	28	0	0	0	0	18	0	9	0	0	0	1
2001	HTW	3	72	57	15	1	0	0	36	0	4	14	0	0	17
2001	HTW	4	155	101	54	7	0	0	27	0	29	47	0	0	45
2001	LNO	1	17	13	4	2	5	0	0	0	8	2	0	0	0
2001	LNO	2	22	20	2	2	0	0	0	0	20	0	0	0	0
2001	LNO	3	18	18	0	0	0	0	0	0	18	0	0	0	0
2001	LNO	4	20	19	1	0	0	0	0	0	19	1	0	0	0
2001	LNP	1	15	14	1	0	1	0	0	0	13	1	0	0	0
2001	LNP	2	27	27	0	0	0	0	3	0	21	0	0	3	0
2001	LNP	3	17	16	1	1	0	0	0	0	15	0	0	1	0
2001	LNP	4	22	22	0	0	0	0	0	0	19	0	0	3	0
2001	MCH	1	11	11	0	0	0	0	0	0	11	0	0	0	0
2001	MCH	2	20	20	0	0	0	0	3	0	16	0	0	1	0
2001	MCH	3	22	22	0	0	0	0	0	0	22	0	0	0	0
2001	MCH	4	22	20	2	1	0	0	0	0	20	1	0	0	0
2001	PHG	1	7	7	0	0	0	0	3	0	4	0	0	0	0
2001	PHG	2	30	30	0	0	0	0	16	0	13	0	0	0	1
2001	PHG	3	22	17	5	1	0	0	11	0	5	4	0	0	1
2001	PHG	4	43	22	21	0	0	0	11	0	11	21	0	0	0

Appendix 3. List of equipment and materials necessary for a single replicate season of small mammal sampling.

Equipment	**Amount**	**Purpose**
Sherman traps	225	3”x3.5”x9” collapsible traps for ease of transporting to sites.
Oats	3 (42oz) container	For baiting traps
Peanut Butter	8 (4lb) jars	For baiting traps
Bacon	5 lb	For baiting traps
Dye free paper towels	1 package	For making bait packets
Latex gloves	1 boxes (100 gloves/box)	For sanitary handling of animals
Cloth gloves	2 pairs	For handling of small mammals
Leather gloves	2 pairs	For handling of larger mammals
Clear plastic and mesh handling bags	15 plastic and 2 mesh	For handling of small mammals
Small and medium resealable bags	25	For storing dead specimens
10g spring loaded scale	1	To weigh shrews and juveniles
30g spring loaded scale	1	To weigh mice and voles
60g spring loaded scale	1	To weigh mice and voles
100g or spring loaded scale	1	To weigh large voles and chipmunks
500g spring loaded scale	1	To weigh chipmunks, squirrels, and weasels
1000g spring loaded scale	1	To weigh larger mammals
Flexible plastic metric ruler	1	For measuring length of mammals
100-m tapes	2	To lay out trapping grids
Compass	2	To orient trapping grids
Sturdy canvas backpacks	3	To carry collapsible traps
Clipboard	1	For datasheet
Waterproof data sheets	25	For inclement weather
Nestlets	300	For nesting material and temperature regulation in trap
Vial of sugar water	1	For reviving dehydrated or sick animals
Roll of flagging	1	As needed
Permanent marking pen	1	Marks-a-lot brand for marking small mammals
Soil moisture meter	1	Kelway brand for monitoring soil moisture
36” Surveyor Flags	1000	To mark trap stations

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