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POWER TO DETECT TREND IN SHORT-TERM TIME SERIES OF BIRD ABUNDANCE

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Abstract. Avian point counts for population monitoring are often collected over a short timespan (e.g., 3–5 years). We examined whether power was adequate (power ≥ 0.80) in short-duration studies to warrant the calculation of trend estimates. We modeled power to detect trends in abundance indices of eight bird species occurring across three floodplain habitats (wet prairie, early successional forest, and mature forest) as a function of trend magnitude, sample size, and species-specific sampling and among-year variance components. Point counts (5 min) were collected from 365 locations distributed among 10 study sites along the lower Missouri River; counts were collected over the period 2002 to 2004. For all study species, power appeared adequate to detect trends in studies of short duration (three years) at a single site when exponential declines were relatively large in magnitude (more than -5% year⁻¹) and the sample of point counts per year was ≥ 30 . Efforts to monitor avian trends with point counts in small managed lands (i.e., refuges and parks) should recognize this sample size restriction by including point counts from offsite locations as a means of obtaining sufficient numbers of samples per strata. Trends of less than -5% year⁻¹ are not likely to be consistently detected for most species over the short term, but short-term monitoring may still be useful as the basis for comparisons with future surveys.

Key words: floodplain habitat, Missouri River, point counts, Poisson regression, power, trend estimation, underdispersion.

Poder para Detectar Tendencias en la Abundancia de las Aves en Series Temporales Cortas

Resumen. Generalmente los conteos en puntos para monitorear poblaciones de aves son realizados durante periodos de tiempo cortos (e.g., 3–5 años). Examinamos si el poder para calcular estimaciones de tendencias era adecuado (poder ≥ 0.80) en estudios de corta duración. Modelamos el poder para detectar tendencias utilizando índices de abundancia de ocho especies de aves que se distribuyen en hábitats de planicies inundables (praderas inundadas, bosques en estado sucesional temprano y bosques maduros), como función de la magnitud de la tendencia, tamaño muestral y componentes de la varianza asociados al muestreo de cada especie y entre años. Los puntos de conteo (5 min) correspondieron a 365 localidades distribuidas en 10 sitios de estudio ubicados a lo largo de la parte baja del Río Missouri; los conteos fueron realizados entre los años 2002 y 2004. Para todas las especies estudiadas, el poder pareció ser adecuado como para detectar tendencias en estudios de corta duración (tres años) en un sitio único de muestreo, pero sólo cuando las disminuciones exponenciales fueron relativamente grandes en magnitud (más de -5% año⁻¹) y la muestra de puntos de conteo por año fue ≥ 30 . Los esfuerzos para monitorear las tendencias de las poblaciones de aves con puntos de conteo en áreas manejadas pequeñas

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(i.e., refugios y parques) deben reconocer estas restricciones impuestas por el tamaño de muestreo. Se deben incluir puntos de conteo que se encuentren fuera del área para obtener un número suficiente de muestras por estrato. Para la mayoría de las especies, es menos probable detectar consistentemente tendencias de menos de -5% año⁻¹ durante periodos cortos. Sin embargo, el monitoreo de corta duración puede ser útil como base para comparaciones con muestreos en años futuros.

Surveys for birds on public lands are typically conducted for a limited number of years. However, one of the primary motivations of such surveys is to discern trends in avian abundance. Unfortunately, there is only a small probability that a biologically meaningful trend will be detected in data from only a few years, provided such a trend exists (Hayes and Steidl 1997). Short-term studies, however, are typically all that a natural resource agency can implement. We considered this conundrum by examining a short-term series of point counts (three years) collected in a distributed manner on various lands under public ownership along the lower Missouri River. We asked whether it is worth expending the effort, as defined by time, money, and lost opportunities, to collect data for a purpose that may not be adequately addressed.

At the heart of this issue of worthiness is whether surveys conducted in most short-term studies are likely to detect biologically meaningful trends in population size. This topic of trend detection may be addressed with arguments relating to precision or power. These arguments are not entirely unrelated; we address power here, but see Gray and Burlew (2007) for arguments relating to precision. Power is a statistical measure describing the probability of detecting an effect in a population when one actually exists. Failure to confidently identify trends in population abundance may, for instance, result in a failure to recognize a population headed toward extinction (Field et al. 2004). The adequacy of monitoring programs depends on interactions among sample size (e.g., number of point counts), duration (period of monitoring), frequency of surveys during the period of monitoring, sampling variability, among-year variability in mean counts, and the ability to control variability in counts because of other factors (e.g., habitat effects, observer differences, and weather). If the probability of detecting trends is not assessed for monitoring programs, managers risk wasting resources, because their sampling design may be inadequate to detect biologically meaningful trends (Field et al. 2004).

Power is defined as $(1 - \beta)$, where β is the probability of failing to accept a hypothesis when it is true (Type II error; Snedecor and Cochran 1989). Increasing power requires increasing the probability of incorrectly accepting an alternative hypothesis (i.e., of committing a Type I error, or α). Setting conservative Type I error rates (e.g., $\alpha < 0.05$) lowers power to detect trends, but decreases the probability of alerting natural resource managers to apparent population declines that, in fact, do not exist.

Type II errors can be costly for natural resource managers (Field et al. 2004). If a significant decline in a threatened species is not identified, then a species may decline in abundance beyond a threshold at which recovery is no longer possible. In contrast, if managers respond to a perceived decline that is not real (managing a species not in decline), then resources may be wasted in the short term. Fortunately, this sort of 'false alarm' is likely to be recognized, mitigating the importance of this kind of error. If sample sizes and survey frequencies are inadequate, a monitoring program will fail to provide the precision needed to detect population changes over time.

METHODS

Ten study sites were chosen within the lower Missouri River alluvial floodplain, stretching from northwestern Missouri (near St. Joseph) to east-central Missouri (near St. Louis), U.S.A. These 10 sites were located in three U.S. Fish and Wildlife Service refuges (Big Muddy National Fish and Wildlife Refuge, Swan Lake National Wildlife Refuge, and Squaw Creek National Wildlife Refuge), three Missouri Department of Conservation Areas (Overton Bottom South, Eagle Bluffs, and Jameson Island), and the Department of Defense's Fort Leavenworth. All sites were on public land and all except two (Swan Lake National Wildlife Refuge and Squaw Creek National Wildlife Refuge) were riverward of a levee. Land cover at the various study sites consisted of wet prairie, early successional forest, and mature floodplain forest. Floodplain wet prairies, possessing $<5\%$ tree coverage, were open habitats composed of herbaceous and emergent plants and grasses. Early successional forest was generally comprised of densely forested habitat with trees <10 years of age, whereas mature floodplain forest consisted of upper canopy trees >10 years of age and >15 m tall (Thogmartin et al. 2006).

Bird counts were conducted in spring and summer (15 April–30 June) from 2002 to 2004 at up to 365 locations (survey points), spaced >250 m apart. Survey points were a stratified random sample of the three habitat types at the 10 study areas. The habitat types were defined based on digital maps of the study areas and field reconnaissance (Young et al. 2004).

All bird surveys were conducted using 5 min point counts within a 50 m radius circle (7854 m²; 0.8 ha; Ralph et al. 1993). There were nine observers during the course of the study, although some operated only in concert with another observer. The number of points surveyed varied each year, depending on weather conditions and other logistical limitations. Methods for estimating detection probability were used but were not included in these analyses (Thogmartin et al. 2006). Thus, our estimates of power are based on raw (simple) counts typically gathered on conserved lands.

Of all species observed along the lower Missouri River, we chose eight that were representative of the three land cover classes, sampled sufficiently, and, when possible, species of conservation concern

(Thogmartin et al. 2006). The Song Sparrow (*Melospiza melodia*) and Dickcissel (*Spiza americana*) were representative of wet prairie habitat, and the Dickcissel was a species of regional conservation concern. The Tufted Titmouse (*Baeolophus bicolor*), Indigo Bunting (*Passerina cyanea*), and Baltimore Oriole (*Icterus galbula*) represented early successional forest species. The Yellow-billed Cuckoo (*Coccyzus americanus*), Red-bellied Woodpecker (*Melanerpes carolinus*), and Wood Thrush (*Hylocichla mustelina*) represented mature forest species.

Variance components needed to estimate power to detect future temporal trends at a single study site were estimated from the study dataset. These components, sampling and among-year variation (denoted σ^2 and τ_{yr}^2 , respectively), were estimated with Poisson regression. Under- or overdispersion of residuals with respect to that of a Poisson-distributed random variable was addressed by equating the sampling variance to the mean (the sampling variance of a Poisson random variable equals the mean, μ) multiplied by ϕ , defined as Pearson's chi-square statistic divided by its degrees of freedom (McCullagh and Nelder 1989). Sampling and among-year variation were adjusted for site and year \times site effects; variation among these latter effects (denoted τ_{site}^2 and $\tau_{yr(site)}^2$, respectively) was not always estimable because of small numbers of years and sites. When variation among year effects was not estimable, the variance estimate was set to 0. To limit small-sample issues, species \times year \times site combinations were omitted if they contained fewer than 10 counts. Also, to allow estimation of τ_{yr}^2 , only sites with two or more sampling episodes (each with ≥ 10 counts) were retained. We used pseudolikelihood and the generalized linear mixed modeling procedure for Poisson regressions (PROC GLIMMIX in SAS; Wolfinger and O'Connell 1993, SAS Institute 2005); the mean was presumed to vary on the log scale.

Power for log-linear count models is more conveniently estimated when all variance components are on the log or modeling scale. This may be approximated with the delta method (Oehlert 1992). By this method, sampling variation of an observation on the log scale is estimated by the square of the coefficient of variation (CV = standard deviation divided by the mean; Gray and Burlew 2007), while the sampling variation of the mean is estimated as CV^2/n (where n = the number of point counts per year, here presumed constant). For this analysis, we estimated the CV at the median of the year, site, and year \times site group effects.

The variance of a trend is denoted by the variance of an observation divided by the corrected sums of squares (S_{XX}) associated with elapsed time. As we were interested in estimating trends across annual site means, "observation" in this context corresponds to an annual mean of point counts. Consequently, the variance of the trend estimate may be approximated by:

$$\left(CV^2/n + \tau_{yr}^2 \right) / S_{XX}. \quad (1)$$

Power was estimated by referring to a noncentral t distribution with [year - 2] degrees of freedom. The

associated noncentrality parameter was defined as slope = $\log(1 + r)$ divided by the variance of the slope (Neter et al. 1985:equation 3.26), where r is the postulated trend or change per unit time (here, per annum) on the log scale. This corresponds to an assumption of an exponential population model for mean abundance. The false positive error rate, or α , was set at 0.10.

Power estimates were not adjusted for species detectability. We assumed detection probabilities were constant across sites and years for a given species. The failure to accommodate detection probability and differences in detection probability may lead to greater variability in power estimates across sites and years.

RESULTS

The sampling variance of observed counts conditional on year and site effects was roughly equal to the count mean ($\phi \approx 1$) for the Dickcissel, Indigo Bunting, and Baltimore Oriole, but suggested underdispersion with respect to a Poisson random variable ($\phi \approx 0.5$) for the Song Sparrow, Tufted Titmouse, Yellow-billed Cuckoo, Red-bellied Woodpecker, and Wood Thrush (Table 1). Estimates of CV varied from 0.51 to 0.79, while estimated levels of variation among years, sites, and years \times sites were generally low.

Power to detect trends at a single study site was an increasing function of the trend magnitude and of within-year sample size (Fig. 1), and a decreasing function of the estimated species-specific sampling variance (Table 1). Estimated power increased from lows of ~ 0.2 at trends (r) of -1% year $^{-1}$ and $n = 10$ to a maximum of 1.0 at $r = -10\%$ year $^{-1}$ and, for some species and sample sizes, -5% year $^{-1}$ (Fig. 1). Given low interannual variance estimates, power was lower for the species with the highest sampling variance on the log scale (highest CV; Baltimore Oriole), intermediate for the species with an intermediate CV (Indigo Bunting), and roughly equivalent for the balance of the species (Table 1, Fig. 1). Interannual variance estimates were substantially larger for the Dickcissel and Yellow-billed Cuckoo and power was lower for these species than for species with similar CV estimates but lower τ_{yr}^2 estimates (e.g., compared to the Red-bellied Woodpecker, holding sample size constant). Species representative of early successional forest appeared to have higher sampling variances.

Of the 10 sites considered in this analysis, only four employed designs with sufficient sample size to detect what we defined as biologically meaningful declines. Each of these four sites—Squaw Creek and Swan Lake National Wildlife Refuges, and Jameson Island and Overton Bottom South Conservation Areas—had >50 point counts per year. These >50 point counts, if sampled annually, would typically provide warning for species exhibiting 10%, 5%, and sometimes 3% annual declines. The point counts conducted annually at the other six sites were too few in number to typically identify less than catastrophic ($\sim 10\%$ year $^{-1}$) declines.

DISCUSSION

We began by asking whether it was worth expending the effort to collect point count data over a short timespan to identify site-specific trends in bird abundance. For the species studied, we found that, under certain circumstances (e.g., dense within-year sample sizes [≥ 30 point counts] and trends of a relatively large magnitude [more than -5% year $^{-1}$]), reasonable power would be attained to detect trends in studies of short duration (three years).

Power to detect trends in bird abundance indices assumes moderately accurate variance component estimates, an assumption that, for the current study at least, may bear some consideration. For example, sampling variances appeared similar to or substantially lower than that expected under a Poisson distributional assumption. However, observed counts are typically substantially overdispersed with respect to that of a Poisson random variable (Cameron and Trivedi 1999, Thompson et al. 2002, Purcell et al. 2005). Sources of this underdispersion may include a combination of nontrivial observer effects and only one or a few observers per site, and detection probabilities that decrease at locations at which bird counts are extreme (i.e., unusually rare or abundant). Also, because breeding birds are often territorial, they are likely to be distributed more regularly than random (i.e., in a manner resulting in underdispersion; Royle and Nichols 2003). However, here we addressed effects of spatial correlation within sites by allowing for random site and random site \times year effects.

Our data also suggested small among-site and among-year variation. It may be worth routinely investigating whether among-site and among-year estimates have been deflated because of spatial and temporal correlation, respectively (Hox 2002). In the presence of substantial interobserver variability, site-observer associations may induce temporal correlation among means from the same site, and a tendency for a small number of observers to work only in neighboring sites may induce spatial correlation among site means. Adjusting among-year variance estimates upward will yield decreased power estimates, regardless of within-year sample size (Urquhart et al. 1998). The annual and site variation estimates in this study derive from small numbers of years and sites; the accuracy of these variance estimates will increase and any negative bias will typically decrease as the number of years and sites increases.

The importance of interannual variation to trend precision is seen by estimating the variance of a mean at different sample sizes. For example, on the log scale, the estimated proportion of variation represented by among-year effects for the Yellow-billed Cuckoo at $n = 10$ point counts was $43\% (\tau_{yr}^2 / [\tau_{yr}^2 + CV^2/n]) = 0.023 / (0.023 + 0.550^2/10)$, a value that increased to 60% at $n = 20$ point counts. Among-year and total variation on the log scale is unaffected by increases in within-year sample size.

If precision appears inadequate to detect biologically meaningful trends at a single site, investigators

TABLE 1. Sampling statistics associated with the power to detect trends in avian surveys of short duration (three years) along the lower Missouri River, including sampling variance (σ^2), coefficient of variation in counts (CV), and among-site (τ_{site}^2), among-year (τ_{yr}^2), and among-year \times site ($\tau_{yr(site)}^2$) variance estimates by species. The quasi-likelihood multiplier (ϕ) of the variance-covariance matrix accommodated estimated over- ($\phi > 1$) or under- ($\phi < 1$) dispersion with respect to a Poisson distributional assumption (conditional on site, year, and year \times site effects). Sampling variance and CV were estimated at the median of site, year, and site-year effects. Estimates are missing (na) when inestimable.

Species ^a	Sites	Years	Years \times sites ^b	Median site count ^c	ϕ	σ^2	CV	τ_{site}^2	τ_{yr}^2	$\tau_{yr(site)}^2$
Song Sparrow (<i>Melospiza melodia</i>)	2	3	5	1.69	0.54	0.91	0.57	na	0.006	na
Dickcissel (<i>Spiza americana</i>)	3	3	8	3.84	1.04	3.99	0.52	0.049	0.039	0.034
Tufted Titmouse (<i>Baeolophus bicolor</i>)	6	3	13	1.67	0.47	0.78	0.53	0.022	0.005	na
Indigo Bunting (<i>Passerina cyanea</i>)	5	3	12	3.37	1.26	4.25	0.61	0.033	0.001	0.008
Baltimore Oriole (<i>Icterus galbula</i>)	3	3	8	1.97	1.22	2.41	0.79	0.015	na	na
Yellow-billed Cuckoo (<i>Coccyzus americanus</i>)	2	3	6	1.89	0.57	1.07	0.55	0.002	0.023	na
Red-bellied Woodpecker (<i>Melanerpes carolinus</i>)	6	3	13	1.55	0.41	0.63	0.51	0.008	na	na
Wood Thrush (<i>Hylocichla mustelina</i>)	5	2	10	1.71	0.44	0.75	0.51	0.034	na	na

^a Species are ordered phylogenetically within habitat (wet prairie: Song Sparrow and Dickcissel; early successional forest: Tufted Titmouse, Indigo Bunting, and Baltimore Oriole; mature forest: Yellow-billed Cuckoo, Red-bellied Woodpecker, and Wood Thrush).

^b Year \times site combinations with < 10 point counts were omitted.

^c Median of site mean counts.

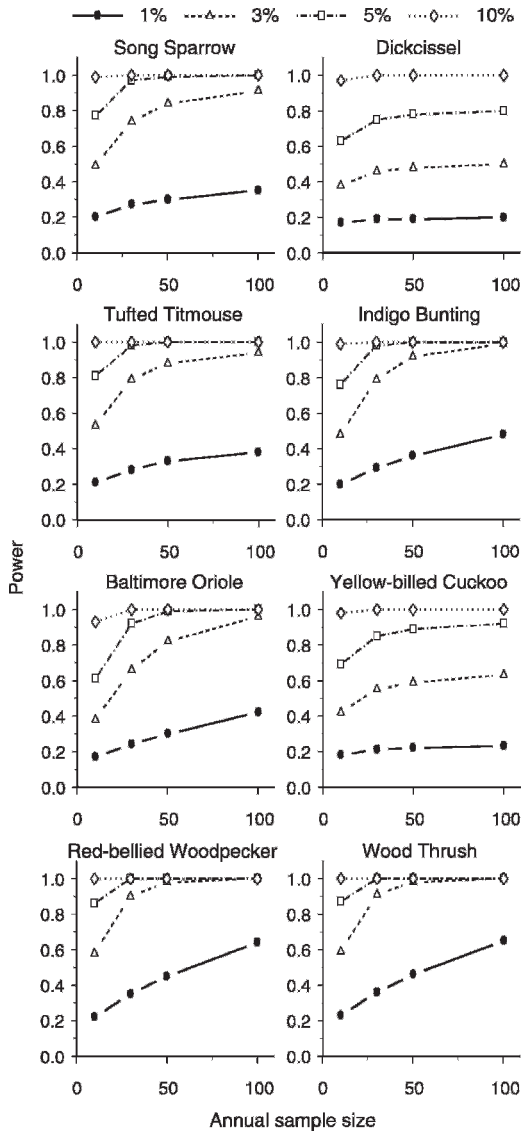


FIGURE 1. Power to detect a decline in avian abundance in short-term surveys (three years) conducted along the lower Missouri River varies between species but increases as a function of sample size and magnitude of population decline. The trends assessed were declines of 1%, 3%, 5%, and 10% year⁻¹. Species are ordered phylogenetically within studied habitats (wet prairie: Song Sparrow and Dickcissel; early successional forest: Tufted Titmouse, Indigo Bunting, and Baltimore Oriole; mature forest: Yellow-billed Cuckoo, Red-bellied Woodpecker, and Wood Thrush).

may consider estimating a temporal trend across multiple sites. Unfortunately, estimating the precision of a single grand trend across multiple sites is challenging when trends vary substantially by site

(VanLeeuwen et al. 1996). For the collections of sites and species in this study and relative to the single-site calculations presented, power would be uniformly greater to detect a given trend if sites exceeded four (VanLeeuwen et al. 1996:equation 3.2, and under the possibly naive assumption that slopes would vary no more than trivially among sites). If among-site differences in means are, as we estimated, minor, then this suggests that fewer sites will be needed to characterize a single trend across multiple sites. If site effects are ultimately found to be important, an argument may be made for trading annual sampling for sampling more sites but on a less than an annual basis (Larsen et al. 2004).

Increased sampling frequency would narrow confidence intervals around trend estimates and consequently lead to increased power. As noted above, the variance in trend estimate for a single site is the sum of the sampling and annual variances divided by S_{xx} , the corrected sum of squares of elapsed time (equation 1). Observations in three consecutive years (years 1, 2, and 3, as examined in this study) yields $S_{xx} = 2$. In contrast, three observations spaced five years apart (e.g., years 1, 6, and 11), as might occur if we traded annual sampling for the sampling of more sites, yields $S_{xx} = 50$. As a consequence, the variance of the trend estimate would decrease by 96% ($100 \times [1 - 2/50]$, holding variance components constant), illustrating that a lengthened sampling interval would lead to greater certainty in the trend estimate and therefore increased power to detect trends. This is intuitively obvious when recognizing that the trend has more years over which to operate in the latter case, resulting in a greater difference in mean observations between the first survey and the last. Although lengthening this sampling interval leads to greater power to detect trends, among-year variation is difficult to estimate. Therefore, among-year variation may be estimated in the example five-year interval by sampling in sets of two consecutive years (e.g., in years 1 and 2, 7 and 8, and 13 and 14). Regardless, these longer sampling schemes also result in a delay in trend recognition and the implementation of management prescriptions necessary to forestall further declines.

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