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MONITORING LONG-TERM POPULATION CHANGE: WHY ARE THERE SO MANY ANALYSIS METHODS?[1]

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Abstract. Monitoring long-term population change is an integral part of effective conservation-oriented research and management, and is central to the current debate on the status of Neotropical migrant land birds. However, the analysis of count data such as the Breeding Bird Survey is complicated by the subjective nature of trend estimation, and by limitations inherent to extensive, volunteer-based surveys, such as measurement error and missing data. A number of analysis methods have been used that differ in their approach to dealing with these complications and produce different estimates of population change when applied to the same data. There is, however, no consensus as to which method is the most suitable. Many analytical issues remain unresolved, such as model of trend, observer effects, treatment of missing observations, distribution of counts, and data selection criteria. These issues make it difficult to evaluate the relative merits of the methods, although a number of new approaches (nonlinear regression, Poisson regression, estimating equations estimates) offer promising solutions to some problems. I suggest the use of Monte Carlo simulations to empirically test the performance of the methods under realistic, spatially explicit scenarios of population change, and provide an example of the approach.

Key words: annual indices of abundance; comparison of analysis methods; monitoring population change; Monte Carlo simulation; Neotropical migrants; North American Breeding Bird Survey; population trend estimate.

INTRODUCTION

Population monitoring plays a critical role in conservation biology, by providing the information necessary to identify conservation problems at an early stage and to suggest possible solutions (Goldsmith 1991, James and McCulloch, *in press*). For Neotropical migrants, and other land bird species, the primary source of population information at the regional and continental scales is the North American Breeding Bird Survey (BBS). The BBS is typical of many national and international monitoring schemes in that it is based upon a large number of permanent monitoring sites, each of which is surveyed annually in a standardized manner by a skilled volunteer observer. Unfortunately, the determination of changes in population size from this kind of data is not straightforward. A number of methods of analysis have been proposed, but there is currently no consensus as to which is the most suitable. By applying three methods of trend-estimation to the same subset of BBS data for 115 species, Thomas and Martin (*in press*) have shown that the method of analysis can have important consequences for the estimates produced and inferences drawn. It is thus critical to determine which of the methods is most appropriate for a given application.

In the first part of the paper I review the factors that make the analysis of count data difficult and have led to the development of many different analysis methods. I summarize the major features of each method in Tables 1 and 2 and provide key references. In the second part, I suggest the use of Monte Carlo simulations to empirically test the performance of the methods under plausible scenarios of population change. I provide an example of the approach, in which the accuracy of three methods of trend estimation is evaluated in the context of a number of spatially explicit models of population decline. Much of the emphasis in this paper is on the estimation of long-term population change for Neotropical migrants from the BBS; however these methods and the discussion are applicable to any population where repeated counts are performed at a large number of sites.

ANALYSIS OF SURVEY DATA UNDER IDEAL CONDITIONS

Since it is not feasible to census the entire population of most bird species, population monitoring is almost always based upon surveys of a sample of the population. In an ideal sample survey, the population of

[1] For reprints of this Special Feature, see footnote 1, p. 1.

TABLE 1. Methods that have been used to estimate annual indices from bird count data.*

	Chaining[1,2]	Mountford[3,4]	Imputing[5,6]	Poisson regression[7]	Linear[8]	Residual[8]
Unit of analysis	count	count	count	count	count	trend
Annual indices calculated as	chained ratio of successive counts relative to base year	year effect from site × year model	average count (after estimation of missing data using size × year model)	year effect from site × year model	average estimated count from site × year model	overall trend from linear route regression + average of residuals for that year
Assumed distribution of counts	n/a	lognormal, autocorrelation between counts	lognormal[6], autocorrelation between counts[5]	Poisson	lognormal	n/a
Transformation of counts (c)	n/a	see (3)	$\log(c + 1/6)$[5]/ $\log(c + 1)$[6]	none	$\log(c + 0.5)$	n/a
Treatment of missing data	exclude count if no survey in successive year	implicit	use estimated values in place of missing data	implicit	use estimated values in place of raw data	implicit
Observers incorporated?	yes	no[3]/yes[4]	no, but possible	no, but possible	yes	yes
Precision weighting?	no	no	no	not developed	yes	n/a
Variance estimated by	parametric[1]/ bootstrap[2]	parametric[3]/ bootstrap[4]	bootstrap[5]/ not estimated[6]	parametric	not estimated	not estimated

* References: (1) Bailey (1967); (2) O'Connor (1992*b*); (3) Mountford (1985); (4) Peach and Baillie (1994); (5) Moses and Rabinowitz (1990); (6) Böhning-Gaese et al. (1993); (7) ter Braak et al. (1994); (8) Sauer and Geissler (1990).

interest is divided into non-overlapping units, a sample of these units is selected according to a pre-defined procedure, and a complete census is taken of each unit. In bird surveys, the sample unit is usually all the birds within a specified *survey site* ("*route*" in BBS terminology). Because population change (i.e., change in abundance) is the main parameter of interest, the most efficient procedure involves repeated annual censuses of the same sample of sites. Two issues arise in the estimation of population change from the resulting data: (1) selection of the unit of analysis, and (2) model of trend.

1. Selection of the unit of analysis

Population change is usually portrayed using *annual indices of abundance* or as *population trend* (prevailing tendency underlying annual indices). In the ideal situation, annual indices are given by the average of the counts in each year, and population trend is the average of the trends at each survey site. If the sample design involves stratification, then the average within each stratum should be weighted by the proportion of units in the stratum, usually calculated as the proportion of the total area covered that is in the stratum (*area weighting*). In addition, trends should be weighted by the abundance of individuals at the sample site (*abundance weighting*), since declines at a site containing many individuals represent a greater proportion of the population than declines at a site containing few individuals (see James et al. 1990 for an overview of weightings).

Alternatively, population trends can be calculated from the estimates of annual indices, or annual indices could be calculated as the residuals from a trend analysis. In the ideal situation, these indirect estimators produce the same results as using counts as the unit of analysis. However, because the analysis methods used in practice are modified to deal with departures from the ideal, the estimates may diverge. Hence, analyses that present both annual indices and population trends (e.g., Greenwood et al., *in press,* Peterjohn et al., *in press*) tend to use estimates of one parameter to produce the other. The unit of analysis for each of the methods is given in Tables 1 and 2.

2. Model of trend

Repeated counts at a survey site form an ecological time series, and so may be thought of as being composed of four sources of pattern: trend (prevailing tendency), interventions (irregular perturbations such as environmental effects), autocorrelation (due to population processes such as density dependence), and sampling error (Barker and Sauer 1992). Estimating population trend thus involves separating the prevailing tendency from the other components. Because the relative contributions of these components are not known a priori, the model used depends upon the judgment of the analyst (Jassby and Powell 1990). Many of the

TABLE 2. Methods that have been used to estimate population trends from bird count data.*

	Linear route-regression[1,2]	Nonlinear route-regression[3]	Rank-trends[4]	Estimating equations[5]	Poisson regression[6]	Regression of annual indices[7,8]
Unit of analysis	count	count	count	count	count	annual indices
Model of trend	linear-multiplicative	additive	ranks	linear-/polynomial-multiplicative	linear-/polynomial-multiplicative	linear-/polynomial-multiplicative
Population trend calculated as	average of site (route) trends	average of site (route) trends	average of site (route) trends	population trend effect from model	population trend effect from model	regression of annual indices
Assumed distribution of counts	lognormal	not specified	not specified	not specified/ Poisson	Poisson	n/a
Transformation of counts (c)	$\log(c + 0.5)$[1]/ $\log(c + 0.23)$[2]	$\sqrt{c}$	none	none	none	n/a
Abundance weighting?	yes	implicit	no	yes	not developed	n/a
Observers incorporated?	yes	no (nonparametric)/yes (semiparametric)	no	yes	possible	n/a
Precision weighting?	yes	no	no	yes	not developed	n/a
Variance estimated by	bootstrap[1]/jackknife[2]	parametric	parametric	bootstrap	parametric	bootstrap[7]/not estimated[8]

* References: (1) Geissler and Sauer (1990); (2) Erskine et al. (1992); (3) James et al. (1996); (4) Titus et al. (1990); (5) Link and Sauer (1994); (6) ter Braak et al. (1994); (7) Greenwood et al. (*in press*); (8) Böhning-Gaese et al. (1993).

major differences between trend-estimation methods can be attributed to differences in the way that trend is modeled.

Although there are a number of methods available for deriving trends from time series (Jassby and Powell 1990), analyses of bird counts have focused on regression-based approaches. Four regression models have been applied to count data (Table 2), each of which can produce a different picture of the prevailing tendency (Fig. 1), and each of which may be advantageous under certain conditions.

1) In *linear–multiplicative models* (e.g., linear route-regression), population change is modeled as a multiplicative process in which the expected count in year $y + 1$ is a multiple of the expected count in year y, leading to $E(C_y) = E(C_0)\beta^y$, where $E(C_y)$ is the expected count in year y and β is the trend. This model is conceptually simple, involves estimating the minimum number of parameters, and is unbiased in the face of random interventions and autocorrelation. However, it will be inaccurate if the prevailing tendency changes over time (i.e., nonlinear trend).

2) *Polynomial-multiplicative models* (e.g., Poisson regression) allow for changes in trend over time, using polynomial parameters to estimate higher order effects, i.e., $E(C_y) = E(C_0)\beta_1^y\beta_2^{y^2} \ldots \beta_n^{y^n}$. The order of the model (n) is usually determined a priori, but could be derived from the data using model selection techniques, such as likelihood ratio tests (Burnham and Anderson 1992). Although more general, it is also more susceptible to incorporating inappropriate patterns such as population cycles.

3) In *additive models* (e.g., nonlinear route-regression), no a priori model is postulated, and nontrend sources of variation in counts are removed through filters such as LOESS. The advantage of this model is that no explicit assumptions are made about the shape of the trend. One disadvantage is the requirement for specification of a smoothing parameter, which determines how much variation is filtered from the data. This must be assigned subjectively. A common criticism of both polynomial–multiplicative and additive models is that it is hard to draw inferences about overall patterns of population change since they estimate many parameters. Both can, however, provide a single parameter estimate if trend is defined as the difference in expected count between the beginning and the end of the time period of interest (e.g., James et al. 1996).

4) In *rank models* (e.g., rank-trends), trend is modeled as the tendency for counts to increase or decrease over time. Counts are ranked, and the distribution of ranks over time is used to estimate whether the population has shown a tendency to increase or decline (Fig. 1). Because ranks are used, only the direction of trend is estimated, not its magnitude. This model is only appropriate if the prevailing tendency is uni-directional, but does not require that the trend be linear.

Ultimately, the most appropriate model of trend will depend upon the pattern of population change in the population being modeled. While the true pattern is not

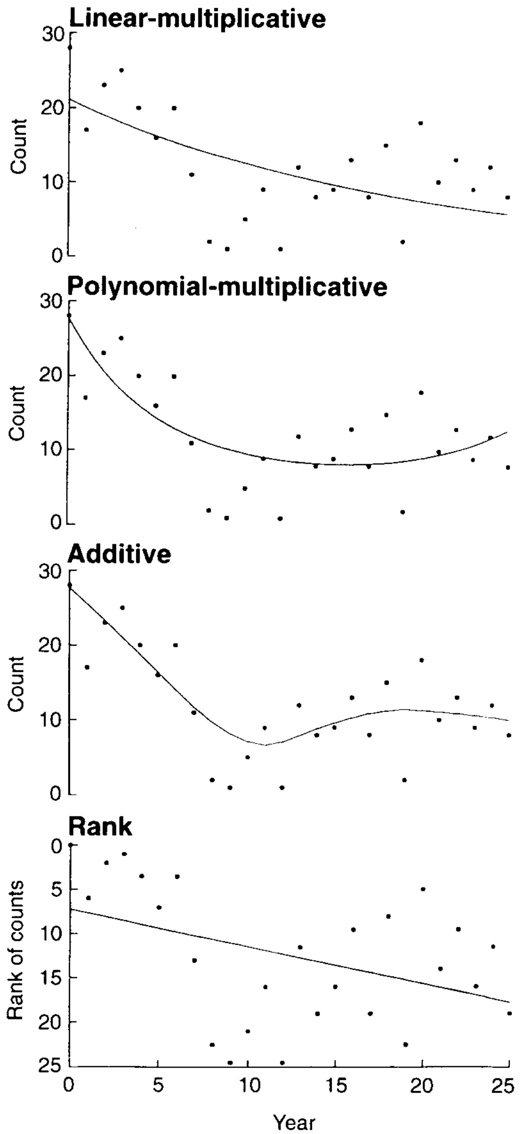

FIG. 1. Four models of trend applied to the same hypothetical count data.

known for any species monitored, the robustness of alternative models given expected population changes over space and time can be tested using population simulations (e.g., Anganuzzi 1993). Note also that the accuracy of all models is conditional on the length of the time series and the validity of the starting point (Jassby and Powell 1990, Barker and Sauer 1992). For example, the increasing trends recorded for some species in Breeding Bird Survey analyses may be due to recoveries from pesticide use or severe winters that occurred near the start of the survey (references in Barker and Sauer 1992).

Additional Factors that Complicate the Analysis of Survey Data

Extensive bird surveys often depart from the ideal situation in three ways: (1) the population from which the sample is drawn is a subset of the population about which information is required (incomplete sampling frame); (2) the number of individuals within each survey site is not counted accurately (measurement error); and (3) most sites in the selected sample are not censused every year (missing data).

1. Incomplete sampling frame

If the sample frame is incomplete, then the degree to which results obtained from the sample can be extrapolated to the population as a whole is limited. For example, in the BBS, survey sites are allocated along secondary roads; hence limited inferences can be drawn about species in which a large proportion of individuals breed in habitats such as floodplains, mountains, or forest interior that are not in close proximity to roads, or in the roadless northern part of Canada. Although this can be an important limitation to the utility of the survey program (Droege 1990), it does not affect the way that estimates are derived from the population sampled, and so is not considered further here.

2. Measurement error

Measurement error is a feature of almost all bird counts. The probability that a bird is counted given that it is present at the survey site that year (its detectability) is a function of a large number of factors, such as survey method, species, observer, habitat, time of year, weather, bird density, and breeding phenology (references in Droege 1990). Hence, estimates of population size or absolute abundance are not possible from the count data, and indices of abundance are seldom comparable between species or survey programs. In addition, sources of error that change systematically over time, such as habitat, traffic noise, inter- and intra-observer ability, can generate spurious estimates if they are not modeled in the analysis method (Barker and Sauer 1992). In particular, differences in detection rates between observers are often cited as a major potential source of bias (Barker and Sauer 1992, Sauer et al. 1994, Peterjohn et al., *in press*). Sauer et al. (1994) used several approaches to show that the number of birds counted by observers has increased, independently of trend, for the majority of species in the BBS. This result was corroborated by James et al. (1996), who found that trends were usually more positive if observer differences were not modeled in their nonlinear trend-estimation technique. However, incorporating observer differences (or other systematic sources of error) into models may result in a drastic reduction in efficiency due to overfitting, and the resulting accuracy of the estimate may be decreased. Currently, some analyses model observer differences, while others do not (Tables 1 and 2).

The problem of the inclusion of additional explanatory variables at the expense of precision has been encountered in other areas of ecology (e.g., mark–recapture studies: Lebreton et al. 1992), and the use of model selection techniques such as likelihood ratio tests and Akaike's Information Criterion have proven to be useful in selecting the most parsimonious model (Burnham and Anderson 1992). One possible solution, therefore, is to incorporate these techniques into analyses of count data that use likelihood-based estimation techniques (e.g., Poisson regression) to determine whether inter-observer differences should be modeled on a case-by-case basis. Another solution is offered by an extension to the estimating equations approach to trend estimation, in which observer differences may be modeled using relatively few parameters, with a corresponding increase in efficiency (Link and Sauer 1994). A third possibility is to apply the BBS field technique in areas where more intensive studies are taking place in an attempt to quantify observer differences and, more generally, to estimate the effect of variation in detectability. Lastly, simulation studies can be used to determine the robustness of the methods of analysis to inter-observer variation and other systematic sources of error (e.g., Geissler and Link 1988).

3. Missing data

Missing data occur when sites are not surveyed in a particular year, when sites are discontinued or new ones initiated, or when the location of a site changes. Such occurrences are a ubiquitous feature of extensive volunteer-based surveys: in the BBS ≈30% of sites are missed each year. Complete time series are not required to estimate population trends, although missing data do decrease the precision of the estimate. For this reason many published trend analyses do not include sites that are rarely or irregularly surveyed, although criteria used to exclude sites varies widely (e.g., Geissler and Sauer 1990, Böhning-Gaese et al. 1993, James et al. 1996). A complementary technique used to increase overall precision is to weight site trends by their precision when averaging them to produce regional trends (*precision weighting*). In this way, site trends estimated with low precision have little influence on the overall population trend. The calculation of the weightings should not be based upon the estimated variance of the site trends, because the autocorrelation of counts between years makes this estimate biased (James et al. 1990). Methods of analysis that use precision weightings (Tables 1 and 2) base the weightings on statistics that are proportional to the expected precision, calculated from the number and distribution of surveys.

For annual indices, imbalance in the data structure caused by missing observations is a potential source of bias and must be accounted for (Geissler and Noon 1981, James et al. 1990, ter Braak et al. 1994). A number of approaches have been taken, most of which involve either implicitly or explicitly replacing missing observations with predicted values (Table 1). The chaining method is an exception in that it creates balance by excluding observations. This further decreases precision of chaining indices and is one reason why that method is not recommended (Geissler and Noon 1981, Mountford 1985, Peterjohn et al., *in press*).

Missing data and measurement error combine to obscure the true pattern of population change over space and time, and thus make the selection of an appropriate statistical model of counts difficult. A number of methods assume that counts are log-normally distributed (Tables 1 and 2). This simplifies the calculation of estimates, but has a number of disadvantages, including the implicit assumption that counts are continuous (i.e., non-integer), and the need to add a constant before log-transformation (Collins 1990, Link and Sauer 1994). These problems are avoided by Poisson regression (Tables 1 and 2), which assumes that counts are a Poisson random variable. All of these methods may, however, be biased if the model of counts is incorrect (e.g., Geissler and Link 1988). Because of this a number of trend estimation methods use non-parametric estimators, which do not require specification of the distribution of counts (Table 2), but are usually less efficient where a parametric model would be justified.

Evaluating the Methods Using Monte Carlo Simulations

From the above discussion it is clear that there are a number of unresolved issues that prevent a complete evaluation of the relative merits of the methods of analysis. One approach is to use Monte Carlo simulations (Manly 1991) to generate count data containing known population changes, and to use these data to evaluate the methods of analysis. While the true patterns of population change and measurement error over space and time are not known, a number of plausible hypotheses can be used to construct realistic count data. These data can then be used to determine the accuracy of the methods under a variety of models for population change. An example of the approach is given below.

Partners in Flight, the multi-agency coalition responsible for initiating the Neotropical Migratory Bird Conservation Program (Finch and Stangel 1993), has suggested that continental monitoring programs such as the BBS should be able to detect a 50% decline in the population of a species over 25 yr with 90% power

(Butcher et al. 1993). The purpose of the simulation study described here was to evaluate the bias and efficiency of three BBS trend-estimation methods in the context of linear-multiplicative declines of this magnitude and under three different geographic scenarios of population change. The methods evaluated were the same as those compared in Thomas and Martin (*in press*): the U.S. National Biological Service (USNBS) implementation of linear route-regression (Geissler and Sauer 1990), the Canadian Wildlife Service (CWS) implementation of the same approach (Erskine et al. 1992), and rank–trends analysis (Titus 1990). The USNBS and CWS co-administer the BBS program, and results from their analyses are widely cited in secondary publications. The methods of analyses are very similar: both log-transform the count data, calculate trends for each site (route) as the slope of a linear regression of log count against time (hence "linear route-regression"), and take a weighted average of these slopes to produce an overall trend for the population of interest. However the implementations differ in many details, such as the criteria for the exclusion of data, the constant that is added before log-transformation, the treatment of observer differences, the way that site trends are averaged to form the population trend (arithmetic mean in the USNBS implementation, geometric mean in the CWS version), the method of back-transforming the population trend from the log scale to the arithmetic scale, the method of calculating area, abundance, and precision weightings, and the technique used to estimate confidence intervals. Rank–trends analysis is a nonparametric method that estimates the direction and consistency (but not the magnitude) of population change over time at each site, and averages these statistics over sites to produce an estimate of population change. No weightings are used, and no attempt is made to account for observer differences. Thomas and Martin (*in press*) found that these three methods produced different results when they were applied to the same subset of BBS data; however, because there was no baseline of known trends, they could not tell which method was the most accurate.

Wilcove and Terborgh (1984) have proposed that species undergoing population declines will show one of three broad geographic patterns of population change: (1) declines in abundance at the center of the species geographic range but little contraction in the size of the range; (2) range contractions with little change in abundance at the center; and (3) both declines at the center and range contraction. These scenarios were modeled by assuming that species abundance was proportional to the normal distribution with respect to distance from the center of the range (Brown 1984, Lacy and Bock 1986, Lawton 1993), as shown in Fig. 2. Population declines were simulated by reducing the

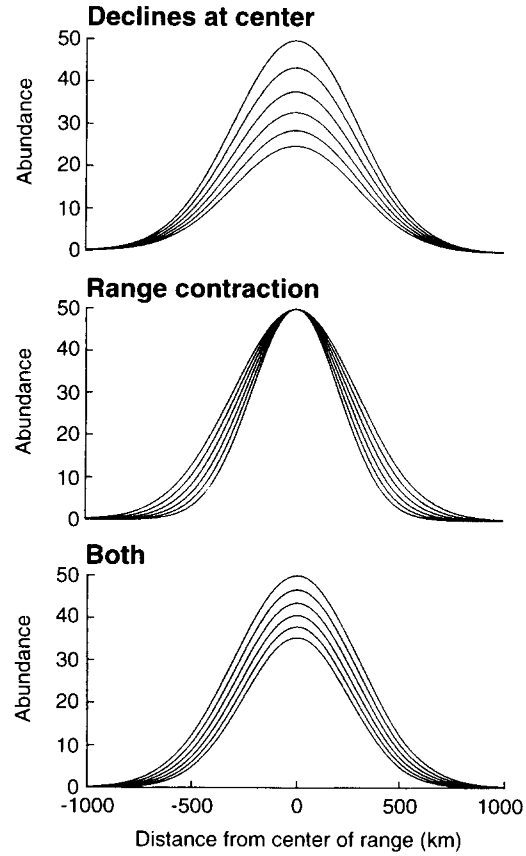

FIG. 2. Three geographic scenarios of population decline. In each case the total population declines by 50% over 25 yr. Curves show the distribution of abundance across a cross section of the species range at successive 5-yr intervals, with the outer top line being year 0 and the inner bottom line being year 25.

overall abundance by 2.73%/yr (i.e., 50% over the 25 yr of the survey, Fig 2). Annual variation in addition to trend was modeled by multiplying the overall abundance by a random log-normal variable with mean (on the log scale) of 0 and variance set by a model parameter.

The actual count at a site was calculated as the abundance at that site, given its distance from the center of the species range and the year, multiplied by variables representing random within-site variation and measurement error due to inter-observer differences, and rounded to the nearest integer value. Within-site variation was modeled as a random variable in a manner similar to annual variation. Each observer was assigned an observer error from a random log-normal distribution with a mean that was proportional to the year that the observer began surveying the route, and variance set by a model parameter. This allowed the modeling of increases or decreases in inter-observer quality over

TABLE 3. Parameters used in the simulation study of the performance of three BBS trend-estimation methods in the context of overall population declines and under three different geographic scenarios of population change.

Parameter	Value
Abundance in year 0 at center of range	50
Variance of range in year 0 (km²)	90 000
Rate of change of abundance at center of range (%/yr)	−2.73/0/−1.37*
Rate of change of range variance (%/yr)	0/−2.73/−1.37*
Year-to-year fluctuations about trend (variance, log scale)	0.16
Within-site variation in counts (variance, log scale)	0.04
Rate of change of inter-observer mean (log scale)	0.05
Inter-observer variation (variance, log scale)	0.16
Site density (sites/km²)	0.0005
Probability that a new observer adopts a site	0.1
Probability of an observer surveying a site in a given year	0.8
Probability of the observer never surveying the site again	0.2

* Where three values are given, the first refers to the scenario declines at center of range, the second to range contraction, and the third to both declines at the center and range contraction (see Fig. 2).

time (Sauer et al. 1994). The locations of sample sites were selected at random from the circular sample space centered around the center of the species range and with a radius such that 99.9% of the initial population was contained in the space. The pattern of surveillance at each site was simulated using three parameters: the probability that a new observer adopts the site, the probability of the observer not surveying the site in a given year and the probability of the observer never surveying the site again.

In a full simulation study, data would be generated at a number of levels for each parameter. Here, for the purposes of illustration, results are presented for each geographic scenario of population change at one level of the other parameters (Table 3). These parameter values were chosen to represent median values obtained from an analysis of BBS data for British Columbia. For each scenario, 100 sets of data were generated and analyzed by the three methods. The main features of the results were:

1) Both route-regression methods were efficient in "detecting" the declines (i.e., high proportion of significantly declining trends) in all three scenarios (Fig. 3). O'Connor (1992*a*) has suggested that these methods may be less sensitive to declines that are concentrated at the edge of species ranges, due to the abundance weightings used. No evidence for this was found in these simulations.

2) Rank–trends analysis produced inaccurate results when declines were concentrated at the center of the species range (Fig. 3). Observer differences were not accounted for in the rank–trends method, but increasing observer quality was a major component of this simulation, resulting in increasing counts on some routes. Rank–trend estimates for scenarios involving range contraction were "correct" due to the rapid decline in abundance at peripheral sites outweighing the tendency for increases due to observer differences. Since route trends are not weighted by abundance in this method, trends at peripheral sites were given equal weight to those at the center. These results demonstrate the potential importance of accounting for observer differences, and the effect of weightings on the overall estimates.

3) The CWS route-regression method showed a small negative bias in all three scenarios (median: $-0.82\%/yr^{-1}$), while the USNBS method was not significantly biased (Table 4). The bias may be due to the use of the geometric mean in estimating overall population trends from route trends rather than the arith-

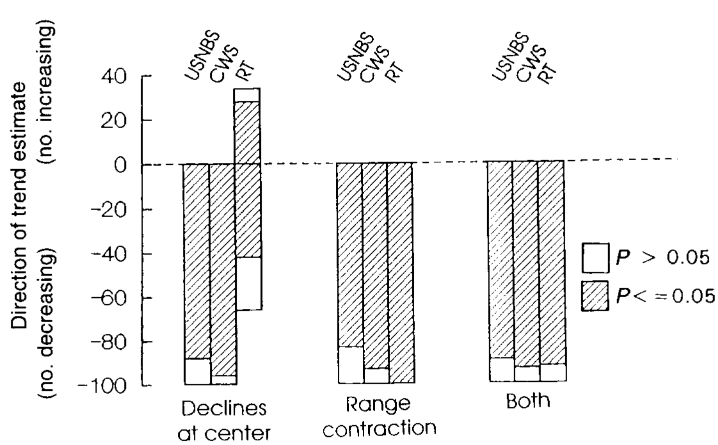

FIG. 3. The direction and statistical significance of trends estimated by three methods of trend analysis from 100 simulations of BBS data. The hatched portion of the bars indicates the number of trends that were significantly different from 0 (α = 0.05). USNBS = U.S. National Biological Service linear route-regression method; CWS = Canadian Wildlife Service linear route-regression method; RT = rank–trends method.

TABLE 4. Bias and precision of two trend estimators (USNBS and CWS† linear route-regression) from 100 simulations of Breeding Bird Survey data at each of three geographic scenarios of population decline.

	Median bias (rate of change per year, %)		Standard deviation‡ (rate of change per year, %)			
			USNBS		CWS	
	USNBS	CWS	Median estimated	Actual	Median estimated	Actual
Declines at center	0.27^{NS}	−0.64***	0.54***	1.45	0.60***	1.28
Range contraction	-0.14^{NS}	−1.11***	0.27***	1.29	0.22***	1.44
Both	-0.21^{NS}	−0.86***	0.60***	1.36	0.68***	1.40

† USNBS = U.S. National Biological Service; CWS = Canadian Wildlife Service.
‡ Asterisks indicate the statistical significance of Wilcoxon signed-rank sum tests of the hypothesis that the median bias is 0 (Median bias columns) or that the median estimated standard deviation is equal to the actual standard deviation calculated from the 100 estimates (Standard deviation columns). *** $P < 0.001$; $^{NS} = P > 0.05$.

metic mean, although there are many other differences between the methods (Thomas and Martin, *in press*).

4) Both route-regression methods underestimated the standard deviation of the trend estimates (Table 4). This implies that the frequency of Type I errors (significant population trends detected when none are occurring) may be higher than the nominal level of $\alpha = 0.05$, and suggests that estimates of the number of species adequately sampled by the BBS based upon power analysis (e.g., Peterjohn et al., *in press*) may be overestimates. Simulations under the null hypothesis of no population change are required to evaluate this possibility.

In summary, this example has demonstrated the utility of simulation studies in highlighting potential areas of bias in the methods under a small number of "What if. . ." scenarios.

CONCLUSIONS

In order to be effective, population monitoring programs must provide efficient and reliable estimates of population change. However, as I have shown in this paper, estimating population change from extensive survey data is a considerable statistical challenge. The diversity of analysis methods available reflect the number of unresolved issues: the appropriate unit of analysis, model of trend, treatment of observer differences and other measurement errors, treatment of missing surveys, distribution of counts, and data selection criteria.

The field of population change estimation is evolving rapidly: almost all of the methods presented in this paper have been developed or modified in the past 5 yr, and further statistical advances will undoubtedly lead to the development of more methods in the future. As the length of monitoring time series increases, other approaches to modeling trend, such as autoregressive trend analysis (e.g., Edwards and Coull 1987) will become possible, although the limitations of the data will remain an important constraint. Due to the number of unresolved issues, it is difficult to determine which analysis method is appropriate to test a given hypothesis. Monte Carlo simulations offer a useful empirical approach for evaluating the methods under explicit models of population change, and hence of determining which analysis method is likely to be the most reliable under a wide variety of realistic scenarios.

Confronted with all of the problems outlined above, the reader may be tempted to ask whether reliable estimates of population change can ever be derived from large-scale monitoring schemes such as the BBS. To some extent they already are: trends from the BBS are positively correlated with those from a wide range of independent monitoring schemes, each of which has different survey biases and uses different methods of analysis (Butcher et al. 1990, Hagan et al. 1992, Hussell et al. 1992, Cyr and Larivée 1993). Extreme changes in the population status of birds well sampled by the BBS are almost certain to be detected using any of the methods of analysis discussed here (Sauer et al. 1994). However, the degree to which these schemes can meet more stringent targets for accuracy, such as those laid out by Partners in Flight (Butcher et al. 1993), is unknown. Determining and realizing the full potential of extensive monitoring programs will require the continued commitment of field biologists to investigate potential sources of bias and of statisticians to minimize bias while retaining the greatest possible efficiency.

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