# Power and precision of replicated helicopter surveys in mixed bushveld

**B.K. REILLY and R.H. EMSLIE**

Reilly, B.K. and R.H. Emslie. 1998. Power and precision of replicated helicopter surveys in mixed bushveld. *Koedoe* 41(1): 47–56. Pretoria. ISSN 0075–6458.

It is well known that aerial game counts in South Africa are often applied in a non-standardised, unreplicated fashion. They contribute to poor management decisions based on their results as they may be subject to large statistical Type I and II errors. Replicate counts of large herbivores were conducted in a 8 500 ha sample site in the Loskop Dam Nature Reserve in July 1991. These data were used to estimate precision of the counts and estimate statistical power to detect population changes for different combinations of replications and significance levels.

Keywords: Game counting, helicopter counts, precision, statistical power.

*B.K. Reilly, Department of Nature Conservation Faculty of Forestry, University of Stellenbosch, Private Bag X1, Matieland, 7602 Republic of South Africa* (contact address: Gauteng Directorate of Nature Conservation, Private Bag X209, Pretoria, 0001 RSA)*; R.H. Emslie, Ecoscot Consulting Services, P.O. Box 1212 Hylton, 3245 Republic of South Africa.*

## Introduction

The use of aerial surveys as a tool in the management of large ungulates can be traced back as far as 1935 (Cahalane 1938). The use of aircraft and particularly helicopters has grown consistently since the 1950s and today is almost universally applied in enumerating many species of wild ungulates, white tailed deer (Beasom *et al.* 1986), mule deer (Bartman *et al.* 1986) and elk in Michigan (Otten *et al.* 1993). Furthermore the method has been applied to many species of African ungulates and results have been recorded by many authors. Van Lavieren & Esser (1979) report results of counts of giant eland *Tragelaphus derbianus*, roan antelope *Hippotragus equinus*, defassa waterbuck *Kobus ellipsiprymnus defassa*, bohor reedbuck *Redunca redunca* and Bubal hartebeest *Alcelaphus buselaphus* in west Africa and several species such as impala *Aepyceros melampus*, topi *Damaliscus lunatus*, sitatunga *Tragelaphus spekii*, eland *Tragelaphus oryx* and waterbuck *Kobus ellipsiprymnus* in Rwanda. The method has been used in both East Africa (Dasman & Mossman 1962; Melton 1978) and South Africa (Hirst 1969; Goodman 1977; Collinson 1985; Bothma *et al.* 1990; Eiselen 1994; Van Hensbergen *et al.* 1996).

The use of the technique has moved beyond application in national parks and nature reserves and, particularly in the South African context, is consistently applied to provide management guidelines for many game ranching undertakings. Furthermore, these applications are often applied by inexperienced staff unschooled in the underlying statistical principles involved (Adcock *pers. comm.*).

The helicopter survey has many detractors due to failure to ensure that underlying assumptions have been met (Melton 1978; Caughley 1974). The essential requirement for skilled observers, high observer fatigue, difficulty in standardisation and high costs mitigate against the technique (Caughley

1979; Norton-Griffiths 1978; Seber 1992), yet it remains the technique of choice over much of South Africa. It provides a data set in a short time for smaller properties and is less sensitive to habitat diversity and population sizes than many sampling methods. Visibility from the air is also generally better than from the ground in most bushveld applications.

A plethora of publications exists on accuracy of helicopter counts (Rice & Harder 1977; Beasom 1979; Payne 1981; De Young 1985; Beasom *et al.* 1986; Seber 1992; Bothma *et al.* 1990) and the effect of various factors such as height and speed on accuracy. Other investigations quantify observer bias (Pennycuick & Western 1972; Norton-Griffiths 1978; Melton 1978; Graham & Bell 1989) but there are few references on precision and power of these applications.

Unless counts, or for that matter, any monitoring action are replicated thus allowing estimation of variance, any resulting management decision made with unknown within-technique variation could lead to a decision subject to large Type I or Type II error. Replication allows the assessment of the risks involved in making decisions based on these counts.

The prime objective of most large ungulate aerial counts is to show population changes over time. In the accepted context this change is from one year to the next. The precision, if unmeasured, may exceed the magnitude of population change to be measured and thus make the resulting estimate valueless. At best, large ungulate counts are subjected to a high degree of standardisation, on the assumption that high precision will be achieved, an assumption which has not been verified and which may very often be violated and compounded by poor observer performance (Adcock *pers. comm.*). Obviously the high cost of helicopter aerial game counting operations mitigates against the replication of counts (Collinson 1985).

Power analyses in wildlife management applications are not widely known or applied in South Africa with the exception of Emslie (*pers. comm.*) on forest antelope and Emslie, Fourie & Reilly (1994) on aerial counts of cycads. More recently Hayes & Steidl (1997) discussed the concept as far as amphibian population trends are concerned and Reed & Blaustein (1995) in terms of population decline. The topic is covered in environmental management and fisheries (Conquest 1983; Peterman 1990; Conquest 1993; Mapstone 1995). This is in contrast with the literature on concepts such as accuracy of gamecounting. Hinds (1984) refers to power analysis in monitoring long-term trends in terrrestrial ecosystems. The use of power analysis is however documented in the behavioural sciences (Cohen 1988), medicine (Freiman *et al.* 1978) and psychiatry (Rothpearl *et al.* 1981). Further, power analysis is widely used in market research and quality control where the technique is referred to as process analysis (Kraemer 1987) and is often used to determine sample sizes. Researchers are generally preoccupied with $\alpha$ or the probability of a Type I error (Lipsey 1990), often ignoring the probability of a type II error. Cohen (1988) defines the power of a statistical test of a null hypothesis ($H_0$) as the probability of rejecting the null hypothesis when it is in fact false and denotes it as $1-\beta$.

In 1991 complete aerial surveys (of an 8 500 ha sample site of Loskop Dam Nature Reserve) were made over four successive days with an experienced team. This data set was used to establish statistical power to detect population change and precision of gamecounting as conducted on the Reserves of the then Transvaal Chief Directorate of Nature and Environmental Conservation.

## Study area

Loskop Dam Nature Reserve is 24 800 ha in area and situated in the western Mpumalanga Province 180 km northeast of Pretoria. The area has an annual rainfall of 600 mm and consists of 18 distinct vegetation communities (Theron 1978). The topography ranges from flat grassland plains to wooded gorges.

The study was conducted over the 8 500 ha section of the reserve north of the dam. This portion is a closed area because it is bounded on the south and east by the dam and the north and west by a game-proof fence.

## Method

A four-seat Bell Jet Ranger III helicopter (with a pilot, navigator and two experienced counters) was used. Counting marker bars were fitted and set to delineate a 330 m wide strip at a height of 53 m above ground level. Height was regulated using a radar altimeter and counts were done with the rear doors removed. All counts were done during July after leaf drop and replicates were conducted on consecutive days at the same time of day. Speed was constant at 96 km/h. Navigation was accomplished by heading and counter heading and tracked by the navigator on a 1 km² grid overlay of the 1:50 000 topocadastral map.

Data were recorded onto handheld recorders and later transcribed to data sheets using the grid numbers of the overlay. All animals seen were counted and, where necessary, herds were flushed to ensure accurate enumeration. The navigator marked the positions of herds crossing strips on the map to avoid overcounting. Precision of the counts is expressed as a coefficient of variation, i.e. $x/s$ where $s$ is the standard deviation and $x$ the average of the four counts.

*Post hoc* analyses can attempt to answer three questions. Firstly, what number of replicates would be needed to detect a difference (effect size) of the magnitude observed in the data with pre-selected α and β? In other words, how many replicates would be needed to detect real change in the number of animals from one year to the next. Secondly, what is the smallest difference that can be detected for a given number of replicates, again, with pre-selected values of α and β, and finally, what is the statistical power of the test procedure? In this case analyses will centre around the latter two questions given the fixed sample size of four replicates. A Poisson distribution would be appropriate for number of animals per unit area for example but in this study the source of variability is mainly sightability and with limited sample size this would be impossible to verify. The assumption is that the counting error has a normal distribution, which is hopefully not unreasonable since the total counts are quite large, thus making it possible to invoke the central limit theorem.

In order to determine the power of detecting a change in the population size, the assumption is made that the population variance in year 1 is equal to the population variance in year 2.

Let $s$ denote the standard deviation of the replicated counts in year 1, and let $x_1$ be the mean in year 1 and $x_2$ in year 2. The number of replicates in year 1 is $n_1$ and in year 2 the number of replicates is $n_2$.

The null hypothesis is:-

$$H_0 : \mu_1 = \mu_2 \qquad \text{i.e. } \mu_1 - \mu_2 = 0$$

i.e. the population means in year 1 and year 2 are equal.

One of the alternative hypotheses is that the mean in year 1 is larger than the mean in year 2, i.e. there is a decrease in size:

$$H_a: \mu_1 - \mu_2 > 0$$

If the null hypothesis is true, then

$$P\left\{ \frac{\bar{x}_1 - \bar{x}_2}{s\sqrt{\frac{1}{n_1}+\frac{1}{n_2}}} > t_{n_1+n_2-2;\alpha} \right\} = \alpha$$

$$\text{i.e. } P(t > t_{n_1+n_2-2;\alpha}) = \alpha$$

where

$$\frac{\bar{x}_1 - \bar{x}_2}{s\sqrt{\frac{1}{n_1}+\frac{1}{n_2}}} = t$$

and $t_{n_1+n_2-2;\alpha}$ is the critical value of the $t$-distribution with $n_1 + n_2 - 2$ degrees of freedom. Hence, the variable $t$ is assumed to have a $t$-distribution with $n_1 + n_2 - 2$ degrees of freedom.

To determine the power of detecting a change of $\Delta \bar{x}_1$ where $\Delta<1$, the following probability is calculated:

$$P\left\{ t - \frac{\Delta \bar{x}_1}{s\sqrt{\frac{1}{n_1}+\frac{1}{n_2}}} > t_{n_1+n_2-2;\alpha} \right\}$$

$$\text{i.e. } P\left\{ t > \frac{\Delta \bar{x}_1 + t_{n_1+n_2-2;\alpha} \cdot s\sqrt{\frac{1}{n_1}+\frac{1}{n_2}}}{s\sqrt{\frac{1}{n_1}+\frac{1}{n_2}}} \right\}$$

Similarly, in order to detect an increase in size from year 1 to year 2 the following is calculated:

$$\text{i.e. } P\left\{ t > \frac{\Delta^* \bar{x}_1 - t_{n_1+n_2-2;\alpha} \cdot s\sqrt{\frac{1}{n_1}+\frac{1}{n_2}}}{s\sqrt{\frac{1}{n_1}+\frac{1}{n_2}}} \right\}$$

where $\Delta^* > 1$

The standard $t$-distribution is used in this case although an argument can be made to use the non-central $t$-distribution (Pearson & Hartley 1976). The power values obtained using the standard $t$-distribution may differ from those obtained from the non-central $t$ but serve as good indicators of the power of detecting change in an animal population.

## Results and Discussion

The lowest coefficient of variation is for kudu (6.0 %) and the largest for giraffe (36.0 %) (Table 1). It has long been suspected that aerial counts of giraffe show large variability due to their behaviour and cryptic colouring of the animals, although in this case, the small population size probably contributes to the high variability.

McCullough (1994), in documenting herd composition counts of black tailed deer, describes large variance, consequently questioning the data's value in decision making. Beasom (1979) reports variations in replicated counts using helicopters ranging from 0.9–32.3 % in Texas, while Le Resche & Rausch (1974) report coefficients of variation ranging from 16–41 % in fixed wing counts of moose. Hirst (1969) reports on variability of 12.8–32.2 % on strip counts of blesbok on Rietvlei Nature Reserve.

Table 1

*Precision and summary statistics of four replicate total helicopter counts at Loskop Dam Nature Reserve, July 1991*

| Species | Count 1 | Count 2 | Count 3 | Count 4 | Mean | $s$ | Var | CV | Max | Min |
|---|---|---|---|---|---|---|---|---|---|---|
| Kudu | 263 | 274 | 260 | 237 | 258.50 | 15.6 | 241.7 | 6.0 | 274 | 237 |
| Zebra | 262 | 236 | 243 | 214 | 238.75 | 19.8 | 392.9 | 8.3 | 262 | 214 |
| Mt Reedbuck | 163 | 149 | 170 | 141 | 155.75 | 13.2 | 172.9 | 8.4 | 170 | 141 |
| Sable | 43 | 41 | 45 | 36 | 41.25 | 3.9 | 14.9 | 9.3 | 45 | 36 |
| Wildebeest | 160 | 192 | 200 | 192 | 186.00 | 17.7 | 314.7 | 9.5 | 200 | 160 |
| Impala | 285 | 346 | 291 | 345 | 316.75 | 33.2 | 1108.3 | 10.5 | 346 | 285 |
| Ostrich | 24 | 27 | 28 | 23 | 25.50 | 2.3 | 5.7 | 9.3 | 28 | 23 |
| White rhinoceros | 32 | 33 | 25 | 30 | 30.00 | 3.6 | 12.7 | 11.8 | 33 | 25 |
| Warthog | 35 | 37 | 46 | 37 | 38.75 | 4.9 | 24.3 | 12.7 | 46 | 35 |
| Giraffe | 33 | 14 | 19 | 26 | 23.00 | 8.3 | 68.7 | 36.0 | 33 | 14 |

For discussion purposes it is essential to clarify the issue of statistical power and its value in wildlife monitoring. The significance level (α or Type I error) or the probability of rejecting a true null hypothesis is usually set as small as possible in scientific experimentation. It follows that the smaller the value the more rigorous the rejections of the null hypothesis thence, the existence of the phenomenon in question is proven. Small alpha values often lead to relatively small power values although power values are also dependent on other factors such as the alternative hypothesis. The Type II (β error) or the rate of probability of failing to reject a false null hypothesis is related to power ($\beta=(1-(1-\beta)=(1-\text{power})))$, low power thence relating to large values of β.

Power to detect various levels of population change is presented for kudu, blue wildebeest, impala, white rhinoceros, mountain reedbuck and zebra (the species of the greatest significance in terms of management) (Tables 2–7). The results are given for 15 % and 10 % significance levels, the premise being that 5 % would be impractical for monitoring actions of this nature. The power achieved at 5 % significance is so low that the effort may be considered a waste of time. Lipsey (1990) makes the case that additional variability stemming from inconsistent measurement procedures under field conditions necessitates the relaxing of significance levels from the accepted laboratory standards. These data are also presented for various replication levels from year 1 to year 2 viz. 2–2, 3–3 and 4–4 (the 1–1 option being untestable due to inability to calculate a standard error).

For purposes of comparison and as a realistic management scenario it was decided to use 20 % as the benchmark detectable difference from year 1 to year 2.

The power to detect a 20 % population change in kudu is 80 % (Table 2). For the remaining species, power values are lower:

*Power to detect percentage population change using a two replicate level in year 1 and year 2 at 15 % significance*

| % Population change | Kudu | Wildebeest | Impala | White rhinoceros | Mountain reedbuck | Zebra |
|---|---|---|---|---|---|---|
| –40 | 0.98 | 0.90 | 0.87 | 0.81 | 0.93 | 0.94 |
| –35 | 0.96 | 0.85 | 0.80 | 0.71 | 0.90 | 0.90 |
| –30 | 0.94 | 0.76 | 0.69 | 0.59 | 0.83 | 0.84 |
| –25 | 0.90 | 0.62 | 0.53 | 0.44 | 0.72 | 0.73 |
| –20 | 0.80 | 0.44 | 0.37 | 0.31 | 0.53 | 0.54 |
| –15 | 0.57 | 0.28 | 0.24 | 0.21 | 0.33 | 0.34 |
| –10 | 0.30 | 0.17 | 0.16 | 0.15 | 0.19 | 0.20 |
| –5 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 |
| 0 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 |
| 5 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 |
| 10 | 0.30 | 0.17 | 0.16 | 0.15 | 0.19 | 0.20 |
| 15 | 0.57 | 0.28 | 0.24 | 0.21 | 0.33 | 0.34 |
| **20** | **0.80** | **0.44** | **0.37** | **0.31** | **0.53** | **0.54** |
| 25 | 0.90 | 0.62 | 0.53 | 0.44 | 0.72 | 0.73 |
| 30 | 0.94 | 0.76 | 0.69 | 0.59 | 0.83 | 0.84 |
| 35 | 0.96 | 0.85 | 0.80 | 0.71 | 0.90 | 0.90 |
| 40 | 0.98 | 0.90 | 0.87 | 0.81 | 0.93 | 0.94 |

Table 3

*Power to detect percentage population change using a two replicate level in year 1and year 2 at 10 % significance*

| % Population change | Kudu | Wildebeest | Impala | White rhinoceros | Mountain reedbuck | Zebra |
|---|---|---|---|---|---|---|
| –40 | 0.97 | 0.83 | 0.77 | 0.65 | 0.89 | 0.90 |
| –35 | 0.95 | 0.73 | 0.64 | 0.49 | 0.83 | 0.84 |
| –30 | 0.91 | 0.58 | 0.48 | 0.37 | 0.70 | 0.72 |
| –25 | 0.83 | 0.40 | 0.32 | 0.25 | 0.51 | 0.53 |
| –20 | 0.64 | 0.25 | 0.21 | 0.17 | 0.32 | 0.33 |
| –15 | 0.36 | 0.16 | 0.14 | 0.12 | 0.19 | 0.19 |
| –10 | 0.17 | 0.10 | 0.10 | 0.10 | 0.11 | 0.11 |
| –5 | 0.10 | 0.10 | 0.10 | 0.10 | 0.10 | 0.10 |
| 0 | 0.10 | 0.10 | 0.10 | 0.10 | 0.10 | 0.10 |
| 5 | 0.10 | 0.10 | 0.10 | 0.10 | 0.10 | 0.10 |
| 10 | 0.17 | 0.10 | 0.10 | 0.10 | 0.11 | 0.11 |
| 15 | 0.36 | 0.16 | 0.14 | 0.12 | 0.19 | 0.19 |
| 20 | 0.64 | 0.25 | 0.21 | 0.17 | 0.32 | 0.33 |
| 25 | 0.83 | 0.40 | 0.32 | 0.25 | 0.51 | 0.53 |
| 30 | 0.91 | 0.58 | 0.48 | 0.37 | 0.70 | 0.72 |
| 35 | 0.95 | 0.73 | 0.64 | 0.49 | 0.83 | 0.84 |
| 40 | 0.97 | 0.83 | 0.77 | 0.65 | 0.89 | 0.90 |

Table 4

*Power to detect percentage population change using a three replicate level in year 1 and year 2 at 15 % significance*

| % Population change | Kudu | Wildebeest | Impala | White rhinoceros | Mountain reedbuck | Zebra |
|---|---|---|---|---|---|---|
| –40 | 1.00 | 0.99 | 0.98 | 0.96 | 0.99 | 0.99 |
| –35 | 1.00 | 0.97 | 0.96 | 0.93 | 0.99 | 0.99 |
| –30 | 0.99 | 0.95 | 0.92 | 0.87 | 0.97 | 0.97 |
| –25 | 0.99 | 0.89 | 0.84 | 0.77 | 0.93 | 0.94 |
| –20 | 0.96 | 0.76 | 0.69 | 0.61 | 0.84 | 0.85 |
| –15 | 0.86 | 0.56 | 0.49 | 0.41 | 0.64 | 0.66 |
| –10 | 0.60 | 0.32 | 0.29 | 0.25 | 0.38 | 0.39 |
| –5 | 0.24 | 0.16 | 0.15 | 0.15 | 0.18 | 0.18 |
| 0 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 |
| 5 | 0.24 | 0.16 | 0.15 | 0.15 | 0.18 | 0.18 |
| 10 | 0.60 | 0.32 | 0.29 | 0.25 | 0.38 | 0.39 |
| 15 | 0.86 | 0.56 | 0.49 | 0.41 | 0.64 | 0.66 |
| 20 | 0.96 | 0.76 | 0.69 | 0.61 | 0.84 | 0.85 |
| 25 | 0.99 | 0.89 | 0.84 | 0.77 | 0.93 | 0.94 |
| 30 | 0.99 | 0.95 | 0.92 | 0.87 | 0.97 | 0.97 |
| 35 | 1.00 | 0.97 | 0.96 | 0.93 | 0.99 | 0.99 |
| 40 | 1.00 | 0.99 | 0.98 | 0.96 | 0.99 | 0.99 |

Table 5

*Power to detect percentage population change using a three replicate level in year 1 and year 2 at 10 % significance*

| % Population change | Kudu | Wildebeest | Impala | White rhinoceros | Mountain reedbuck | Zebra |
|---|---|---|---|---|---|---|
| –40 | 1.00 | 0.98 | 0.97 | 0.94 | 0.99 | 0.99 |
| –35 | 1.00 | 0.96 | 0.94 | 0.89 | 0.98 | 0.98 |
| –30 | 0.99 | 0.92 | 0.88 | 0.81 | 0.95 | 0.96 |
| –25 | 0.98 | 0.83 | 0.76 | 0.66 | 0.90 | 0.90 |
| –20 | 0.94 | 0.66 | 0.57 | 0.47 | 0.76 | 0.77 |
| –15 | 0.80 | 0.42 | 0.36 | 0.30 | 0.52 | 0.53 |
| –10 | 0.46 | 0.22 | 0.19 | 0.17 | 0.27 | 0.27 |
| –5 | 0.16 | 0.11 | 0.10 | 0.10 | 0.12 | 0.12 |
| 0 | 0.10 | 0.10 | 0.10 | 0.10 | 0.10 | 0.10 |
| 5 | 0.16 | 0.11 | 0.10 | 0.10 | 0.12 | 0.12 |
| 10 | 0.46 | 0.22 | 0.19 | 0.17 | 0.27 | 0.27 |
| 15 | 0.80 | 0.52 | 0.36 | 0.30 | 0.52 | 0.53 |
| 20 | 0.94 | 0.66 | 0.57 | 0.47 | 0.76 | 0.77 |
| 25 | 0.98 | 0.83 | 0.76 | 0.66 | 0.90 | 0.90 |
| 30 | 0.99 | 0.92 | 30 | 0.81 | 0.96 | 0.96 |
| 35 | 1.00 | 0.96 | 0.94 | 0.89 | 0.98 | 0.98 |
| 40 | 1.00 | 0.98 | 0.97 | 0.94 | 0.99 | 0.99 |

Table 6

*Power to detect percentage population change using a four replicate level in year 1 and year 2 at 15 % significance*

| % Population change | Kudu | Wildebeest | Impala | White rhinoceros | Mountain reedbuck | Zebra |
|---|---|---|---|---|---|---|
| –40 | 1.00 | 1.00 | 1.00 | 0.99 | 1.00 | 1.00 |
| –35 | 1.00 | 0.99 | 0.99 | 0.98 | 1.00 | 1.00 |
| –30 | 1.00 | 0.98 | 0.97 | 0.95 | 0.99 | 0.99 |
| –25 | 1.00 | 0.96 | 0.93 | 0.88 | 0.98 | 0.98 |
| –20 | 0.99 | 0.88 | 0.83 | 0.75 | 0.93 | 0.94 |
| –15 | 0.95 | 0.71 | 0.64 | 0.55 | 0.79 | 0.80 |
| –10 | 0.75 | 0.44 | 0.39 | 0.33 | 0.51 | 0.52 |
| –5 | 0.33 | 0.20 | 0.18 | 0.17 | 0.22 | 0.23 |
| 0 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 |
| 5 | 0.33 | 0.20 | 0.18 | 0.17 | 0.22 | 0.23 |
| 10 | 0.75 | 0.44 | 0.39 | 0.33 | 0.51 | 0.52 |
| 15 | 0.95 | 0.71 | 0.64 | 0.55 | 0.79 | 0.80 |
| 20 | 0.99 | 0.88 | 0.83 | 0.75 | 0.93 | 0.94 |
| 25 | 1.00 | 0.96 | 0.93 | 0.88 | 0.98 | 0.98 |
| 30 | 1.00 | 0.98 | 0.97 | 0.95 | 0.99 | 0.99 |
| 35 | 1.00 | 0.99 | 0.99 | 0.98 | 1.00 | 1.00 |
| 40 | 1.00 | 1.00 | 1.00 | 0.99 | 1.00 | 1.00 |

Table 7

*Power to detect percentage population change using a four replicate level in year 1 and year 2 at 90 % significance*

| % Population change | Kudu | Wildebeest | Impala | White rhinoceros | Mountain reedbuck | Zebra |
|---|---|---|---|---|---|---|
| –40 | 1.00 | 1.00 | 0.99 | 0.98 | 1.00 | 1.00 |
| –35 | 1.00 | 0.99 | 0.98 | 0.97 | 1.00 | 1.00 |
| –30 | 1.00 | 0.98 | 0.96 | 0.92 | 0.99 | 0.99 |
| –25 | 1.00 | 0.94 | 0.90 | 0.83 | 0.97 | 0.97 |
| –20 | 0.98 | 0.83 | 0.76 | 0.66 | 0.90 | 0.90 |
| –15 | 0.92 | 0.67 | 0.53 | 0.44 | 0.71 | 0.72 |
| –10 | 0.65 | 0.33 | 0.29 | 0.24 | 0.40 | 0.41 |
| –5 | 0.24 | 0.14 | 0.13 | 0.11 | 0.16 | 0.16 |
| 0 | 0.10 | 0.10 | 0.10 | 0.10 | 0.10 | 0.10 |
| 5 | 0.24 | 0.14 | 0.13 | 0.11 | 0.16 | 0.16 |
| 10 | 0.65 | 0.33 | 0.29 | 0.24 | 0.40 | 0.41 |
| 15 | 0.92 | 0.67 | 0.53 | 0.44 | 0.71 | 0.72 |
| 20 | 0.98 | 0.83 | 0.76 | 0.75 | 0.90 | 0.90 |
| 25 | 1.00 | 0.94 | 0.90 | 0.83 | 0.97 | 0.97 |
| 30 | 1.00 | 0.98 | 0.96 | 0.92 | 0.99 | 0.99 |
| 35 | 1.00 | 0.99 | 0.98 | 0.97 | 1.00 | 1.00 |
| 40 | 1.00 | 1.00 | 0.99 | 0.98 | 1.00 | 1.00 |

wildebeest 44 %, impala 37 %, white rhinoceros 31%, mountain reedbuck and zebra 53 % and 54 %, respectively. This relates to an approximately 50 % probability of correctly concluding that a population has changed by 20 % when it has in reality done so. This, however, relates to unacceptably high Type II error probabilities given for the money invested in aerial counts.

If the significance level is raised to 10 % (Table 3) power decreases to as low as 17 % for white rhinoceros accompanied by a 83 % Type II error probability.

If replicate levels are increased to a 3–3 option in year 1 and year 2 at a significance of 15 % (Table 4) an improvement is evident in the power values attained. The smallest, again in the case for white rhinoceros, is 61 % and this declines to 47 % at 10 % significance (Table 5). At the 15 % significance level (Table 6), all other species show high power values hence acceptably low Type II error risk.

Tables 6 and 7 show large power values (particularly if $\alpha = 0.15$). A case can be made to raise the significance as power levels are acceptable at the 10 % level for the first time.

## Summary and conclusion

The results of these analyses have clarified levels of precision achievable for the standardised helicopter count applied in this case. With regard to the species counted, the team conducting these counts showed a precision that is acceptable within the goals of the management objectives in question. The degree of standardisation adhered to in this case was very high and as such contributed to the precision attained. The precision attained is unique to each species involved and is itself variable with each set of replicates.

The *post hoc* power analysis has demonstrated the calculation of minimum Type II error probabilities from replicated monitoring actions in an applied wildlife management scenario. Biologists and managers can, for similar applications, refer directly to demonstrated minimum Type II error probabilities for pre-selected Type I error probabilities. This has implications in management undertakings in terms of technique optimisation and questions the basis on which management decisions for large ungulates are currently made. Viewed holistically, the results indicate that all counts of this nature should be replicated, allowing a statistical analysis of each data set. In order to offset the costs of replication thought must be given to less frequent yet replicated counts. Detectable change will be larger and the results therefore more statistically robust.

The varying power levels obtained also indicate that the general aerial count for all species is inadequate to meet the set operational goals for each species and consideration must be given to each species in terms of required population change to be detected, thence count design and application.

## Acknowledgements

The authors thank John Vinagre, Piet Muller and Dick Carr for collecting the data and Keryn Adcock for commenting on a draft.

## References

BARTMANN, R.M. L.H. CARPENTER, R.A. GARROTT & D.C. BOWDEN. 1986. Accuracy of helicopter counts of mule deer in pinyon-juniperwoodland. *Wildlife Society Bulletin* 14(4): 356–363.

BEASOM, S. L. 1979. Precision in helicopter censusing of white-tailed deer. *Journal of Wildlife Management* 43(3): 777–780.

BEASOM, S.L., F.G. LEON III & D.R. SYNATZSKE. 1986. Accuracy and precision of counting white-tailed deer with helicopters at different sampling intensities. *Wildlife Society Bulletin* 14(4): 364–368.

BOTHMA, J. DU P., M.J.S. PEEL, S. PETTIT & D. GROSSMAN. 1990. Evaluating the accuracy of some commonly used game-counting methods. *South African Journal of Wildlife Research* 20(1): 26–32.

CAHALANE, V.H. 1938. The annual northern Yellowstone elk herd count. *Transactions of the North American Wildlife Conference* 3: 388–389.

CAUGHLEY, G. 1974. Bias in aerial survey. *Journal of Wildlife Management* 38: 921–933.

CAUGHLEY, G. 1979. *Analysis of vertebrate populations.* New York: Wiley.

COHEN, J. 1988. *Statistical power analysis for the behavioural sciences*. Hillsdale, NJ: Erlbaum.

COLLINSON, R.H.F. 1985. Selecting Wildlife Census Techniques. Monograph 6, Institute of Natural Resources, University of Natal, Pietermaritzburg.

CONQUEST, L. 1983. Assessing statistical effectiveness of ecological experiments: Utility of the coefficient of variation. *International Journal of Environmental Studies* 20: 209–221.

CONQUEST, L. 1993. Statistical approaches to environmental monitoring: Boy, did we teach the wrong thing. *Environmental Monitoring & Assessment* 26: 107–124.

DASMANN, R.F. & A.S. MOSSMAN. 1962. Road strip counts for estimating numbers of African ungulates. *Journal of Wildlife Management* 26: 101–104.

DEYOUNG, C. A. 1985. Accuracy of helicopter surveys of deer in south Texas. *Wildlife Society Bulletin* 13 (2): 146–149.

EISELEN, R. 1994. Estimating the trends in herbivore numbers in the southern district of the Kruger National Park. *South African Journal of Wildlife Research* 24(4): 95–100.

EMSLIE, R.H., S. FOURIE & B.K. REILLY. 1994. A statistical assessment of the utility of replicated aerial strip censusing as a method to monitor a major cycad population. Unpublished report. Ecoscot Consultancy Services. P.O. Box 1212, Hylton, 3245.

FREIMAN, J.A.; CHALMERS, T.C.; SMITH, H. & R.R. KUEBLER. 1978. The importance of beta, the type II error and sample size in the design and interpretation of the randomized control trial: Survey of 71 "negative" trials. *New England Journal of Medicine* 299: 690–694.

GOODMAN, P.S. 1977. An evaluation of three techniques for estimating animal numbers in a wildlife population. Unpublished mimeographed report, Natal Parks Board, Pietermaritzburg.

GRAHAM, A. & R. BELL. 1969. Factors influencing the countability of animals. *East African Agriculture & Forestry Journal,* Special Issue 34: 38–43.

GRAHAM, A. & R. BELL. 1989. Investigating observer bias in aerial survey by simultaneous double-counts. *Journal of Wildlife Management* 53(4): 1009–1010.

HAYES, J.P. & R.J.STEIDL. 1997. Statistical power analysis and amphibian population trends. *Conservation Biology* 11(1): 273–275.

HINDS, W.T. 1984. Towards monitoring of long term trends in terrestrial ecosystems. *Environmental Conservation* II(1): 11–18.

HIRST, S.M. 1969. Road-strip census techniques for wild ungulates in African woodland. *Journal of Wildlife Management* 33 (1): 40–48.

KRAEMER, H. C. 1987. *How many subjects? Statistical power analysis in research.* London: Sage Publications.

LERESCHE, R.E. & R.A. RAUSCH. 1974. Accuracy and precision of aerial moose censusing. *Journal of Wildlife Management* 38: 175–182.

LIPSEY, M.W. 1990. *Design Sensitivity: Statistical Power for Experimental Research.* London: Sage.

MAPSTONE, D. 1995. Scalable decision rules for environmental impact studies: Effect size; Type I & Type II errors. *Ecological Applications* 5(2): 401–410.

MCCULLOUGH, D. R. 1994. What do herd composition counts tell us? *Wildlife Society Bulletin* 22(2): 295–300.

MELTON, D.A. 1978. Undercounting bias of helicopter census in Umfolozi Game Reserve. *Lammergeyer* 26: 1–6.

NORTON-GRIFFITHS, M. 1978. Reducing counting bias in aerial census by photography. *East African Wildlife Journal* 12: 245–248.

OTTEN, M.R.M., J. B. HAUFLER, S. R. WINTERSTEIN & L. C. BENDER. 1993. An aerial censusing procedure for elk in Michigan. *Wildlife Society Bulletin* 21(1): 73–80.

PEARSON, E.S. & H.O. HARTLEY. 1976. *Biometrika tables for statisticians.* Norfolk: Cambridge University Press.

PAYNE, N. F. 1981. Accuracy of aerial censusing for beaver colonies in Newfoundland. *Journal of Wildlife Management* 45(4): 1014–1016.

PENNYCUICK, C.J. & D. WESTERN. 1972. An investigation of some sources of bias in aerial transect sampling of large mammal populations. *East African Wildlife Journal* 10: 175–191.

PETERMAN, R.M. 1990. Statistical power analysis can improve fisheries research and management. *Canadian Journal of Fisheries and Aquatic Science* 47: 2–15.

REED, J.M. & A.R. BLAUSTEIN. 1995. Assessment of non-declining amphibian populations using power analysis. *Conservation Biology* 9: 1299–1300.

RICE, W.R. & J.D. HARDER. 1977. Application of multiple aerial sampling to mark—recapture census of white-tailed deer. *Journal of Wildlife Management* 41: 197–206.

ROTHPEARL, A.B., R.C. MOHS & K.L. DAVIS. 1981. Statistical power in biological psychiatry. *Psychiatry Research* 5: 257–266.

SEBER, G.A.F. 1992. A review of estimating animal abundance II. *International Statistical Review* 60(2): 129–166.

THERON, G.K. 1978. Plantegroei tipes van Loskopdam Natuurreservaat. Unpubl. D.Sc. thesis, University of Pretoria.

VAN HENSBERGEN, H.J., M.P.S. BERRY & J. JURITZ. 1996. Helicopter based line transect estimates of African herbivore populations. *South African Journal of Wildlife Research* 26(3): 81–87.

VAN LAVIEREN, L.P. & J.D. ESSER. 1979. Numbers, distribution and habitat preference of large mammals in Bouba Ndjida National Park, Cameroon. *African Journal of Ecology* 17: 141–153.