



Guidelines for designing population surveys of burrowing petrels

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SUMMARY

Robust population estimates are needed for conservation management of burrowing petrel populations. Estimates of population size for burrowing petrels are often obtained by extrapolation of surveys in a small proportion of the available habitat to a population- or island-wide scale. However, extrapolation may also amplify bias or error, giving rise to potentially large error bounds and imprecise estimates of population size. This hinders risk assessments for these species, and limits the ability to detect trends in population size over time. We review methods for estimating the breeding population size of burrowing petrels by extrapolation from surveys, focusing in particular on the error associated with population estimates of the larger petrels (including *Procellaria* spp.).

Sources of error in the extrapolation of survey data are divided into five key areas: (1) uncertainty of burrow contents (2) timing, (3) availability bias, (4) burrow detection probability, and (5) observer bias.

The review highlights that there is no single method for minimising error levels in population estimates. Rather, the most accurate and precise studies are those designed according to the specifics of each study in terms of resources available, target species and site characteristics, and we discuss a range of the factors that are important to consider.

The timing of burrow occupancy checks can help minimise assumptions about what proportion of breeding birds has not yet laid or has already failed. If sampling sites are not representative, or if part of the occupied habitat is not accessible, this availability bias can affect extrapolation. Extrapolation errors occur when the area sampled is not representative of the total area occupied. Burrow detection rates can also affect the accuracy of extrapolation, so the assumption that every burrow in the sampled area was detected should be checked. To produce an accurate estimate of breeding population size from burrow counts, it is important to determine burrow contents, and to distinguish between breeding and non-breeding birds in burrows. If a proportion of occupants is missed, further error is introduced to the population estimate, so it can be valuable to check detection probability. Whether planar map area or true surface area is used for extrapolation can be a further source of error. It is also important to test for differences between observers in their ability to detect burrows or burrow contents.

Several key points are relevant to all studies: the need for a robust pilot study to identify and measure sources of error in the main survey; the need for sufficient time to adequately sample the available habitat, while including contingency for weather; and the need to document survey methods in enough detail to be repeatable.

For reference, we summarise the key problems associated with extrapolation of burrowing petrel surveys, and the implications if these error sources are not addressed (**Table 2**).

INTRODUCTION

Understanding demographic parameters and population dynamics is critical for the conservation management of burrowing petrel populations, and robust data are required to estimate these parameters. However, since burrowing petrels nest in underground burrows and cavities, frequently on remote islands that are hard to access, obtaining good quality data is often much more difficult than for surface-nesting seabirds (Brooke 2004; Schumann et al. 2013).



Historically, subjective population estimates were often made from brief visits to colonies (Bailey and Sorenson 1962; Taylor 1988), but this approach has largely been replaced by attempts to obtain quantitative data to estimate population size (Rayner et al. 2007a; Ryan et al. 2012; Bell et al. 2013a; Whitehead et al. 2014). Estimates of population size for burrowing petrels are often derived from extrapolation of burrow surveys from a proportion of a population to a population- or island-wide scale. This typically involves surveying active nests, obtaining a representative sample of burrow density via transects or plots, then correcting that density estimate for burrow occupancy and applying it to the available area of nesting habitat (e.g., Burger and Lawrence 2001; Lawton et al. 2006; Baker et al. 2008; Lavers 2015).

However, extrapolation may amplify bias or error, giving rise to potentially large error bounds and imprecise estimates of population size. Wide error margins hinder risk assessments for threatened species and limit the ability to detect trends in population size over time (e.g., Oppel et al. 2014). Trends may not be detected if population estimates lack precision ($\pm 25\%$ or more), or unless the changes in breeding numbers are large, because the error margins of successive estimates will most likely overlap (R. Phillips pers. com.).

To reduce the error margins around population estimates, it is vital to consider potential sources of error and develop methodologies to lessen their effects on estimates. There are three general categories of error associated with estimates of population size: temporal variability, spatial variability and detection probability (Wolfaardt and Phillips 2011). We divide these into five key areas for review and discussion below: (1) uncertainty of burrow contents (2) timing, (3) availability bias, (4) burrow detection probability, and (5) observer bias.

We review field methods for estimating the breeding population size of burrowing petrels, aiming to identify key sources of error that impact accuracy and precision when survey data are extrapolated. Specifically, we focus on data for point estimates of population size rather than for trend estimation.

Studies on the largest burrowing petrels (*Procellaria* spp.) form the backbone of this work, but we also draw widely from literature on other seabird species.

Table 1. Definitions of key terms used in this text (in **bold** first time used in text)

	Definition
Burrow density	the number of burrows per unit area
Burrow detection probability	the number of burrows counted as a proportion of the number available, expressed as a rate
Burrow occupancy	proportion of burrows that contain the seabird species of interest, expressed as a rate
Detectability	the ease with which burrows are found (see <i>burrow detection probability</i>)
False-negative error	the number of burrows where an occupant was missed, expressed as a rate
False-positive error	the number of burrows where a burrow is incorrectly classed as occupied, expressed as a rate
Grubbing	feeling for a burrow occupant by hand or with a probe
Habitat availability	the proportion of petrel habitat available for sampling
Main laying period	when all birds except a few very late birds have laid and begun incubating
Occupant detection	the probability of correctly determining burrow contents, expressed as a probability
Observer bias	bias resulting from differences among observers
Planar area	one-dimensional area of a landscape as on a map (cf. <i>surface area</i>)
Representativeness	the extent to which sampled areas are representative of the habitat to which a survey is extrapolated
Surface area	three-dimensional surface area of a landscape (cf. <i>planar area</i>)
Timing	the period when the survey is undertaken relative to the breeding phenology of the target species

BURROW CONTENTS

Burrow occupancy

To produce an accurate and precise population estimate from burrow counts, it is important to accurately determine the proportion of burrows that contain the seabird species of interest (**burrow occupancy**), distinguishing breeding birds from non-breeding or failed birds. Burrow occupancy is typically assessed from a sub-sample of burrows, with few exceptions (Barbraud et al. 2009; Rexer-Huber et al. 2014). The most widely used tools to determine burrow occupancy are the burrowscope, e.g., Lavers 2015; Waugh et al. 2015), call-playback response (e.g., Barbraud et al. 2009; Soanes et al. 2012) or feeling for an occupant by hand/with a probe (**grubbing**) (e.g., Schulz et al. 2005). Some studies use inspection hatches (e.g., Waugh et al. 2006; Bell et al. 2009), and many use a combination of techniques like burrowscoping with excavation (Lawton et al. 2006), playback with grubbing (Burger

and Lawrence 2001; Ryan and Ronconi 2011; Dilley et al. submitted) or grubbing and digital cameras (Baker et al. 2008). Most studies use direct inspection methods like these that rely on detection of bird(s) in a burrow, but indirect methods (e.g., using activity sign at burrow entrances) are still occasionally combined with other methods (Ryan et al. 2012; Schumann et al. 2013). Each method has limitations, and these limitations drive the choice of which is most appropriate for a particular species.

Response to playback of calls appears to be useful for some species, and minimally so for others. For example, a high proportion of white-chinned petrels responded to playback (Berrow 2000; Barbraud et al. 2008; Barbraud et al. 2009), but less than a quarter of Atlantic petrels *Pterodroma incerta* responded to playback (Rexer-Huber et al. 2014). Response rates can vary within species (18–70%; Ratcliffe et al. 1998; Vaughan and Gibbons 1998), over a breeding season (Ryan et al. 2006), and even within individuals (Berrow 2000). The probability of a bird responding is influenced by factors including breeding condition, sex of bird in burrow, time of day, and playback features like the range of calls used, duration and volume (Berrow 2000; Burger and Lawrence 2001; Barbraud and Delord 2006; Soanes et al. 2012). However, even when playback reliably indicates occupancy, the presence of non-breeding birds can obscure true breeding numbers. Since population estimates typically hinge on knowing whether birds present in burrows are breeding, methods that visually inspect the contents of nesting chambers, like a burrowscope or inspection hatches, are critical.

Burrowscopes work particularly well for species with relatively large, straight burrows like those of Westland petrels *Procellaria westlandica* (Waugh et al. 2003) and white-chinned petrels *Procellaria aequinoctialis* (Rexer-Huber et al. 2015). Burrows can be inspected in their entirety with confidence that no branches or chambers were missed, and the small number not fully inspected reported as inaccessible and removed from calculations (e.g., Cuthbert 2001; Rexer-Huber et al. 2014). Complex burrows are more difficult to inspect fully by



burrowscope; for example, some species have entrance moats, multiple entrances and chambers, collapsed chambers, and deviating tunnels (Hamilton 2000; McKechnie et al. 2007; Newman et al. 2008; Cuthbert et al. 2013; Dilley et al. submitted). Moats can be spanned by adding a firm but bendable splint to the burrowscope cable, but highly curved, tight or deviating burrows can be harder to inspect in full. For example, burrow occupancy was underestimated by up to 17% when a burrowscope was used to check the typically long, convoluted burrows of sooty shearwaters *Puffinus griseus* (McKechnie et al. 2007), and by 10–19% in Hutton's shearwater *Puffinus huttoni* burrows (Cuthbert 2001; Cuthbert and Davis 2002).

For some species, burrows can be accessed by hand or with a flexible probe (burrow grubbing) (e.g., Schulz et al. 2005; Baker et al. 2008; Ryan and Ronconi 2011; Schumann et al. 2013), especially when progressive access holes are dug to allow full inspection of the nesting chamber (G. Taylor pers. com.). This technique is widely used for its ease and minimal equipment needs, and there is no ambiguity about burrow contents when access is by hand (e.g., Barbraud and Delord 2006). However, grubbing with a probe is less accurate, particularly during incubation, and access to deeper or more convoluted burrows is often not possible by grubbing.

A number of other techniques are used to inspect burrow contents. Access hatches dug into or closer to the nesting chamber are sometimes used to assess burrow occupancy; typically, these are long-term study burrows established in successful, occupied burrows (e.g., Cuthbert 2001; Rayner et al. 2008; Bell et al. 2009), which may introduce positive bias in occupancy (Hunter et al. 2001). Digging access holes may be disruptive to an unknown extent (Hamilton 1998; Ryan et al. 2006), depending on the species and the timing. Some burrowing seabird species are more prone to deserting their nests when disturbed during early incubation (Warham 1990), although to our knowledge there are no records of desertion as a result of study during incubation for *Procellaria* species.

Occasionally, digital cameras are used to view burrow contents (Baker et al. 2008). Dogs trained in detecting seabirds can be helpful for both burrow surveys and detecting occupancy (Bell et al. 2013b; G.B. Baker pers. com.), but a well-trained dog and its handler are costly. Since the detection ability of dogs differs like that of people, the accuracy of a given dog must be calibrated (G.B. Baker pers. com.). Other burrow occupancy techniques being tested are the use of heat sensors (E. Bell pers. com.) and CO₂ sensors (S. Waugh pers. com.).

Occupant detection

Burrow occupancy may vary on both spatial and temporal scales (Berrow 2000; Lawton et al. 2006; McKechnie et al. 2009; Baker et al. 2010; Whitehead et al. 2014). Spatial variation can be minimised by using representative sites for sampling burrow occupancy (e.g., Sutherland and Dann 2012; Whitehead et al. 2014). Occupancy rates will best reflect the maximum number of breeding pairs early in the breeding season, just after laying has finished. However, temporal variation in occupancy rates may become more important when it is not possible to be present at that point in time (Sutherland and Dann 2012).

It is important to consider the assumption that occupants are always detected. **False-negative errors**, arising from the failure to record an occupant when it is present, can potentially occur under any of the burrow occupancy methods discussed here. For some species like sooty shearwaters, inaccuracy from failure to detect some birds was greater than between-observers and between-sites sources of error (McKechnie et al. 2007). There is a large literature on response rate correction for call playback estimates of burrow occupancy (e.g., Burger and Lawrence 2001; Barbraud and Delord 2006; Ryan et al. 2006; Soanes et al. 2012; Rexer-Huber et al. 2014), and playback is rarely used without accounting for response rate. However, the probability of detecting burrow occupants (**occupant detection probability**) is less frequently checked in studies applying other methods; for example, less than half of studies using burrowscopes checked for occupant detection rates. Occupant detection is assessed in a smaller proportion of *Procellaria* studies reviewed than in those on other burrowing petrel species. This may be due to the large, simple burrows dug by some *Procellaria* relative to those of smaller burrowing

petrel species, which ease wall-to-wall inspection by burrowscope (Vaugh et al. 2003). With smaller and more complex burrows, features that impede detection are more obvious, which may explain why their impact on occupant detection is more frequently accounted for.

Occupant detection probability is most commonly assessed by variants of resampling methods: two observers checking the same sub-sample of burrows for occupants (e.g., Whitehead et al. 2014), or via repeated checks of the same burrows over a time period (MacKenzie et al. 2006; Ryan et al. 2006; Bailey et al. 2007; Baker et al. 2008; Rayner et al. 2009). Occupant or playback detection can also be assessed against a different inspection method, such as burrowscoping (Barbraud et al. 2008; Barbraud et al. 2009), grubbing (Ryan et al. 2012) or excavation / study hatches (Barbraud and Delord 2006; McKechnie et al. 2007; Baker et al. 2008; Baker et al. 2010). Detection is generally calculated simply as the probability of detecting an occupant, but may be estimated via a form of capture-mark-recapture (CMR) modelling where burrows represent ‘individuals’ and the presence of an occupant is treated as a ‘capture’ (Sutherland and Dann 2012; Whitehead et al. 2014). It is worth emphasising that to gauge occupancy, apparently unused burrows also need to be inspected. This is particularly relevant for those species where burrows entrances can quickly become covered with leaf-litter or mud, making occupied burrows appear inactive.

On the other hand, **false-positive errors** arise when a burrow is incorrectly classed as occupied. False-positives might occur during direct inspection due to mistaken identity, i.e. when multiple species occur at a site (Rayner et al. 2008; Cuthbert et al. 2013; Whitehead et al. 2014), or when non-breeding birds are included in breeding bird totals. False-positive errors are thought to be less prevalent than false-negative errors (Tyre et al. 2003), mainly because mistaken identity or breeding status should rapidly become rare with even small amounts of observer experience and training. False positives are also minimised by the use of direct inspection tools rather than indirect activity-sign based methods (Rexer-Huber et al. 2014).

TIMING

Strictly speaking, temporal variability is a process resulting from variation in demographic parameters, not an error source, and can be detected using estimates of population size at different times (C. Barbraud pers. com.). However, when a burrowing petrel population is infrequently surveyed, **timing** should be treated as an important source of error in estimating population size and accounted for in study design. Timing has particular relevance for determining burrow contents, while burrow entrance counts are less sensitive to this issue (Ryan et al. 2012). **Burrow occupancy** data collected at the most appropriate time in the breeding season can help avoid assumptions about what proportion of breeding birds has not yet laid or has already failed. For burrowing seabirds, the contents of burrows is ideally assessed immediately following the **main laying period**, when all birds except perhaps a few very late birds have laid and begun incubating (Lawton et al. 2006; Schumann et al. 2013; Defos du Rau et al. 2015). Surveying too early in the egg-laying period will miss a proportion of breeders that have not yet attempted to breed and thus underestimate the breeding population (Sutherland and Dann 2012). Monitoring burrows after early incubation may also underestimate the year’s breeding attempts, since failures prior to survey may remain undetected (Hunter et al. 2001). There may also be fewer non-breeders occupying burrows after egg-laying is complete than before and during egg-laying (B. Dilley, pers. com.), helping to minimise **false-positive errors**.

A number of approaches are used to lessen the impact of conducting burrow surveys outside of the breeding season. For example, Ryan et al. (2012) conducted an island-wide survey of white-chinned petrels at the end of the breeding season as more workers were available during this time, but collected occupancy data during the main laying period. Westland petrel surveys were carried out over a four year period but occupancy was assessed during early incubation (Wood and Otley 2013). Demographic data from previous work can help estimate and account for failure rates during incubation (e.g., Cory's shearwater *Calonectris diomedea*, Granadeiro et al. 2006).

Frequently, data on the timing of the laying period are not available. When demographic studies only exist for some populations, it may be possible to make inferences from other populations. For example, lay dates for white-chinned petrels from the well-studied population on Bird Island, South Georgia (Hall 1987; Berrow 2000) were used to guide survey planning for the Auckland Island population (Rexer-Huber et al. 2016). Laying data exist for most *Procellaria* petrels (Zotier 1990; Waugh et al. 2006; Barbraud et al. 2008; Bell et al. 2013b). Ideally a pilot study aimed at estimating the main laying period should be conducted the year prior to burrow surveys. For species where this is not possible and there has not been previous demographic study, we recommend using all available information to predict the likely main laying period. This may include anecdotal records and descriptions from workers with experience at the location.

Some species may not have a clear laying period, it may vary among populations, or it may change from year to year. For example, the lay dates of some white-chinned petrel populations can vary, driven by late-melting snow and ice cover in some years preventing birds from accessing burrows (Berrow 2000). Grey petrels *Procellaria cinerea* on Marion Island start laying two weeks earlier than grey petrels on Gough Island (B. Dilley pers. com.). The lay period of black petrels *Procellaria parkinsoni* on Great Barrier Island also varies from year to year, and they lay over an extended



Grey Petrel chick

Photo courtesy of Ross Wanless

period of six weeks (E. Bell pers. com.). To account for this, researchers collect population estimate data during three visits to the study colony (Bell et al. 2009). An alternative method for species that have asynchronous laying is a capture-mark-recapture (CMR) study on a sample of burrows to collect information on occupancy, and using these CMR data to estimate the total number of burrows occupied in a given year (Williams et al. 2011). Although methods for investigating population trends is not an objective of this review, it is worth noting that consistency in the timing of surveys between years is important if population estimates are to be compared (Bell et al. 2009; Rexer-Huber et al. 2014).

Other aspects of timing are important to consider; for example, whether investigator disturbance is greater at a particular stage in the breeding season. This is often discussed in the context of

shearwaters (Warham 1990), or focuses on the impacts of handling (Carey 2009). However, to our knowledge disturbance to *Procellaria* petrels from burrow inspection that does not involve handling (burrowscope, access hatch, acoustic playback) has not been recorded. Weather may also be important: the number of non-breeding birds can fluctuate greatly from one day to the next, depending on weather- or food-driven attendance cycles (R. Phillips pers. com.). Ryan et al. (2006) ascribed an increase in burrow occupancy to windy and misty conditions causing an influx of non-breeding and prospecting birds. This illustrates the advantage of collecting occupancy data over a number of days in order to identify and sample such phenomena.

AVAILABILITY BIAS

Habitat availability

An availability bias results from some part of a petrel's burrowing range not being accessible, or available, to sample (Wolfaardt and Phillips 2011), creating problems for extrapolation. Despite this, relatively few studies report the proportion of habitat that could not be surveyed (but see Catry et al. 2003; Barbraud and Delord 2006).

Burrowing petrels frequently inhabit remote, mountainous and steep terrain, areas with difficult vegetation, or areas that are simply too fragile to access (Granadeiro et al. 2006; Baker et al. 2008; Oppel et al. 2014; Lavers 2015). If that terrain is not available for sampling, a sizeable proportion of the population may go unaccounted for. For example, an unknown proportion of grey petrels on Antipodes Island nest on cliffs, so Bell et al. (2013a) used dawn, day and dusk observations from vantage points to obtain presence-absence



White-chinned Petrel habitat, Adams Island

Graham Parker

data. Spotlighting or acoustic surveys are also sometimes used to determine whether and/or which species are utilising inaccessible sites (Bell 2002; G. Taylor pers.com). It may be possible to calibrate vantage point, spotlighting or acoustic surveys against areas where quantitative data were collected (Oppel et al. 2014). If an important proportion of the population inhabits difficult terrain, presence-absence surveys would ideally be followed by access to at least some areas to quantify burrow numbers and estimate occupancy. In the case of cliffs, appropriate rope-access equipment could be

used to survey the area (e.g., Defos du Rau et al. 2015), providing nests are not damaged or destroyed.

Fragile ground can also affect the ability to re-sample areas. For example, Ryan et al. (2006) assessed the accuracy of single compared to multiple burrow occupancy checks, but then limited repeat visits to colonies because of the risk of collapsing burrows during each visit to a colony. In very fragile areas, possible solutions are boards or plywood squares to distribute weight during surveys, or access paths developed while breeding birds are absent from an island (Baker et al. 2010; G.Taylor pers. com.). However, this is clearly not always possible or the impact and effort justifiable, and other tools like acoustic monitoring may be necessary (Oppel et al. 2014).

Representativeness

Extrapolation errors occur when the area sampled was not representative of the area that the samples are extrapolated to (Hunter 2001; Baker et al. 2010). Ensuring that sampling areas are fully representative of the extrapolation area can be a particular challenge when working on patchily-distributed species. A truly randomised sampling design may be representative but not encounter any burrows, even when present, making it necessary to target burrowed areas for sampling (e.g., Rayner et al. 2008; Parker et al. 2015; E. Bell pers. com.). However, a targeted sample is clearly not representative of adjacent areas, so simple extrapolation would not provide meaningful numbers.

A number of approaches have been used for representative sampling of patchy distributions. If every colony can be identified, non-colony areas could be excluded from extrapolation. For example, in an area with good prior knowledge of colony distribution, colonies can be sampled (or if small enough, counted in their entirety) and the **burrow density** extrapolated specifically to the colony area (Burger and Lawrence 2001; Baker et al. 2008; Rayner et al. 2008; Baker et al.



2010; Sutherland and Dann 2012). If burrows occur according to a pattern – perhaps at greater density above 300 m, or on flatter terrain with drier soil, or in a particular forest type – then a stratified design may reduce extrapolation error (Cuthbert 2004; Lawton et al. 2006; Rayner et al. 2007a; Charleton et al. 2009; Francis and Bell 2010; Bell et al. 2011), but not always (Schumann et al. 2013). Stratification can also be important if dealing with very large or spatially discontinuous geographical areas; for example, on Kerguelen or South Georgia islands (Barbraud et al. 2009; Martin et al. 2009). This ensures that samples collected within a particular area, or stratum, are extrapolated only to the area of that stratum, and those summed for the overall population estimate. However, useful patterns are not always available to guide stratification, or produce strata with few burrows and result in large error bounds. A systematic study design can be useful to improve precision in these circumstances (Buckland et al. 2001; Fewster et al. 2009). For example, a small island known to contain a dense

cluster of burrows along one edge and few elsewhere could be sampled with systematic transect lines (the sampling unit) spanning the low- to high-density gradient (Fewster et al. 2009). More patchily-distributed species may require more closely-spaced systematic transects, increasing survey effort (Lormée et al. 2012; Schumann et al. 2013). Alternatively, an adaptive cluster sampling design (Thompson 1991) can be useful to capture relatively rare species whose burrows are distributed in patches (C. Barbraud, pers. com.). This is not an exhaustive list of study design options (see Borchers et al. 2002; Morrison et al. 2008 for thorough review). Rather, we discuss the more commonly-used approaches in the burrowing petrel literature.

BURROW DETECTION

Extrapolation errors commonly arise when burrow counts under- or overestimate actual burrow numbers, and potentially when extrapolation utilise planar map areas rather than true landscape **surface areas**.

Burrow detection probability

Accurate extrapolation depends on the assumption that every burrow in the sampled area was detected. If a number of burrows are missed during counts, extrapolated burrow numbers will be inaccurate to an unknown extent. Correcting for missed burrows may increase the variance around an estimate, with the advantage that this variance is more likely to overlap the true population size (Sutherland and Dann 2012).

The ease with which burrows are detected, or **detectability**, may vary with factors like vegetation, topography and burrow size (Lawton et al. 2006; Ryan et al. 2006; Rayner et al. 2007b; Barbraud et al. 2009). Therefore it follows that the likelihood of missing some burrows also varies (Francis and Bell 2010). Sampling design may also have an influence, since it is easier to miss some burrows in large plots or wide transects, for example, than in smaller plots. Even direction of travel can affect detectability: burrows are typically easier to detect when moving upslope than down in some habitats (Lawton et al. 2006; Rexer-Huber et al. 2015), a factor to consider if, for example, transects are oriented at random with respect to slope.

Even a moderate, realistic **false-negative rate** (the converse of detection rate) can have a large effect on study conclusions (Tyre et al. 2003; Archaux et al. 2012), particularly on the effect of habitat on site occupancy (e.g., whether burrows are detected at that site or not).

Burrow detection probability is often derived from repeat surveys, with all sites surveyed several times over a visit or season, or via repeat surveys at a subset of sites (MacKenzie et al. 2006). A double-observer approach may be appropriate to determine burrow detection probability, where other observer/s revisit a subset of transects or plots (Lawton et al. 2006; Baker et al. 2008; Baker et al. 2010; Lormée et al. 2012; Defos du Rau et al. 2015). This assumes that the detection probability of the subset is the same at the other plots (MacKenzie et al. 2006), so care is required that this subsample is representative. The accuracy of sampling methods can be tested within small colonies or areas by exhaustive strip-search counts to determine the true burrow number (Barbraud et al. 2009; Ryan et al. 2012). In turn, exhaustive strip-search counts can be checked for detection probability using validation transects run in a different direction and angle to search strips (Parker et al. 2015). Some approaches

to estimate burrow numbers explicitly take detection probability into account, like capture-mark-recapture and distance sampling methods (Buckland et al. 2001; Morrison et al. 2008). Distance sampling is becoming more widely used for burrowing petrels (Lawton et al. 2006; Barbraud et al. 2009; Defos du Rau et al. 2015; Rexer-Huber et al. 2015). This uses the distance from a line to an object to account for objects becoming less detectable with distance, and incorporates this detection probability into estimates of burrow numbers (Buckland et al. 2001).

Extrapolation area

Another source of error when extrapolating samples of burrow counts is area; in particular, whether **planar area** (one-dimensional area as represented on a map) is used, rather than true **surface area** (three-dimensional surface area of a landscape). If counts within a given sampling area (plot diameter or transect width) on the landscape are extrapolated by the planar/map area of the study (Bell 2002; Sommer et al. 2010), true numbers may be underestimated (Jenness 2004). This is less of an issue when extrapolating counts based on planar sampling area to other planar areas; for example, if the sampled area is line length measured by GPS (Jamieson and Waugh 2015; Waugh et al. 2015; but see Baker et al. 2008). Calculating line length this way may however introduce measurement error from the GPS, which is rarely reported (but see Waugh et al. 2015).

A range of approaches can be used to correct planar map area to reflect true landscape surface area. Repeated slope measurements were used to estimate slope-corrected surface areas (Newman et al. 2008; Barbraud et al. 2009; Rexer-Huber et al. 2016); topographic contours used find the slope with which to correct planar area (Martin et al. 2009); and digitised contours, spot-height data or extensive GPS tracks are used to create digital elevation models in geographic information systems software (Catry et al. 2003; Lawton et al. 2006; Rayner et al. 2007b; Rayner et al. 2008; Defos du Rau et al. 2015).

OBSERVER BIAS

Producing standardised and replicable results is a critical aspect of animal surveys, yet observers differ (MacKenzie et al. 2006). Observers may differ in their ability to detect burrows or burrow contents, to reach birds, or to estimate burrow numbers in a discrete area (Cuthbert 2001; Cuthbert and Davis 2002; Ryan and Ronconi 2011; Lormée et al. 2012), and this difference should be tested for (MacKenzie et al. 2006; Morrison et al. 2008). Those observers with longer arms may have better bird detection rates in some burrow types, and observer experience can play a role in some methods to determine occupancy. Similarly, as observers could become more experienced over time, there may be a need to account for this within-observer source of bias.

The most common way to calibrate the reliability of observers is through the use of multiple counts (Morrison et al. 2008). For example, multiple counts were conducted along a proportion of transects for Westland petrels (Baker et al. 2008) and flesh-footed shearwaters *Puffinus carneipes* (Baker et al. 2010). Although neither study showed a difference among observers, testing for bias can increase confidence in the results. Scan counts of the number of burrows in discrete areas always showed differences among observers when checked against thorough ground counts (Ryan and Ronconi

2011). Counts of spectacled petrels *Procellaria conspicillata* differed by as much as 2–5% between observers in any given year (Ryan and Moloney 2000; Ryan et al. 2006; Ryan and Ronconi 2011), and observers of white-chinned petrels on Marion Island consistently under and over-estimated the number of burrows (81–123%) (Ryan et al. 2012).

A number of approaches help minimise observer bias. Training is a logical approach to ensure that search effort is consistent across workers. Observers may vary in their perception of key measures like transect width or plot radius, but this measurement error can often be addressed with simple tools (e.g., tape, length of line, plastic pipe) (Morrison et al. 2008). When surveys are compared over years, having leaders with experience can help standardise methodologies between years (Ryan and Ronconi 2011; Bourgeois et al. 2013; Dille et al. submitted). Furthermore, it may be useful for a single observer to do aspects of the work that may be more sensitive to observer error and experience, such as burrow inspection by burrowscope (Cuthbert 2001; Parker et al. 2015; S. Waugh pers. com.).

RECOMMENDATIONS

Burrowing petrel surveys are time-consuming, labour intensive and often time-limited by funding and logistics (Schumann et al. 2013). It is important to design surveys that will produce accurate and precise estimates despite these constraints.

There is no single best method for minimising error in population estimates; rather, the most accurate and precise studies are those designed according to the specifics of the study site, available resources and target species.

It is widely accepted that population estimates, usually expressed as numbers of breeding pairs, requires that counts of burrows be corrected by the proportion of burrows that contain a breeding pair (burrow occupancy rate). A range of tools are used to check burrow contents, but are not equally useful for all burrowing petrel species. That is, the accuracy and precision of a tool (e.g., burrowscope, or acoustic playback) varies according to species and breeding site. To reduce extrapolation errors, it is critical to test the assumption that every occupant will be found (occupant detection probability) and that breeders can be accurately distinguished from non-breeders.

A good pilot study that tests sampling techniques and determines minimum sample sizes can be a valuable tool for reducing the variance around population estimates (e.g. Defos du Rau et al. 2015), particularly if time is limited. A pilot study may also allow the main laying period to be estimated.

Burrow detection probability, testing the assumption that every burrow in an area is found, can have a large impact on accuracy of population estimates. Surprisingly, less than a third of studies reviewed accounted for burrow detection. The appropriate method to determine burrow detection is species- and site-dependent, but can involve simple repeat surveys or checking counts against other sampling methods. Alternatively, methods that explicitly take detection probability into account (e.g., variants of capture-mark-recapture or distance sampling) may be appropriate. Burrowing petrel estimates can also be influenced heavily by the area used for extrapolation. Seabird colonies with variable topography or elevation may require an area correction to ensure

that available surface area is not underestimated.

Observer bias may introduce unnecessary variance around a point estimate of population size. The assumption that well-trained and experienced observers will be consistent (Baker et al. 2008) does not always hold (Ryan and Ronconi 2011), so it is important to check burrow count and contents data for observer differences.

For reference, we summarise the key problems associated with extrapolation of burrowing petrel surveys, and the implications if these error sources are not addressed (**Table 2**).

Table 2. Error sources in extrapolation of burrowing petrel surveys to obtain population estimates: problems and implications if not addressed

Factor	Problem	Potential implications
Burrow contents		
<i>Accessibility</i>	Occupancy of burrows with unconfirmed contents may not be the same as burrows where the contents can be confirmed	Under or overestimate of burrow occupancy in breeding population estimate
<i>Occupant detection</i>	Incorrectly assign occupied burrow as empty	Underestimate burrows occupied by breeding birds
	Non-breeding birds included in breeding bird totals	Include non-breeding burrow occupants in breeding population estimate; overestimate number of breeding pairs
	Mistaken identity	Include other burrowing seabird species in breeding population estimate
Timing		
	Burrows not occupied by the majority of the season's breeding pairs	Under or overestimate population size depending on breeding phenology
	Detection probability varies with time	Detection probability for some survey methods affected (response to taped playbacks, sign)
	Some species may be more disturbance-prone during incubation	Detrimental disturbance to study species
	Inter-annual surveys conducted at different times of breeding season	Between year estimates not comparable
Availability bias		
<i>Habitat availability</i>	Incorrectly assume birds are present or absent in the non-surveyed habitat	Decreased accuracy of population estimate
	Assume burrow density or occupancy consistent across habitat	Decreased accuracy and precision of estimate
<i>Representativeness</i>	Sampled area not representative of the area that samples are extrapolated to	Extrapolation error resulting in poor accuracy and precision
	Incorrect or no stratification of sampled areas when required	Estimate less precise
Burrow detection probability		
	Burrows that are present are not detected	Underestimate burrow numbers
		Increased variance from correction factor for missed burrows
Observer bias		
	Burrows not detected	Decreased precision due to difference in observers data
	Burrows of different species not accurately discerned	Decreased precision due to difference in observers data
	Burrow occupants not accurately identified	Decreased precision due to difference in observers data
	Distances (e.g., transect width) estimated differently	Decreased precision due to difference in observers data

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