

Chapter 10

Taxon-Specific Issues

In this chapter, we consider some of the problems that may be encountered for specific taxa. We make no attempt to be comprehensive; rather, we have selected a few taxa to illustrate the types of issue that arise in some circumstances. The absence of a taxon from this chapter does not indicate that distance sampling methods are inappropriate for that taxon.

10.1 Songbirds

In this section, we discuss some of the issues that arise in breeding season surveys of songbirds. We draw on material from Buckland et al. (2008), and we make frequent reference to the Montrave case study (Buckland 2006).

10.1.1 Line Transect Versus Point Transect Surveys

When both line transect and point transect sampling are feasible, then line transect sampling tends to give higher precision (Buckland 2006). This is because we need to estimate the probability density function (line transect sampling) or the slope of the probability density function (point transect sampling) at distance zero (*i.e.* $f(0)$ or $h(0)$ respectively). For line transect sampling, a higher proportion of recorded distances is close to zero than for point transect sampling, simply because there are more birds available to be detected close to a line than to a point, and so we have better information for estimating $f(0)$ than we do for $h(0)$. A second reason for the higher precision of line transect sampling is that a higher proportion of time spent in the field is spent actively searching for birds; time travelling between lines is typically appreciably less than time travelling between points because there tend to

be many more points in a point transect survey than lines in a line transect survey. Further, there tends to be less bias arising from bird movement (Sect. 10.1.2) or from inaccurate distance estimates (Sect. 10.1.4) for line transect sampling than for point transect sampling.

To offset these advantages, there are several disadvantages in using line transect sampling. First, we would like to have a random design for our survey, to avoid bias. In difficult terrain, or study areas with access difficulties, it tends to be easier to reach a random point than to traverse a random line. If a completely randomized design is not feasible, the compromises are likely to be fewer, and hence the bias lower, for point transect sampling than for line transect sampling. Second, if it is important to assess how density depends on habitat, it is much easier to quantify habitat at a point than along a line. Third, for songbird surveys, it is common for many species to be recorded. For observers attempting to detect and record a number of individuals from various species, it is easier if they are stationary at a point than if they also have to concentrate on walking along a randomly-located line in potentially difficult terrain.

10.1.2 Minimizing Bias from Bird Movement

10.1.2.1 Movement Independent of the Observer

Conceptually, we think of animals frozen in location while we survey them. When this is not the case, density tends to be over-estimated. For line transect sampling, bias is slight provided average speed of the animals is under half that of the observer (see Fig. 11.8). However in standard point transect sampling, for which an observer records from each point for a predetermined length of time (typically between 5 and 20 min), there is potential for substantial bias. In the Montrave case study, for which the observer recorded all male birds detected for a period of 5 min at each visit to a point under the standard method, this movement generated clear bias for one of the four species surveyed. The great tit (Fig. 10.1) tended to keep on the move, so that, if the observer stood at a point long enough, one would eventually move into detection range. Territory mapping indicated that there were 21 territories km^{-2} . By contrast, standard point transect sampling gave an estimate of 58, with 95 % confidence interval of (39, 94) territories. A snapshot method eliminated this bias, giving an estimate of 22 territories km^{-2} , with a 95 % confidence interval of (12, 39) territories (Buckland 2006).

In the snapshot method, after arriving at the point, the observer notes any birds detected, and as far as possible, keeps track of them. The snapshot moment occurs a pre-determined time after reaching the point, say 2 min. Observers estimate distances to any birds whose location is known at the snapshot moment. They may take as much time as they wish after the snapshot moment to confirm locations of birds, but should not record new birds which may have moved onto the plot after the snapshot moment. If a bird was located either before the snapshot moment or

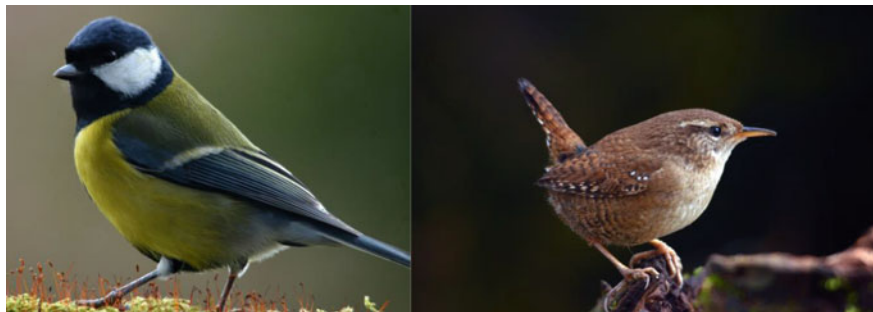


Fig. 10.1 Of the four species surveyed in the Montrave case study, the great tit (*left*) showed the greatest tendency to move large distances during the 5-min counts used in the standard point transect method. This generated clear bias in the abundance estimate, which was avoided by using the snapshot method. The winter wren (*right*) exhibited the strongest evidence of responsive movement, probably because it was typically at or near ground level, while other species tended to be in trees. Photos: Steve Buckland

after, or both, but its location was uncertain at the snapshot moment, then it is not recorded. This does not cause bias, unless locations cannot be confirmed for birds at or very near the point, which we assume are certain to be recorded (Sect. 10.1.3). After the snapshot moment, the observer is allowed to move away from the point, to check locations of birds, and to allow triangulation to singing birds, so that distances may be estimated more accurately.

More careful observer training tends to be needed if the snapshot method is adopted, as the rules about how the observer should operate in the field are less rigid and more open to interpretation than those typically adopted for standard point transect sampling based on timed counts.

If it is not feasible to implement the snapshot method, but movement independent of the observer is known to be a problem, then the cue-count method (Sect. 9.4.2) should be considered. Because the recording unit for that method is short songbursts, bird movement that is independent of the observer does not cause bias.

10.1.2.2 Responsive Movement

As for movement independent of the observer, responsive movement generates bias in both line transect and point transect sampling, and again, the potential for bias is greater for point transect sampling than line transect sampling. Some songbirds will be attracted to an observer; in such cases, if they are not detected prior to responding, upward bias in density estimates will occur. This bias is larger if the observer double-counts a bird that approaches the point, believing it to be two different birds. If birds move away from the observer before detection, under-estimation of density occurs, which might be substantial if many such birds move beyond detection range.

In the Montrave case study, we noted in Sect. 5.2.3.3 an indication that some European robins might have moved away from the point before detection. There was stronger evidence of avoidance behaviour for the winter wren (Fig. 10.1). In the right-hand plot of Fig. 10.2, we show a histogram of estimated distances of detected wrens from the point, together with the fitted probability density function corresponding to the hazard-rate model for the detection function. In the left-hand plot, we show the fitted hazard-rate detection function. There appears to be clear evidence of too few detections near the point, and too many at distances of around 40–50 m. The effect is exaggerated in the bottom plot, due to the rescaling of frequencies; the top plot is the better guide. In fact, goodness-of-fit tests even here do not give compelling evidence of avoidance — the χ^2 test is not quite significant at the 5 % level, and the Kolmogorov–Smirnov and Cramér–von Mises test statistics are not significant at the 10 % level. Nevertheless, all four survey methods (standard point transect, snapshot point transect, cue count and line transect) gave data showing similar evidence of avoidance; if we pool the evidence, it is clear that some avoidance occurred.

Provided any avoidance does not take birds beyond detection range, bias due to responsive movement away from the line or point tends to be fairly modest. This is because we constrain fitted detection functions to be non-increasing functions. As a consequence, the fitted model tends to average out the lack of detections near the line or point with the excess of detections at mid-distances. This effect is evident in Fig. 10.2.

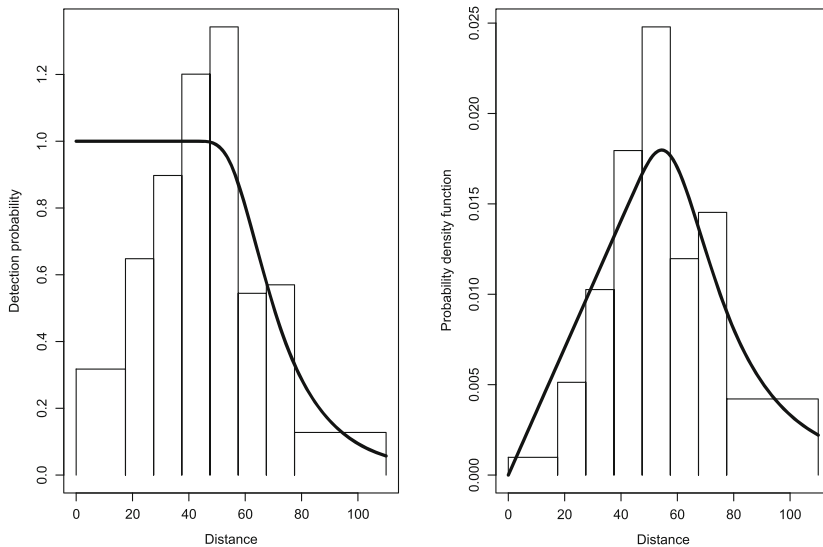


Fig. 10.2 *Left*: the fitted hazard-rate detection function, plotted with scaled distance frequencies. *Right*: histogram showing relative frequencies of estimated distances to detected wrens for the snapshot point transect method, together with the fitted probability density function corresponding to the hazard-rate model

10.1.3 Minimizing Bias from Failure to Detect All Birds on the Line or Point

For most species of songbird during the breeding season, the male sings frequently for at least part of the day, so that for surveys conducted at that time of the day (usually early morning), males near the line or point are readily detected. However, females may be silent, and may be hidden on nests. Thus we cannot expect to detect all females even if they are on the line or point. It is usually preferable therefore to survey just territory-holding males, for which the assumptions of distance sampling are likely to hold. We thus obtain an estimated density or abundance of territories; and for most species, doubling this number is likely to give a better estimate of adult abundance than would a survey in which all detected adults are recorded.

However, some species do not sing, or do not sing frequently enough to ensure that they will be detected through their song if they are at the point. In open habitats, this might not be a problem, if such birds are readily detectable visually. In forest habitats for example, birds may be high up in the canopy, and only detectable if they sing or call. Similarly, birds in dense bush may be undetectable visually. If individuals of such species are frequently silent, then line transect sampling is unlikely to give reliable abundance estimates. In the case of point transect sampling, the time spent at the point might be increased, so that a male is likely to sing at some point during a visit to a point. However, the longer the observer spends at the point, the greater the bias from movement of the bird. A bird moving around will tend to be detected when by chance it passes close to the observer, so that recorded distances tend to be too small. This generates upward bias in abundance estimates, which will cancel with downward bias arising from not detecting all birds at the line or point only if the choice of time spent at each point is fortuitously just right. Rather than rely on this, cue counting (Sect. 9.4.2) is likely to be a better option. In cue counting, there is no requirement for the observer to detect a bird at the line or point with certainty; instead, the song (or call) of such a bird should be detected with certainty. That is a much more plausible assumption for many cryptic species. A further advantage of cue counting is that the observer can remain at the point for as long as necessary; bird movement that is independent of the observer does not bias the method.

We anticipate that many songbird surveys in future will be conducted as acoustic surveys. A suitable acoustic array can be left at a point for a period of time. Distances of singing birds from the point can be estimated by triangulation, given the slightly different times of arrival of a songburst at the different elements of the array. The data may then be analysed using the cue-count methods of Sect. 9.4.2. Alternatively, acoustic data may be analysed using spatially-explicit capture–recapture methods (Sect. 10.3.3.1). For these approaches to be cost-effective, computer software that automatically extracts songburst data and identifies species would be very useful, and is technically feasible. If it is further possible to identify individuals by their song, then such methods may also be an important aid in estimating cue rates (mean number of songbursts per bird per unit of time during the survey period).

10.1.4 Minimizing Bias from Inaccurate Distance Measurement

Errors in estimating distances are more of an issue for surveys of breeding songbirds than for most distance sampling surveys. There are two reasons for this. First, typically, a high proportion of birds are detected by sound alone, and it is more problematic to estimate distance to a sound than to a visual detection. Second, for most taxa, line transect sampling tends to be preferred to point transect sampling, but because songbird surveys are often multi-species, with a risk of the observer being 'swamped' by the number of songbursts of various species in a short time, and because it can be difficult to navigate a random line on foot, point transect sampling is often preferred for breeding songbird surveys. Measurement error creates much greater bias for point transect estimates than for line transect estimates (Sect. 11.3).

For point transect sampling, it is common for observers to record distances from the point in distance intervals: the data are counts by distance interval. This eases the task of the observer if many birds might be detected at a single point. However, it is still necessary to record the correct distance interval for each detected bird, as far as possible.

Laser rangefinders are inexpensive and invaluable tools for estimating distances in point transect surveys. In our view, songbird distance sampling surveys should not be conducted without them. They allow distances to a visual cue to be measured to the nearest metre. While they are less useful for aural cues, they still remove one element of guesswork: to estimate the distance to an aural cue, the observer must first estimate where the bird is, and then estimate the distance to that estimated location. Rangefinders remove the estimation error from the second step. (If the location is not visible, the observer can measure the distance to a visible location that is thought to be around the same distance from the point.) Also, rangefinders are invaluable for checking observers' abilities to estimate distances in training, in distance estimation experiments, and as feedback during the survey.

Because of the difficulty in estimating distances to birds that can be heard but not seen, often point counts are carried out without recording distances. Nichols et al. (2000) developed a removal estimator to allow for imperfect detection on the plot in the absence of detection distances, in which a primary observer 'removes' birds, and a secondary observer records birds that he or she detects that were undetected by the primary observer. Farnsworth et al. (2002) developed a similar removal estimator approach, but divided the time at the point into periods, with new detections in later periods being recorded. The difficulty with this approach is that removal estimators can be badly biased in the presence of unmodelled heterogeneity (differences among birds in the probability of being detected). The main source of unmodelled heterogeneity in this case is due to distance of the bird from the point. Farnsworth et al. (2002) attempted to address heterogeneity by allowing two categories of bird: those with high detectability and those with low. However, heterogeneity arising from distance of the bird from the point will tend to be a

smooth function of distance, with detection probability decreasing with distance. For a more detailed discussion, see Buckland et al. (2004, pp. 352–354).

If detection distances are recorded, then the problem of heterogeneity may be overcome by combining removal estimators with distance sampling methods, as described by Buckland et al. (2004, pp. 354–356). However, bias also arises from bird movement. If birds move around during the count period, the size of the plot being surveyed is not well-defined because birds initially outside the surveyed circle may enter it. The effect of this for the method of Farnsworth et al. (2002) is that too many new detections are recorded in later time periods, leading to overestimation of bird density. For the method of Nichols et al. (2000), both observers tend to record too many detections, and overestimation again occurs.

10.2 Seabirds

For colonial seabirds, it is generally simpler to estimate the size of breeding colonies than to estimate numbers at sea. For smaller colonies, complete counts can be attempted; for larger colonies, some form of plot sampling is typically used. However, it is often necessary to estimate numbers at sea. This may be because a species does not nest in well-defined colonies that can easily be surveyed, or because the location of some colonies is unknown. In other cases, it may be because density of birds at sea rather than abundance of a biological population is of interest. For example, if an offshore windfarm is proposed and an impact assessment is required, we wish to quantify the density or abundance of birds within the footprint of the proposed windfarm to assess potential impact. Once such a windfarm is constructed, we would like to assess to what extent numbers have changed, relative to a control area. These issues are discussed in Sects. 3.3 and 7.4.3.

Seabirds at sea are surveyed by observers on board a ship or boat or in an aircraft. Increasingly, high-resolution video or photography taken from aircraft is used. The issues for each of these three options differ, and we treat them separately below.

10.2.1 *Shipboard Surveys*

For birds on the sea, it is generally straightforward to conduct a line transect survey. Diving species may be problematic; if so, observers should search using binoculars, so that birds on or near the line should be easily detectable on the surface between dives.

Distance estimation is problematic at sea. Laser rangefinders do not work for measuring to a point on the surface, unless it is possible to ‘hit’ the bird. Further, in high density areas, it is not practical to attempt to record distance to each detected bird. Typically therefore, a count of birds in each of several distance intervals is made. A judgement is still required to place each detected bird in one of the

intervals. Binoculars with reticles, ideally with reticle marks located to correspond to cutpoints between distance intervals, can be very useful. Failing this, a laser rangefinder can be used opportunistically to measure distances to floating objects, so that the observer can calibrate his or her estimates of distance against known distances.

If the target species can occur at very high densities, strip transect sampling (Sects. 1.4.1 and 6.2), with a relatively narrow strip to ensure that all birds within it are detected, might be preferred to line transect sampling.

Seabirds in flight typically have a mean speed that exceeds that of the observer, so that standard line transect methods would substantially over-estimate abundance. If such birds do not respond to the ship, a simple solution is to record birds in flight at their location when they pass abeam of the ship (Buckland et al. (2001, p. 202); see also Sect. 11.4.1). If a previously-detected bird is no longer visible at this moment, it is not recorded. In the case of strip transect sampling, the bird is only counted if it is in the strip when passing abeam.

Another solution to birds in flight is to survey only birds on the sea, then separately estimate the proportion of time spent flying, so that the abundance estimate can be adjusted for flying birds. Satellite tags, which provide data to determine whether a bird is flying, on the sea or on land, make this a more feasible option.

Spear et al. (1992) carried out strip transect sampling, and included detected birds in flight in the counts. They then adjusted their biased estimates, to take account of both speed and direction of travel for birds in flight (Spear and Ainley 1997a,b).

However, the most popular approach for dealing with birds in flight is plot sampling, using the method of Tasker et al. (1984), or a variant of this method. Instantaneous counts of birds in flight within a defined plot ahead of the ship are made at pre-determined points in time. Problems with the method are: the count in practice cannot be instantaneous; it is difficult to determine the boundaries of the plot, to decide which birds are in and which are out; and for smaller species, birds within the plot may pass undetected. Nevertheless, the method is likely to lead to substantially lower bias than would standard line or strip transect sampling in which all detected birds in flight are recorded.

General issues to consider with shipboard surveys are addressed by Buckland et al. (2001, pp. 288–291).

10.2.2 Aerial Surveys

10.2.2.1 Visual Surveys

Birds in flight do not need to be treated separately when carrying out aerial surveys, as the aircraft travels much faster than the mean speed of the birds. However, aerial observers can easily get ‘swamped’ in areas of high density. To reduce the likelihood

of this, distance intervals are defined, and counts of birds in each interval made. This task is made easier if markers are placed on wing struts (see Fig. 7.9 of Buckland et al. (2001, p. 261)), together with markers on the windows, and the observer simply positions him or herself so that the markers are aligned, and counts the number of birds passing between each set of markers.

Typically for aerial surveys, it is necessary to left-truncate the distance data, as birds directly below the aircraft may not be visible, or may be difficult to count. See Fig. 7.8 of Buckland et al. (2001, p. 260).

For aerial visual surveys, the aircraft must fly at low altitude, and this may result in birds taking flight as the aircraft approaches. In this circumstance, observers should ensure that they count birds in the distance interval that they were in before responding to the aircraft.

Relative to shipboard surveys, aerial surveys have the advantages that birds in flight do not generate significant bias to the standard method; birds that respond to the aircraft typically do not have time to move far before detection; survey costs tend to be lower (unless a small boat can be used rather than a ship). Disadvantages are that many areas are not accessible to land-based aircraft; identification of birds is more difficult; observers may be unable to make accurate counts in areas of high density; and for diving birds, some will be underwater and unavailable to be detected when the aircraft passes. In the case of availability, it may be necessary to estimate the proportion of time that birds are unavailable, to correct for birds that are underwater. In this case, it is important that availability is clearly defined, to ensure consistency between the aircraft observers and the study to determine proportion of time available. For example, the aerial observer may detect and count a bird ahead of the aircraft which then dives so that it is underwater at the time the aircraft passes abeam.

For a discussion of issues in conducting visual aerial surveys, see Buckland et al. (2001, pp. 280–287).

10.2.2.2 Surveys Using High-Resolution Imagery

Advances in technology and software mean that aerial surveys conducted using high-resolution video or photography can now out-perform visual surveys. The surveys are conducted at substantially higher altitude than visual surveys, which has two advantages: the aircraft does not disturb the birds, so no responsive movement occurs; and in studies to assess the impact of a windfarm on bird densities, the aircraft fly well above the turbines, allowing randomly positioned transects to be followed. For visual surveys, if aircraft are allowed onto the footprint of the windfarm at all, they must stay well away from turbines, compromising the random design.

A further advantage of digital surveys over visual surveys is that the birds counted are on record, and can be validated by different software or experts. As technology advances, resolution improves, allowing more reliable and verifiable species identification relative to visual surveys.

For diving species, availability is more tightly defined than for visual aerial surveys. For digital stills, the bird is available if it is at the surface when the still is taken, and not if it is underwater. For digital video, a line perpendicular to the transect may be superimposed on the video, and if a bird is at the surface when it intersects that line, it is available. In both cases, a simple estimate of proportion of time spent underwater allows the abundance estimate to be corrected for diving birds. This estimate might be obtained by observing a representative sample of birds, or by placing tags on them that provide data on whether or not they are underwater.

Given the altitude that the aircraft fly at, and the set-up of cameras, detectability does not fall off with distance from the trackline, and for birds at the surface, detectability within the surveyed strip can be assumed to be certain. Thus data are analysed using strip transect methods (Sects. 1.4.1 and 6.2). Computer software is used to detect birds on the images. Species identity might be done by the software, with validation by experts, or by experts checking the images of birds detected by the software.

Trials were conducted on the winter flock of common scoter (*Melanitta nigra*) in Carmarthen Bay in the UK, to compare two digital methods (one using high-resolution video and the other using high-resolution stills) with aerial visual surveys (Buckland et al. 2012). For each method, a systematic grid of transect lines was laid over the bay, and surveyed on four occasions. The digital surveys were analysed as standard strip transect data, while the visual survey was analysed using standard line transect methods. In addition to standard design-based methods, a density surface was fitted to the data for each method. We show abundance estimates from Buckland et al. (2012) in Table 10.1. The two digital methods agree very well, while the visual survey estimates are substantially lower. Common scoter are particularly challenging for visual observers because they occur in large numbers concentrated in a small area, and it is clear that the visual observers were unable to count numbers accurately.

Table 10.1 Design- and model-based abundance estimates for common scoter in Carmarthen Bay, March 2009, using visual aerial surveys, high-resolution video surveys, and high-resolution stills surveys

Survey type	Design-based \hat{N}	% cv	Model-based \hat{N}	% cv
Visual	6197	22	7976	38
Digital still	17,501	38	16,490	43
Digital video	18,034	32	15,489	23

10.3 Cetaceans

Cetacean surveys have several issues in common with seabird surveys. Relative to terrestrial environments, the marine environment is relatively homogeneous from the perspective of designing a survey, and it is generally straightforward to follow random transect lines. Further, the environment is not fixed, unlike terrestrial environments. Thus whether or not repeat surveys follow exactly the same lines is less of an issue; the correlation between counts taken on repeat visits to a transect on the surface of the sea is likely to be weak compared with that for a terrestrial transect.

Many seabirds and all cetaceans are not continuously available for detection at the surface. When dive times are typically short relative to the time that an animal's location is within detection distance of the observer, mark-recapture distance sampling methods (Sects. 5.4 and 6.4.4) can be used to account for this, especially if the two observers do not search the same area at exactly the same time. However, for aerial surveys, and for shipboard surveys of long-diving species, availability should be addressed (Sect. 11.2).

Cetacean surveys also raise some issues that are less relevant to seabirds. They typically occur at lower densities, so that small sample size is often an issue. They may also only be visible for an instant at each surfacing, which for species that occur mostly as single animals or very small groups, has implications for how observers search for them (*e.g.* search close to the line rather than further out; use naked eye search to give a wide field of view; use multiple observers to search simultaneously).

10.3.1 Shipboard Surveys

10.3.1.1 Reducing Bias from Responsive Movement

Cetaceans may respond to the ship. It is important to attempt to detect animals before they respond, which may favour binocular search over naked eye search. For species that give brief opportunities for detection (*e.g.* a whale blow, or a brief surfacing of a porpoise), probability of detection can be enhanced by having a team of observers, using low-power binoculars (Fig. 10.3) or naked eye search (or both) to give a wide field of view. For species that usually give a continuous cue, such as dolphins that typically occur in large groups, observers might use large tripod-mounted binoculars to ensure detection at larger distance (Fig. 10.4); this both improves sample size and helps ensure that groups are detected before they respond to the ship.



Fig. 10.3 Main observation deck with heated booths for marine mammal observers on the Japanese research vessel *Kaiko Maru* during the SOWER 2009/2010 expedition into Antarctic waters. Note the low-power binoculars with wide field of view, useful for searching for minke whales which tend to occur singly or in small groups Photo: Cornelia Oedekoven

10.3.1.2 Estimating Distances and Angles

Distance estimation at sea is problematic, and it is important both to train observers in distance estimation and to provide aids that allow more accurate estimation. Binoculars with reticles (Buckland et al. 2001, pp. 256–258), allowing measurement down from the horizon (Fig. 10.5), are useful if not essential. A laser rangefinder allows observers to assess opportunistically their ability to estimate distance (but does not work for estimating distance to a point on the sea surface, unless there is a floating object). It is seldom possible to estimate perpendicular distances of animals from the line reliably for shipboard surveys, unless the animals are roughly abeam. Thus observer-to-animal distances are estimated, along with sighting angles from the line, from which perpendicular distances from the line can be estimated (Fig. 1.1). Angle boards (Buckland et al. 2001, p. 263) or angle rings on tripods should be provided for angle estimation.

10.3.1.3 Dealing with Cetacean Groups

Several issues may arise when estimating cetacean group sizes. The entire group may not be at the surface at the same time, animals may move around quickly making them difficult to track, subgroups of the same group may be spread out



Fig. 10.4 Flying bridge setup on the NOAA ship David Starr Jordan during a survey off the California, Oregon and Washington coasts. There were three rotating marine mammal observers and one seabird observer. The inner tripod-mounted binoculars were used by the centre marine mammal observer during chases and the seabird observer for flock counts. Tripod-mounted binoculars with 25 \times or 30 \times magnification have a narrow field of view, and so are unsuited to surveys of cetaceans that occur singly or in small groups that do not give many sighting cues, but they are very effective for detecting schools of dolphins at great distances. Photo: courtesy SWFSC/NOAA

over large areas, and individuals may have different surfacing intervals. Group sizes may be very large so that they would be difficult to estimate even if the other issues did not exist. In addition, different types of cetacean behaviour may give rise to different estimated sizes of the same group by the same observer. For example ‘low swimming’ is an evasive type of behaviour observed in dolphins where surfacing is kept to a minimum to avoid detection. In contrast, ‘running’ is a different type of evasive behaviour where dolphins maximize airtime for maximum speed (Mesnick et al. 2002).

When cetacean groups are composed of several subgroups, it is important to define a clear protocol for field personnel whether to treat the group or the subgroup as the recording unit. For the latter case, distances and group sizes need to be recorded for each detected subgroup. During a survey in the Hawaiian Island EEZ, the target species was false killer whales (*Pseudorca crassidens*), a highly mobile species where subgroups may be spread over tens of kilometres (Bradford et al. 2014). Here, visual and acoustic methods were combined for obtaining distances from the trackline and tracking individual subgroups, while visual observers made



Fig. 10.5 View through large tripod-mounted binoculars. Lining up the 0 reticle reading with the horizon enables conversion of the reticle reading of the sighting to a distance from the observer (which requires knowing the height of the observer above sea level). Image constructed from two photos for illustrative purposes. Photo: Cornelia Oedekoven

group size estimates for each subgroup. Sperm whales also form groups consisting of multiple subgroups which may be spread over several kilometres. These whales are less mobile but dive asynchronously for periods of up to 90 min. Hence, Barlow and Taylor (2005) implemented 90-min counts of sperm whales so that each subgroup could be detected at least once during the count. During these 90-min counts, five or more observers tracked the location and composition of each subgroup. As the recording unit was group, total group size represented the sum of all subgroup sizes.

Dolphins may occur in mixed-species schools (Fig. 10.6), and may have highly variable school sizes possibly containing thousands of individuals. To deal with these issues, observers during eastern tropical Pacific surveys were calibrated by making independent estimates of the size and composition of calibration schools (Gerrodette et al. 2008). These calibration schools generally consisted of single- or mixed-species dolphin groups and were photographed from an aircraft using aerial photogrammetry methods; hence true school size and composition could be established later, to allow errors in estimates to be quantified.



Fig. 10.6 Mixed school of eastern spinner dolphins (*Stenella longirostris orientalis*) and north-eastern offshore spotted dolphins (*Stenella attenuata attenuata*) sighted during a cruise in the eastern tropical Pacific. Photo: courtesy SWFSC/NOAA

10.3.2 Aerial Surveys

10.3.2.1 Visual Surveys

Issues for visual aerial surveys of cetaceans are very similar to those for seabirds (Sect. 10.2.2.1). Distance estimation problems are again less if distance data are recorded by interval, with interval cutpoints delineated by aligned markers on the windows and wing struts. Left-truncation is likely to be necessary to allow for failure to detect animals under the aircraft, unless an aircraft with belly and/or bubble windows is used.

For aerial surveys, a diving response is problematic, as the animals may not be visible by the time the observer scans their location. In this circumstance, availability of animals for detection would be lower than would be the case without disturbance, so that any adjustment for availability, calculated from a separate study for example using tagged animals, may introduce bias.

10.3.2.2 Surveys Using High-Resolution Imagery

The methods described in Sect. 10.2.2.2 are also useful for cetaceans. The issue of correcting for diving animals is even more important than for seabirds. If dives are typically very short, then multiple images can allow correction for availability. For

example, two cameras might be mounted under the aircraft, one pointing forwards and the other backwards. If both also take overlapping images separated slightly in time, this can help. For longer diving species, it may be necessary to tag a representative sample of animals, to allow estimation of availability (Sect. 11.2).

10.3.3 Acoustic Surveys

10.3.3.1 Overview

A somewhat recent development is the use of automated recordings of sounds produced by the animals themselves to estimate their abundance or density. These are referred to as passive acoustic density estimation methods. While in principle these methods can be applied to any sound-producing taxa, they have been developed mostly based on cetacean applications. This is natural because many cetacean species are very hard to survey, spending large portions of their time submerged, yet produce loud and easily distinguishable sounds. For an extensive review of this field, including applications to other taxa, we refer the reader to Marques et al. (2013).

Methods for passive acoustic density estimation of cetaceans might include towed surveys or fixed sensors. The former is a form of line transect sampling while the latter corresponds to point transect sampling. The use of slow-moving devices such as drifting buoys or wave gliders (*e.g.* Klinck et al. 2012) is a current research topic and combines aspects of both.

Towed passive acoustic line transect surveys might be carried out in their own right (*e.g.* Lewis et al. 2007) or in combination with a more traditional visual survey (Barlow and Taylor 2005). As long as perpendicular distances of detected animals from the trackline can be obtained, the analysis methods used are the same as for visual surveys. A potential problem is when only a slant distance is obtained, yet a distance projected onto the sea surface is required (see Sect. 9.5). An example of when this can be safely ignored is a sperm whale survey. These are detectable at tens of kilometres, which means that, for most detections, the slant and projected distances are approximately the same. For a beaked whale detectable up to at most a few kilometres, and able to dive to deep waters, this might be a problem; the methods of Harris et al. (in prep.) should then be considered.

Fixed passive acoustics sensors can be considered as standard point transects, provided distances to vocalizing animals can be estimated based on the sound characteristics alone. This is generally not feasible, although Marques et al. (2011) present an exception with right whales in the Bering Sea. Given the sound propagation characteristics of the shallow continental shelf and the far-travelling right whale 'up' calls, distances to detected sounds could be estimated, allowing conventional distance sampling methods to be used. Within a distance sampling setting, if distances cannot be obtained, two options are still available. A theoretical detection function might be derived based on sound propagation models (Kusel et al. 2011), or a detection function might be obtained using auxiliary information, as in

Sect. 9.2. As with any model-based approach, the former method is only as good as the model used. To estimate the detection function using auxiliary information, data might come in a variety of forms from a range of sources (*e.g.* Kyhn et al. 2012). The key aspect to bear in mind is that data should be collected in circumstances representative of survey conditions, otherwise the estimated detection function might not apply, and consequently bias would arise.

Although beyond the scope of this book, acoustic fixed sensors might be used in a spatially-explicit capture–recapture (SECR) approach, where distances to sounds are not required, but identification of the same sound recorded at different sensors is (Marques et al. 2012; Martin et al. 2012). A non-technical overview to SECR is provided by Borchers (2012), while Borchers et al. (2015) provide a conceptual link between conventional capture–recapture methods and distance sampling, which can be viewed as two extremes within a common framework that includes no and full information on animal location, respectively.

10.3.3.2 An Example with Beaked Whales

The objective of this study was to estimate the density of Blainville’s beaked whales (*Mesoplodon densirostris*) at a US Navy range facility in the Bahamas (Fig. 10.7). The material in this section is based on Marques et al. (2009), where further details are available. Blainville’s beaked whales are deep-diving odontocetes, which are extremely hard to survey visually, but which produce easily detectable echolocation clicks for foraging. These clicks are produced only during the deep part of their dives. These animals tend to be almost metronomic divers, with deep foraging dives alternating with series of relatively shallow dives during which no clicks are produced (Johnson et al. 2004; Tyack et al. 2006).

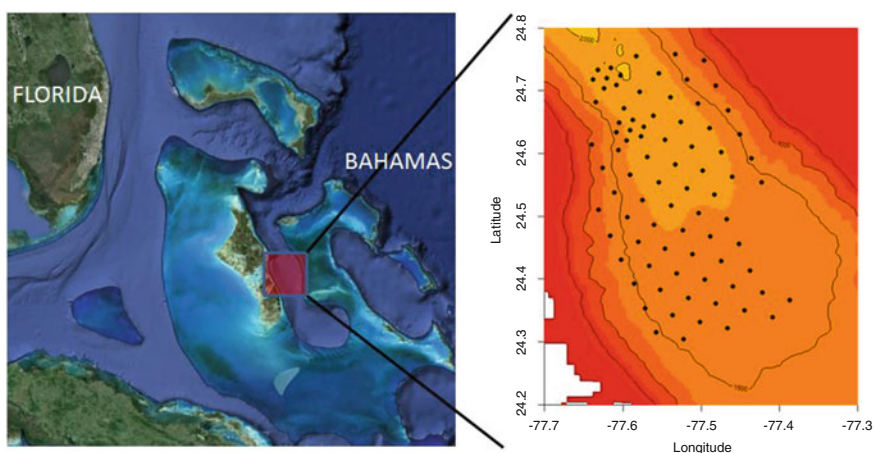


Fig. 10.7 The location of the Atlantic Undersea Test and Evaluation Center (AUTEC) range with an expanded view of the field of 82 hydrophones used for surveying Blainville’s beaked whales

For a period of 6 days, the number of sounds detected and classified as beaked whales in an array of $K = 82$ bottom-mounted hydrophones was counted. Once pooled across hydrophones, almost three million sounds believed to be beaked whale clicks were detected. Therefore, not surprisingly, the click counting process itself was automated. However, no such automated process is perfect. Hence, a required multiplier was the proportion of false positives (*i.e.* a sound assumed to be a beaked whale click, which in fact was not). Due to difficulties in evaluating some sounds, both optimistic and pessimistic scenarios were considered, *i.e.* assuming all or none of the uncertain sounds were beaked whale clicks respectively. Given an estimated density of clicks per unit time, animal density was obtained by dividing this estimate by an estimated average click production rate, obtained from a sample of animals fitted with acoustic tags (Johnson and Tyack 2003).

The density estimator used was given by

$$D = \frac{n_c(1 - \hat{c})}{K\pi w^2 \hat{P}T\hat{r}} \quad (10.1)$$

where n_c is the number of clicks detected, w is the distance from the hydrophones beyond which no cues are assumed to be detected, \hat{P} is the estimated average probability of detecting a cue made within a distance w of a hydrophone, \hat{r} is the estimated cue production rate, \hat{c} is the estimated proportion of false positive detections, K is the number of replicate sensors used, and T is the time that the hydrophones were operating. Note therefore that $n_c(1 - \hat{c})$ is the estimated true number of beaked whale clicks detected, $\hat{r}T$ is the mean number of clicks produced by an animal during the recording period, and $K\pi w^2 \hat{P}$ is the effective detection area. Additional details, including how the different multipliers were obtained, can be found in Marques et al. (2009). Here we concentrate on describing how detectability was estimated; that is, how the mean probability of detecting a click from a given hydrophone, given that the click was produced within a distance w of the hydrophone, was estimated.

Because distances could not be obtained for clicks detected on the hydrophones, additional information was required to estimate detectability. A small number of whales was fitted with a DTAG, an archival tag which records depth, 3D accelerometer, 3D magnetometer and acoustic data, at a very high sampling rate (Johnson and Tyack 2003). Using these data, we can both (1) know exactly when the tagged animal produced each one of its clicks and (2) by correlating this timing information with the sounds detected on the field of surrounding hydrophones, we can obtain the three-dimensional position of the animal in space, as well as its orientation. These now provide trials at known locations, allowing us to determine whether or not each click was detected by each of the surrounding hydrophones within a given radius. This radius was defined as the radius beyond which there was a negligible probability of detecting a click. Given the properties of the acoustical setting, this was set at 8 km. These 0/1 (corresponding to undetected/detected) data can be modelled using logistic regression, in which the probability of detecting a

click is modelled as a function of distance and relative horizontal and vertical angle with respect to the hydrophones. Note that unlike what is usual for conventional distance sampling, we modelled detectability as a function of slant 3D distance instead of a distance projected on the surface or on the bottom. The orientation of the whale with respect to the hydrophone was known to be important *a priori* because these clicks are highly directional in their sound intensity (e.g. Zimmer et al. 2005). The logistic regression model was fitted using the `mgcv` R package. The outcome of this modelling exercise was reassuring, returning the patterns expected. The probability of detection of a click decreases with distance to the hydrophone, and with the off-axis horizontal and vertical angle with respect to the hydrophone (Fig. 10.8).

Once this model was obtained, the remaining step was to estimate the mean probability of detection of a click. This involves averaging over the unknown

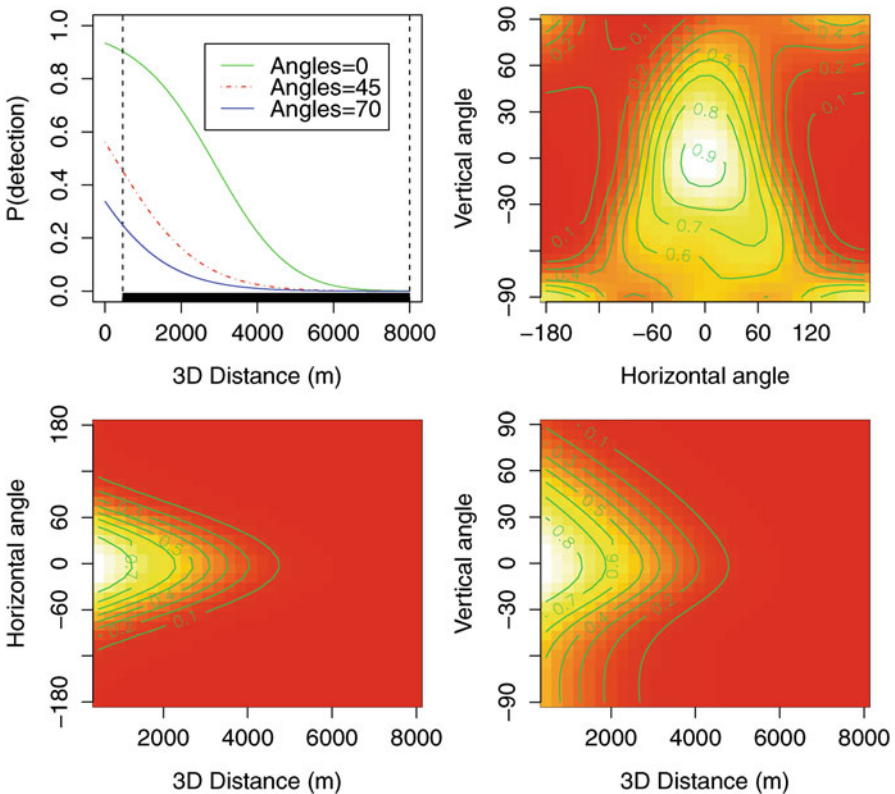


Fig. 10.8 Probability of detection as a function of the hydrophone-to-click three-dimensional distance and off-axis angles. Plots are conditional on a given value for the variable(s) not shown: distance and angle were considered to be 0 m and 0°, respectively (i.e. maximizing the probability of detection). Vertical dashed lines on the top left plot correspond to the maximum and minimum available distances

variables present in the model (distance, horizontal and vertical angle) which we do not observe for the n_c detected clicks. To do so, thousands of 3D tracks were simulated by re-running the existing deep-dive 3D trajectories around hydrophones, assuming a random start and orientation for each track. For each such simulated click, its probability of detection was evaluated given the estimated model and the observed covariates. The average probability of detection of a click was then estimated by the sample average of the individual click detection probabilities, and the corresponding variance. The variability in the estimated detectability model was propagated through to the final density estimate using the delta method.

The estimated density was 25.3 or 22.5 animals/1000 km², depending on assumptions about false positive detections, with corresponding 95 % confidence intervals of (17.3, 36.9) and (15.4, 32.9). These values are consistent with previous density estimates for this area (*e.g.* Moretti et al. 2006).

This example, chosen to showcase passive acoustic density estimation, has features in common with two earlier sections. Here, we do not have direct information about detectability in the survey data, and so we conduct trials with animals located at known distances. This approach is also used for trapping and lure point transects (Sect. 9.2). This case study is also an example of cue counting (Sect. 9.4), but one for which we are not able to record distances of cues from the point in the main survey.

10.4 Primates

Plumptre et al. (2013) review census methods for primates, including line transect sampling, indirect surveys of nest or dung (Sect. 9.3), and lure point transect surveys (Sect. 9.2). Primate line transect surveys often suffer through lack of replication (a very small number of lines, sometimes just one, each of which is walked on multiple occasions) and randomization (lines placed along convenient routes or close to base), so that there is little or no basis for drawing inference on animal densities in a wider area. Good design practice for primate surveys is addressed by Buckland et al. (2010a).

To implement a randomized survey design, it may be necessary to cut transect lines. In this case, the lines should be cut at least a week before the line is surveyed, to avoid bias from disturbance, and a minimal cut should be used, so as not to create an obvious path for either hunters or animals (Buckland et al. 2010a). Each line may be surveyed more than once (Sect. 6.3.3.1), but the number of replicates for variance estimation is the number of lines in the design; repeat surveys of the same line are not independent, and so cannot be considered replicates.

Estimation of distance of detected animals from the line is often problematic for primate surveys. Primates often occur in groups, which may be spread over some distance. It has been common practice in primate surveys to record the distance of the first animal detected from the line, and then to adopt an analysis of clusters, taking the location of the first animal detected to be the location of the group.

However, the first animal detected tends to be closer on average to the line than other animals in the group, so that distances are biased towards zero (often with many being recorded as exactly zero). This leads to overestimation of density, and some primate surveys have thus been conducted using animal-to-observer distances, together with analysis methods that have no mathematical basis, because it is claimed that such methods have lower bias. Buckland et al. (2010b) review this approach, and clarify the problems with it. A far more satisfactory solution is to obtain better estimates of distances from the line.

10.4.1 Dealing with Primate Groups

The following is summarized from Buckland et al. (2010a).

There should be a clear protocol so that fieldworkers can determine what constitutes a group for the purposes of the survey. For example, if animals are separated by at least 20 m from the originally detected group, the protocol might state that these should be treated as a second group. This might result in a single large social unit being recorded as many groups. Any of those groups that is detected and whose centre is located within the survey strip of half-width w should be recorded, and their distance from the line measured or estimated.

Distances of group centres from the line should be measured as accurately as possible. This requires that the position of the line is well defined, so that distances from the line are well defined. Unless distances are sufficiently small to be measured with a tape without undue disturbance or delay, a laser rangefinder should always be used for primate surveys. Vegetation may prevent a direct measurement, but the rangefinder is still an invaluable aid for improving distance estimates (see Sect. 10.1.4).

Primates are often in large, dispersed groups, so that it is difficult to estimate distance except for the individuals first detected. The problem is made worse if the subjects flee from the observer. Where it is impossible to determine location of group centres with sufficient accuracy, but feasible to estimate distances to each detected individual, then you can ignore the existence of groups. Each individual that is detected is recorded, along with its distance from the line (Buckland et al. 2001, pp. 75–76). Standard methods assume that all animals on or very close to the line are recorded, but it does not matter if animals further from the line but in a detected group go undetected.

If it is not feasible to record all detected individuals, together with their distances from the line, then it is important to estimate the size and location of detected groups as accurately as possible. In fact, bias in estimates of the size or location of groups well away from the line need not be problematic, but for groups on or close to the line, bias should be as small as possible. A field protocol should be developed with these issues in mind. For example, if animals do not respond to observers, observing the group for a period of time from different locations on and off the line may allow an accurate assessment of size. If animals do respond, a quick count may be needed,

and multiple observers with slightly different vantage points, and a well-rehearsed protocol for coordinating their count, may be effective.

If neither of these strategies is achievable, it may be necessary to estimate mean group size and spread in a separate study from the line transect survey. In this case, the study should be conducted synchronously with the line transect survey, to ensure that the observed mean size and spread is representative of groups in the survey region at the time of the survey. The criterion for defining what constitutes a group must also be consistent between the line transect survey and the study to estimate group size. Problems with this approach include:

- It may be difficult to achieve an adequate sample size — at least 10, and preferably nearer 20 — especially if group size is very variable.
- If only habituated groups can be monitored in this way, they may not be representative of all groups.
- It is still necessary to estimate the location relative to the line of groups detected during the line transect survey.

To address the last point, it may be necessary to record the distance to the closest individual, whether it is closest to the line or to the observer, and correct either the recorded distances or the effective strip half-width to allow for group spread (Whitesides et al. 1988; Buckland et al. 2010a).

10.4.2 Other Approaches

Buckland et al. (2010a) suggested other distance sampling strategies for when standard methods fail or are impractical. We summarize those suggestions here.

If it is feasible to record each individual that is detected, together with its distance from the line, but it is thought that some individuals on the line are missed, it may be possible to conduct trials by locating individuals, perhaps using radio collars, and then sending observers who are ignorant of animals' positions past those at a known closest distance of approach. These trials generate binary data which may be modelled using logistic regression, with distance from the line and possibly other variables as covariates, from which the probability of detection on the line may be estimated. This estimate and its standard error may then be included as a multiplier in *Distance*, when analysing the line transect survey data. Similarly, if groups instead of individuals are recorded, but some groups on the line may be missed, trials might be set up involving the group instead of an individual.

The cue counting approach for birds (Sect. 9.4.2) may also work for primates that call. Cue rate (number of calls per animal per unit time) should be estimated in a synchronous survey, to allow conversion from number of calls per unit area per unit time to estimated animal density. The disadvantages of this approach are that it can be difficult to estimate distances to calling animals, and it is difficult to ensure that a representative sample of animals is monitored to estimate the cue rate.

If animals can be lured in by playing a call, then lure strip transects may be possible, as implemented in a study of cotton-top tamarins (*Saguinus oedipus*) (Savage et al. 2010). Observers simultaneously travel along two parallel transects, luring animals from within the strip between the transects. If the lure causes animals to respond by calling, but does not attract them in, a line transect version of this approach might be workable, with just one transect at each location. If several observers are positioned along the line, distances of responding groups from the line may be estimated by triangulation. Another possibility is lure point transects (Sect. 9.2), in which trials are conducted on subjects with known location, from which a detection function is estimated, and assumed to hold for the main survey, where a lure is played at each of a number of points systematically spaced through the survey region.

10.5 Ungulates

In this section, we are primarily concerned with surveys of deer and antelope, although the methods are also relevant to many other large terrestrial herbivores. In remote areas, the methods might be used to estimate livestock abundance, or the abundance of introduced populations of for example camels (*Camelus dromedarius* and *Camelus bactrianus*) and donkeys (*Equus asinus*) in Australia.

Deer and antelope are typically vigilant and can travel at high speeds. Consequently investigators employ a variety of strategies to cope with detecting such wary animals. Many ungulates can occur at low densities ($<1 \text{ km}^{-2}$) and range over large areas such that classical sampling techniques may be inadequate for producing defensible density estimates.

Challenges with ungulate studies include the following.

- Non-random transect placement.
- Responsive movement before detection.
- Cluster size estimation.

We consider these challenges in the context of ground-based and aerial surveys below.

10.5.1 Ground-Based Surveys

Ground-based surveys of ungulates often make use of vehicles on roads or tracks. This leads to violation of the assumption of random placement of transects (Sect. 11.1). Ungulates may have atypical densities along roads, for example due to disturbance, hunting pressure, or availability of a different habitat (the track, or perhaps forest edge). Further, roads are unlikely to be placed independently of topography (they avoid rugged terrain) or habitat (they favour dry ground), which

may create larger-scale differences in density between areas near roads and areas far from roads. A recent study by McShea et al. (2011) suggests bias in estimates of deer density can arise from non-random transect placement. Track-based surveys may need supplementary data, e.g. from additional transects perpendicular to tracks (Marques et al. 2013), to allow unbiased estimation of density.

If ungulates are too wary to be detected from the ground during the day, nocturnal surveys with spotlights or thermal imagers are one way of circumventing detection of the observer by the animals prior to detection of the animals by the observer (Gill et al. 1997; Marini et al. 2009; Focardi et al. 2013). Indirect methods (Sect. 9.3) can also be employed to circumvent issues of responsive movement (Sect. 11.4.2): pellet groups are surveyed to give pellet density estimates, which may be converted to estimates of animal density provided we have estimates of deposition and decay rate (see for example Marques et al. (2001); Acevedo et al. (2010); Alves et al. (2013)). The challenge with indirect methods is the need to estimate the deposition and decay rates that apply within the survey region in the lead up to the pellet survey (Laing et al. 2003).

Some species occur in large herds. In this circumstance, it is necessary to develop a field protocol for determining a 'cluster'. This might be taken to be a sub-group of animals within the larger herd for which a distance from the line can be estimated, along with the sub-group size. There is then no need to detect all sub-groups in the larger herd, although all those on or very close to the line (before responsive movement occurs) should be recorded.

In a study of tiger (*Panthera tigris*) prey in Nepal, Wegge and Storaas (2009) found that, when surveys for ungulates were carried out on foot, unseen and unidentified animals were heard fleeing, violating a key assumption. Surveys carried out by vehicle were biased because of the need to use tracks. By contrast, they found that surveys conducted from the backs of elephants (*Elephas maximus*) worked well.

10.5.2 Aerial Surveys

If it is impossible to follow random transects using ground-based surveys, then aerial surveys should be considered. Aerial surveys can also be effective for dealing with the problem of responsive movement, especially if observers have a good forward view, so that they can record the position of detected animals before responsive movement occurs. Either fixed-wing aircraft (Johnson et al. 1991; Guenzel 1997; Whittaker et al. 2003) or helicopters (White et al. 1989; Trenkel et al. 1997) might be used.

Detectability of animals near the flight line may be compromised by the configuration of the aircraft, although bubble or belly windows (Sect. 4.1.2.2) can be employed. Often in aerial surveys, observers search just one side of the aircraft, and the line is offset perhaps 30 m or so, to avoid the difficulty of detecting animals under the aircraft. In this case, the observers should be carefully trained, so that animals detected close to the line are recorded on the correct side. Given some uncertainty over animal location (especially if it is moving), and given observers' inclination to

err on the side of recording animals over which there is any doubt, failure to define a clear protocol, and to ensure that it is correctly implemented, can generate a spike in the histogram at around zero distance for one-sided or offset transects, which leads to overestimates of density.

Double-observer methods (Sect. 5.4) can prove useful for aerial ungulate surveys, allowing estimation of $g(0)$ (Fewster and Pople 2008).

Aerial surveys may allow more reliable estimation of cluster sizes than is possible from ground-based surveys, especially if high-resolution imagery is used. However if views of the ground are obstructed by foliage, then methods of estimating the proportion of animals available to be seen (Sect. 11.2) might be needed.

10.6 Butterflies

The UK Butterfly Monitoring Scheme (Pollard and Yates 1993) comprises a set of sites that are surveyed by walking a transect, and counting numbers of butterflies by species that are detected within an imaginary box ahead of the observer, making the method a form of strip transect sampling. It is often called a 'Pollard walk'. Originally, all sites were selected subjectively, at least in part because they contained good butterfly habitat, and the transect was selected to pass through the best habitat in the site. More recently, random sites have been added to the scheme, through the Wider Countryside Butterfly Survey. The field methods of the scheme have been adopted by a number of other butterfly surveys. However, there are potential problems with the method.

The first problem is that sites are chosen because they are good butterfly habitat, and then transects are placed through the best butterfly habitat within those sites. This has the potential for bias either way: butterfly trends in the wider countryside may be less favourable than in the best habitats; and, once defined, the transect route at a site stays constant, while the distribution of butterfly habitat in the site may change over time — as the transect becomes less effective at sampling the best habitat, downward trends in butterfly numbers might be observed. The wider countryside sites are an attempt to address both issues: they are selected according to a stratified random sampling scheme, and the ideal transects are two parallel straight lines, each 1 km in length. However, neither advantage is fully realised. The wider countryside sites were actually selected for surveying breeding birds, and the volunteers who cover the sites have the option of recording butterflies too. Some self-selection can be expected, with volunteers more likely to do the butterfly survey work in better sites for butterflies. Also, it is usually not possible to access all of the ideal transect routes, so a route approximating it is selected. In arable habitats especially, the actual route mostly follows field edges, where butterflies are more abundant. This is less of a problem in the case of breeding birds, because a wide strip, extending well into other habitats, is surveyed.

In many environments, this problem is essentially insoluble. If it simply is not possible to follow random transects, and the observers are forced to follow edge

habitat, then there is little basis for estimating absolute abundance. The hope then is that counts in edge habitat reflect trends in relative abundance over time. Further, even if it were possible to walk through arable crops for example, few butterflies would be recorded, which in turn would reduce the number of volunteers willing to conduct the surveys, and too few data might be gathered to assess trend.

The second problem is that in better conditions, more butterflies will be flying. This both makes them more detectable and generates upward bias in counts for the same reason that fast animal movement causes bias in line transect sampling. For any given butterfly at a site, the faster it moves, the more likely it is to enter the imaginary box ahead of the observer. Provided there is no trend in weather conditions, this will add noise to the counts and introduce bias to estimates of absolute abundance, but will not bias estimated trends in relative abundance. The methods attempt to reduce noise and bias from this source by restricting surveys to when the weather is favourable for butterflies.

In closed habitats, such as rain forest, distance sampling methods are likely to be of limited use, at best giving estimates of relative abundance, and at worst simply failing to detect sufficient butterflies to allow analysis.

If a survey is not dependent on volunteers, if random transects can be walked, and if the habitat is open, it should be possible to carry out more rigorous surveys, allowing absolute abundance to be estimated. Often, brood size is of greater interest than abundance at any one point in time; in this case, some modelling of the abundance estimates is required to convert them to estimated brood size.

Brown and Boyce (1998) developed a line transect protocol for the endangered Karner blue butterfly (*Lycaeides melissa samuelis*). In a report on how to conduct surveys of the Karner blue, Grundel (2008) gave an excellent summary of the pros and cons of line transect sampling for assessing butterfly numbers. Isaac et al. (2011) compared the Pollard walk with standard line transect sampling, implemented at 13 sites in England and Wales. They found that on average, one third of butterflies in the box were not recorded using Pollard walk methods. This proportion varied by species; for the most detectable species, no individuals were missed, while for the least detectable, around three quarters were missed. The authors concluded nevertheless that the Pollard Walk was the more practical method for large-scale volunteer surveys. They did not address bias arising from butterfly movement, or from the non-random placement of transects.

10.7 Plants

Most distance sampling surveys of plants are straightforward. Plants do not change their location while the survey is taking place, and it is simple to measure distances from the line or point. Schorr (2013) for example used standard line transect sampling to estimate abundance of Weber's saw-wort (*Saussurea weberi*) at a location in the Rockies. He also discussed the merits of line transect sampling relative to plot sampling.

However, surveys for plants are not always as straightforward. First, it may not be possible to identify individual plants if they spread and intermingle. This may render distance sampling irrelevant, and plot sampling based on percent cover of each species may be more useful. However, for some species, it may be possible and useful to estimate density of flower stalks.

A second issue is that, for many species, it may not be possible to ensure that all plants on the line or point are detected, as some individuals may be small and/or hidden by other vegetation. This problem is made worse because the probability that a plant on (or off) the line or point is detected can be very variable, depending on the size of the individual plant, whether it is flowering, how much other vegetation is present, and so on. A further issue is that surveys are sometimes conducted at small sites holding a colony of plants of interest. In the case of line transect sampling, some sites may be too small to allow an adequate number of non-overlapping transects. Especially because the distribution of many plant species is highly clustered, this can mean that there are too few lines to ensure that plants are uniformly distributed with respect to distance from the line, even when a systematic grid of lines is randomly superimposed on the site.

Buckland et al. (2007) proposed a design and analysis to address all but the first of these issues. Two systematic grids of transects were placed at random over the study site, one at right-angles to the other (Fig. 10.9). The key to estimation is then

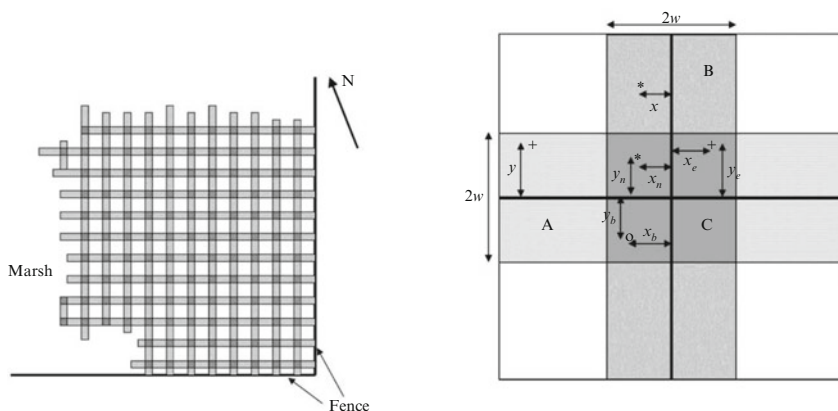


Fig. 10.9 *Left*: design of the Fleecefaulds survey, which comprises two systematic grids of strips, one with strips running approximately N/S, and the other with strips running E/W. The strips are of width $2w$, and the transect lines run down the middle of each strip. Each grid is randomly superimposed on the survey region. *Right*: schematic enlargement of a crossover of perpendicular transects. Detections made from the E/W line only are indicated by +, from the N/S line only by *, and from both lines by o. Within region A, distances y from the E/W line are recorded. Within region B, distances x from the N/S line are recorded. Within the intersection square C, both y and x are recorded; the subscript e indicates detected from the E/W line only, n indicates detected from the N/S line only, and b detected from both lines. The range for each of x and y is $[0, w]$; that is, we record absolute distance from each line, and we truncate observations at distance w . Reproduced from Buckland et al. (2007). ©The International Biometric Society

the squares formed by the intersection of a line from one grid with a line from the other grid. Plants that are detected during the survey from the first grid of lines are marked so that they are uniquely identifiable, but so that the mark does not affect their detectability (*e.g.* a mark on the underside of a leaf), and then plants detected from the second grid of lines are checked for marks. Thus mark-recapture data are available, so that MRDS methods can be used to analyse the data from intersection squares. Further, for any plant detected within an intersection square, whether recorded from both lines or just one, we can calculate its distance from each line (Fig. 10.9). If plants are uniformly distributed with respect to distance from the line, then plants detected from one line should be uniformly distributed with respect to their distances from the line perpendicular to it. If they differ significantly from a uniform distribution, then we can model the actual distribution, and thus eliminate the assumption of uniformity. Further, any covariates recorded on the plant, such as number of flowerheads, plant size, vegetation height, *etc.*, can be used to model heterogeneity in the probability of detection of individual plants.

This approach was used to estimate the density of cowslips (*Primula veris*) at Fleecefaulds Meadow in Fife, Scotland. Full modelling details are given by Buckland *et al.* (2007). Figure 10.10 shows that there is some evidence of non-uniformity of cowslips with respect to distance from the transect, with more plants on average towards the outer edge of the surveyed strips. A summary of density estimates is given in Table 10.2. We see that, if we assume that all plants on the line are detected, it makes very little difference whether or not we include covariates; this

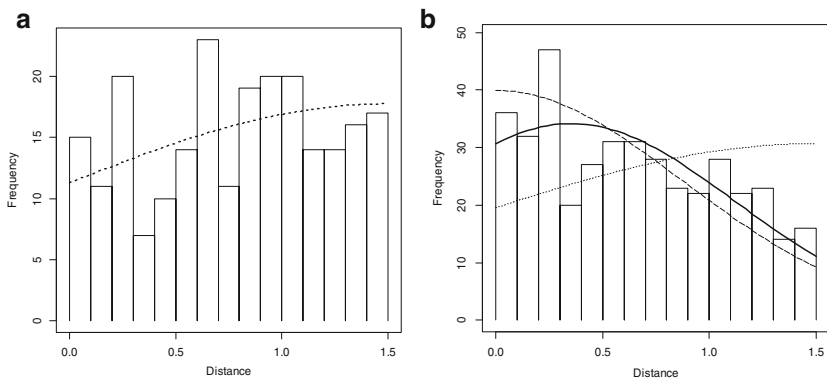


Fig. 10.10 (a) Histogram of the combined distances from the N/S lines of plants detected from the E/W lines, and distances from the E/W lines of plants detected from the N/S lines for cowslip data. The *dotted line* is the estimate of $\pi(\cdot)$ (the availability of plants, as a function of distance from a line), scaled to have the same area as the histogram. (b) Plot of scaled estimated detection probability (*dashed line*), estimated $\pi(\cdot)$ (*dotted line*), and estimated probability density function for observed distances (*solid line*) for cowslip data. The histogram shows number of detections by distance from the line from which they were detected. All functions have been scaled to have the same area as the histogram. Reproduced from Buckland *et al.* (2007). ©The International Biometric Society

Table 10.2 Estimates of density (plants ha⁻¹) of cowslips at Fleecefaulds Meadow (95 % confidence intervals in parentheses)

	Estimator	Density estimate	95 % CI
Conventional	\hat{N}_1	1048	(689, 1593)
distance sampling	\hat{N}_2	873	(568, 1342)
Conventional distance	\hat{N}_1	1052	(696, 1591)
sampling with covariates	\hat{N}_2	877	(574, 1341)
Allowing for	\hat{N}_1	1507	(1105, 2256)
$g(0) < 1$	\hat{N}_2	1258	(929, 1920)
Allowing for $g(0) < 1$ and	\hat{N}_1	1758	(1219, 2710)
non-uniform availability	\hat{N}_2	1467	(1032, 2278)

Estimator \hat{N}_1 assumes that density on the strips is representative of the entire study area, whereas estimator \hat{N}_2 uses data from the cross-strips to correct for any difference in density on and off surveyed strips. See Buckland et al. (2007) for details

is due to the pooling robustness property of conventional distance sampling methods (Sect. 5.1.1). When we allow for missing plants on the line, density estimates increase by around 44 %. When uneven density within the surveyed strips is allowed for, there is a further increase in abundance estimates by around 17 %. However, if we correct for estimated differences in density on surveyed strips and off strips (estimated from the cross-transect data, estimator \hat{N}_2), density estimates decrease by around 17 %.