# Review of potential line-transect methodologies for estimating abundance of dolphin stocks in the eastern tropical Pacific

Cleridy E. Lennert-Cody[1], Stephen T. Buckland[2], Tim Gerrodette[3], Andrew Webb[4], Jay Barlow[3], Peter T. Fretwell[5], Mark N. Maunder[1], Toshihide Kitakado[6], Jeffrey E. Moore[3], Michael D. Scott[1] and Hans J. Skaug[7]

*Contact email: clennert@iattc.org*

ABSTRACT

A twelve-year hiatus in fishery-independent marine mammal surveys in the eastern tropical Pacific Ocean (ETP), combined with a mandate to monitor dolphin stock status under international agreements and the need for reliable stock status information to set dolphin bycatch limits in the tuna purse-seine fishery, has renewed debate about how best to assess and monitor ETP dolphin stock status. The high cost of replicating previous ship-based surveys has intensified this debate. In this review, transect methods for estimating animal abundance from dedicated research surveys are considered, with a focus on both contemporary and potential methods suitable for surveying large areas for dolphin species that can form large, multi-species aggregations. Covered in this review are potential improvements to the previous ship-based survey methodology, other ship-based methods, alternative approaches based on high-resolution imagery and passive acoustics, and combinations of ship-based and alternative approaches. It is concluded that for immediate management needs, ship-based surveys, with some suggested modifications to improve precision, are the only reliable option despite their high cost. However, it is recommended that a top research priority should be development of composite methods. Pilot studies on the use of high-resolution imagery and passive acoustics for development of indices of relative abundance to be used in composite methods should be part of any future ship-based survey efforts.

KEYWORDS: ABUNDANCE ESTIMATE; INDEX OF ABUNDANCE; TRENDS; SURVEY-VESSEL; SURVEY-AERIAL; SURVEY-ACOUSTIC; SURVEY-COMBINED

## INTRODUCTION

For almost 50 years, the tuna-dolphin issue in the eastern tropical Pacific Ocean (ETP) has been studied and debated. Purse-seine vessels fishing for tuna in the ETP have exploited the co-occurrence of yellowfin tuna (*Thunnus albacares*) and conspicuous dolphin species to locate the fish since at least the 1940s (Silva, 1941; NRC, 1992). Purse-seine vessels began encircling dolphins in the late 1950s to catch the tunas (McNeely, 1961; NRC, 1992) and this fishing method resulted in substantial bycatch of dolphins (Perrin, 1968; Lo and Smith, 1986; NRC, 1992; Wade, 1995). Through fishermen's ingenuity and implementation of national and international management measures mortality has been reduced to a very small fraction of previous levels (NRC, 1992; Joseph, 1994; Hall, 1998; IATTC, 2016). Population dynamics modeling of dolphins has been the preferred approach used for evaluating stock status (Hoyle and Maunder, 2004; Gerrodette and Forcada, 2005; Reilly *et al.*, 2005; IATTC, 2006; Wade *et al.*, 2007, Gerrodette *et al.*, 2008) with respect to historical bycatch levels, and those models have relied on estimates of abundance from fishery-independent cetacean and ecosystem assessment surveys conducted by the National Marine Fisheries Service (NMFS) periodically between 1979 and 2006.

As a result of a hiatus in the NMFS surveys since 2006, there are currently no reliable indicators with which to monitor the abundance of the ETP dolphin populations. In addition to the fishery-independent surveys previously conducted by the NMFS, indices of relative abundance from purse-seine observer data have been proposed (Hammond and Laake, 1983; Buckland and Anganuzzi, 1988; Anganuzzi and Buckland, 1989) because of the large amount of observer data that are available, especially relative to data from fishery-independent surveys. At the time these methods were proposed, the primary method of dolphin school detection was by the vessel crew using high-powered binoculars (Buckland and Anganuzzi, 1988; Lennert-Cody *et al.*, 2001). However, since that time searching for dolphins associated with tunas has evolved and sightings associated with helicopter or radar constitute the majority of sightings. There are serious challenges to developing a reliable index from fisheries observer data, including potential differences in availability of sighting information by search method, changes in the use of different search methods depending on the vessel's perception of the local abundance of dolphins with tunas, and non-random distribution of tuna vessel search effort (Lennert-Cody *et al.*, 2001; 2016).

This lack of information on current dolphin stock status is problematic because, despite the current low levels of reported mortality (IATTC, 2016), high levels of historical mortality (Wade, 1995) and low estimated population rates of increase (Gerrodette *et al.*, 2008) have meant that population modelling results are sensitive to assumptions (Gerrodette and Forcada, 2005; Gerrodette *et al.*, 2008;

[1] *Inter-American Tropical Tuna Commission, 8901 La Jolla Shores Drive, La Jolla, CA 92037, USA.*
[2] *Centre for Research into Ecological and Environmental Modelling, University of St Andrews, St Andrews, Fife, KY16 9LZ, Scotland, United Kingdom.*
[3] *Southwest Fisheries Science Center, 8901 La Jolla Shores Drive, La Jolla, CA, USA 92037.*
[4] *HiDef Aerial Surveying Limited, Phoenix Court, Earl Street, Cleator Moor, Cumbria, CA25 5AU, United Kingdom.*
[5] *British Antarctic Survey, High Cross, Madingley Road, Cambridge, CB3 0ET, United Kingdom.*
[6] *Tokyo University of Marine Science and Technology, 5-7 Konan 4, Minato, Tokyo, 108-8477, Japan.*
[7] *University of Bergen, Postboks 7803, Bergen, Norway.*

IATTC, 2015a). The lack of information poses obvious problems for management. For example, the Antigua Convention[8] of the Inter-American Tropical Tuna Commission (IATTC) requires that the status of all species potentially impacted by the tuna fisheries in the eastern Pacific Ocean be monitored. In addition, abundance estimates are needed to ensure that dolphin mortalities are both sustainable and insignificant because the stock mortality limits for the purse-seine fishery are based on estimates of abundance (IATTC, 2006; NMFS, 2016). Recent efforts to obtain MSC certification by a segment of the purse-seine fleet operating in the ETP[9] also require determination of dolphin stock status. These needs provide impetus for updating the assessment of these stocks and resuming regular monitoring. However, fishery-independent surveys are expensive; the latest estimate of a one-year survey, were it to be conducted by the NMFS, is approximately US$10M (in 2017 dollars)[10]. Therefore, development of cost-effective means for monitoring dolphin stock status is desirable. To address this problem, a review of current abundance estimation methods and possible alternatives was undertaken.

In this review, line-transect methods for estimating animal abundance are considered, with a focus on both contemporary and potential methods suitable for surveys of dolphin species that have been involved in the ETP tuna purse-seine fishery. Of particular interest are methods for use on stocks of the offshore pantropical spotted dolphin (*Stenella attenuata*) and the spinner dolphin (*S. longirostris*), stocks that typically occur in large schools over extensive areas of ocean (Dizon *et al*., 1994; Scott and Cattanach, 1998, Scott and Chivers, 2009). This paper focuses on methods for dedicated research surveys; use of fishery-dependent data has been reviewed most recently by Lennert-Cody *et al*. (2016).

[8] *https://www.iattc.org/iattcdocumentationeng.htm.*
[9] *https://fisheries.msc.org/en/fisheries/northeastern-tropical-pacific-purse-seine-yellowfin-and-skipjack-tuna-fishery/.*
[10] Lisa Ballance, NMFS, pers. comm., 15 July 2016.

## SHIPBOARD SURVEYS

### Current methods

Line transect surveys conducted by the NMFS in the ETP began in 1974 using a combination of aircraft and ships (Smith, 1981; Holt and Powers, 1982). Shipboard procedures were refined each year and, by 1979, were close to current procedures. Since 1986, the surveys have used a stratified random design. In general, about three times more effort per unit area has been allocated in the central core area than in the outer or peripheral area (Fig. 1). The core area includes the main dolphin stocks of interest, namely northeastern offshore spotted and eastern spinner dolphins, and is the main area where purse-seiners set on tunas associated with dolphins (IATTC, 1999; 2015b). Because the ETP area is large and the research vessels have a limited range of 20–30 days, it is not possible to lay transect segments strictly at random. Instead, prior to departure, waypoints are chosen to achieve the desired allocation of effort among strata, approximately even spatial coverage within each stratum, and a length of trackline that returns the ship to port at the end of each leg. Since 1986, each survey has utilised 2 ships (3 in 1998) for 120 sea days each, with 4–5 legs per ship from late July to early December. During the survey, the ships proceed from waypoint to waypoint at 10 knots. Waypoints are typically hundreds of miles apart. Search effort takes place when there is sufficient light for effective detection of animals (normally about 30 minutes after sunrise to about 30 minutes before sunset). Search effort is suspended if it is too windy (normally Beaufort sea state > 5), if visibility is severely limited by rain or fog, or if the horizon is not visible due to haze. At night and during such periods of suspended effort, the ships continue along the planned tracklines to stay on schedule.

The surveys have used teams of three observers. Early experiments with helicopters established that dolphin schools ahead of the vessel were seen by observers on the vessel (i.e. that availability bias was low), and that most dolphin schools

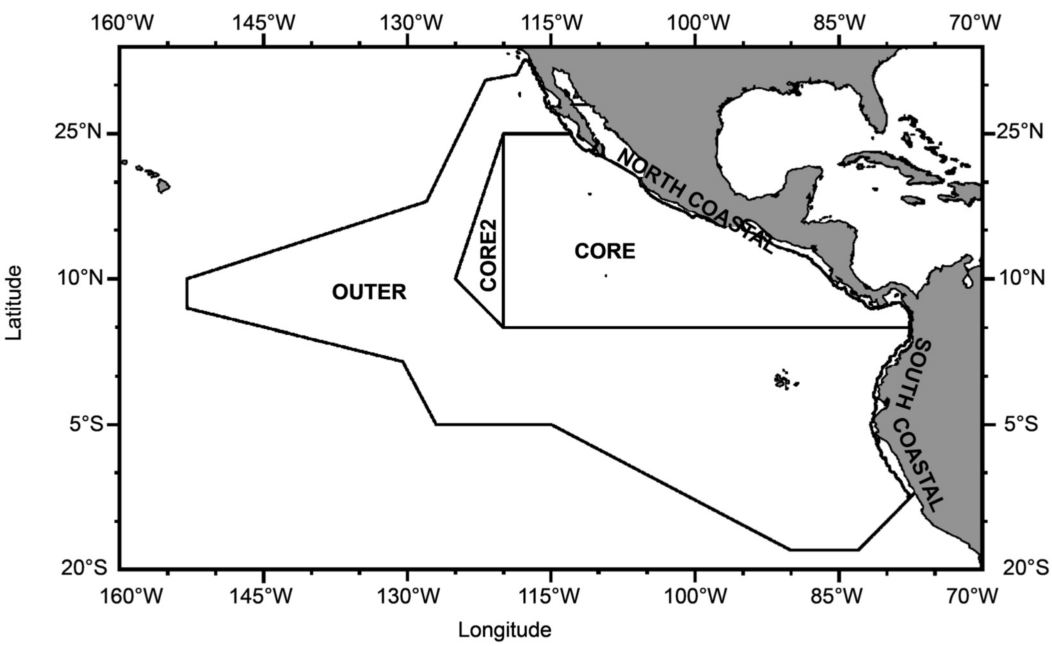


Fig. 1. Strata for the STAR06 cruise (used with permission; Gerrodette *et al*., 2008). The 'core' area was expanded to include the 'core2' area during the 2003 and 2006 surveys.

were detected by observers before there was a significant reaction of the dolphins to the vessel (Au and Perryman, 1982; Hewitt, 1985). Therefore, observer search was based on the assumption that the probability of detection on the trackline, $g(0)$, is 1.0. While 'on effort,' two observers search through 25× pedestal-mounted binoculars, one on each side of the flying bridge, from 90° abeam to the centerline. (In the early years, observers searched out to 10° on the other side as well, to ensure some overlap of effort near the line.) The third observer searches by naked eye or with hand-held 7× binoculars over the whole 180° in front of the ship. Data recorded include the observers on duty, and sighting conditions, such as sea state, swell height and sun angle. When a group of cetaceans is sighted, the angle and reticle to the sighting are recorded (radial distance to the sighting is computed per Kinzey and Gerrodette, 2001), and the observer team typically goes 'off effort' and directs the ship to leave the transect line and approach the dolphins. The purpose of 'closing' on the sighting is to identify the proportion of each species present in the group (because spotted and spinner dolphins often occur in mixed-species schools in the ETP) and to obtain the best possible estimates of school size. Experiments have shown that both kinds of data are compromised if the ship remains on the trackline and does not close on the sighting (Schwarz *et al*., 2010).

Estimating the size of dolphin schools is a crucial but difficult component of absolute abundance estimation. The accuracy of group size estimates made by research vessel observers varies considerably from observer to observer, and from group to group for a single observer. Research vessel observers' estimates have been compared extensively to counts from aerial photographs (Gerrodette *et al*., 2002; Gerrodette *et al*., 2018). On average, over all schools and all observers, group size is estimated accurately for schools of up to about ten dolphins. Above that number, group size tends to be underestimated, and the tendency to underestimate increases with group size. A group of 50 dolphins is underestimated by about 10% on average, but a group of 500 dolphins is underestimated by about 50% on average. Moreover, group size estimates are highly variable, with CVs > 0.5. To reduce this variance and to improve accuracy, the NMFS has used three main strategies. First, during pre-cruise training, observers learn group size estimation techniques. They practice estimating group sizes using photographs, videos and computer simulations. Second, after the ship approaches a sighting, the three on-duty observers make independent estimates of group size. The mean of the three independent estimates is used as it is less variable than single estimates. Third, the tendency of each observer to under- or over-estimate group size has been assessed with aerial photographs of the schools, as described above. Each observer's estimates are adjusted according to his/her individual tendency, and this improves the overall accuracy of group size estimation. These procedures also allow group size estimation error to be included in the variance of the estimate of abundance.

Abundance has been estimated from these survey data using a multivariate extension of conventional line transect methods (Gerrodette and Forcada, 2005; Gerrodette *et al*., 2008). This methodology is used to account for covariate effects on the estimated probability of detection (Buckland *et al*., 2004). Covariate effects considered in the analyses include: school size, sea state, swell height, time of day, survey ship, sighting cue, method of sighting, presence/absence of glare on the trackline, and presence/absence of seabirds.

**Advantages and disadvantages of the current methods**

The above methods have the considerable advantage that they are tried and tested. The target species are appropriate for ship-board surveys because they form large, easily detected schools, and a wide strip can be surveyed using the pedestal-mounted 25× binoculars. In good conditions schools are likely to be detected before any significant response to the vessel occurs. Movement of animals (independent of the vessel) will generate some upward bias in estimates (Glennie *et al*., 2015), and although this bias may not be negligible, incorporation of an explicit animal movement model into the distance sampling methodology allows for bias correction (Glennie *et al*., 2017). In addition, although it can be difficult to estimate group size and species proportions in mixed-species groups, as noted above, aerial photographs of a sample of schools are used to quantify and correct for bias.

There are three main shortcomings of the current methodology. First, Barlow (2015) has conducted analyses that indicate that $g(0)$ might be appreciably below one in all but the best sighting conditions, which may be linked to a reduced window in which a school is available for detection in poorer sighting conditions together with responsive movement. Second, it is costly to conduct effective shipboard surveys over such a large study area, even absent concerns about estimation of $g(0)$. Therefore, conducting surveys in blocks of several years at a time, as has been done previously (e.g. 1986–1990 and 1998–2000), or for multiple years in general, as has been recommended to obtain reliable trend estimates (Punt, 2013), may be prohibitively expensive. Finally, despite the extensive resources dedicated to the surveys, the level of precision of the abundance estimates remains problematic (Gerrodette *et al*., 2008). The precision of the estimates has improved over time, and the most recent five surveys achieved a median CV of 0.17 for the northeastern spotted dolphin and 0.24 for the eastern spinner dolphin. Despite this, it is not possible with the data of these five surveys to distinguish between an expected growth rate of 0.04 (Reilly and Barlow, 1986) and no population increase (i.e. 95% confidence intervals contain both 0 and 0.04), and 95% confidence intervals for growth rates based on the full 10 years of surveys contain the value 0 for both species (see table 13 of Gerrodette *et al*., 2008).

**Suggestions for improvement within the current framework**

Changes in field methods might be made to evaluate and estimate $g(0)$, and improve precision of the abundance estimates (see Oedekoven *et al*., 2018, for some detailed survey design considerations). If $g(0)$ is less than one, using a double-platform approach may allow its estimation (Borchers *et al*., 1998), particularly if the apparent effect on $g(0)$ of sea state may arise due to responsive movement of schools prior to detection. Attempts to study responsive movement in 1998 used an observer searching with 25×

binoculars from a higher 'tracker' platform but failed to detect groups appreciably before the primary search team. However, a drone or helicopter might provide a more effective tracker platform, operating ahead of the survey vessel, and setting up trials for the main observation platform. This allows estimates to be corrected for both responsive movement and $g(0)$ (Buckland and Turnock, 1992). Improved designs based on adaptive sampling may be able to contribute to higher precision, although gains would be expected to be rather modest (Pollard *et al*., 2002). However, it is noted that any changes risk compromising comparability between new and existing time series of abundance estimates.

Model-based analysis methods, applied to data generated under the current survey design, may help to resolve issues associated with the estimation of g(0), and perhaps also provide estimates of abundance with greater precision. As regards $g(0)$ estimation, Barlow (2015) used a generalised additive model approach to estimate the effects of factors such as sea state on the probability of a sighting. Coefficients from the fitted model provided $g(0)$ correction factors for poor sighting conditions, relative to the best sighting conditions. These correction factors were applied to an estimate of absolute $g(0)$ for the best sighting conditions to obtain estimates of $g(0)$ in poor sighting conditions. As regards improving precision, model-based methods are useful both for modeling encounter rate and for modeling the detection function (Buckland *et al*., 2004). In the latter case, using multiple covariate distance sampling methods, it is possible to jointly model data from different species, with species as a factor in the detection function model, to improve precision (e.g. Barlow *et al*., 2011). In addition, given that a larger source of variance is encounter rate (Gerrodette *et al*., 2008), encounter rate modeling also merits more attention, especially in light of recent developments in spatial distance sampling methods (e.g. Yuan *et al*., 2017).

Model-based methods may also be useful if the precision of group size estimates varies with specific covariates because this is not taken into consideration with the current methods and could cause bias. Whether the use of 'uncorrected' group size could lead to a large amount of bias in the estimates of abundance depends on the magnitude of the error in group size and the extent to which the effective strip width depends on the true group size. This source of bias can be minimised by taking into consideration the distribution of uncertainty about observed group size, as a function of covariates, when computing a Horvitz-Thompson-like estimator of abundance (Borchers *et al.,* 1998). Another option for adjusting the estimate of effective strip width for uncertainty in group size would be to estimate the detection function using an errors-in-variables type of model.

Finally, there could be several benefits to further decomposing the precision of the abundance estimates generated under the current survey design according to all the sources of uncertainty. In addition to the variance components attributable to encounter rate, effective strip width (including $g(0)$ uncertainty), and group size, there is uncertainty due to measurement error in perpendicular distances, calibration factors for correcting distance and survey modes, and process error arising from inter-annual/seasonal variation of spatial distribution of dolphins. Estimating these other sources of error and incorporating these estimates into the estimated abundance error would lead to more realistic estimates of overall uncertainty, which could have implications for conservation and management. It could also improve understanding of the main causes of uncertainty and provide information relevant to the design of future surveys, potentially improving precision of the abundance estimates.

In summary, the main suggestions for improving the current framework are:

- Adapt field methods to use a double-platform approach using a tracker platform that can operate with sufficient autonomy and in a range of survey conditions, so that $g(0)$ can be evaluated, and if necessary, estimated.
- Account for covariate effects on encounter rate using model-based methods.
- Improve the estimation of the detection function by pooling data across species (with species as a factor-type covariate) and taking account of uncertainty associated with group size estimates via model-based methods.
- Obtain more realistic estimates of overall uncertainty by estimating all sources of error (e.g. measurement error, calibration factors).

**Ship-based alternatives – use of purse-seine vessels**

With a designed, randomised survey to ensure that units of survey effort are placed randomly with respect to the distribution of groups of animals (Buckland *et al*., 2001; 2004), data might be collected aboard tuna purse-seine vessels to either supplement data collected by research vessels, or as the primary data source for abundance estimation. The best option for a purse-seine vessel survey to produce abundance estimates of similar quality and precision as those of the previous NMFS surveys would be for the commercial vessel survey to replicate all aspects of the NMFS survey methods and design (with the obvious exception of use of the same vessels), including: the number of vessels, the amount of search effort, the set-up of the observation platforms, the use of specially trained observers, and the calibration of observers' estimates of group size by aerial photogrammetry. Ideally, a survey using purse-seiners would be designed to yield unbiased estimates of absolute abundance, and therefore biases due to the use of different vessels, and possibly different observers, would be minimised. Whether such a survey would be advantageous depends in part on vessel cost, which is the most costly aspect of ship-based surveys, and which for purse-seine vessels will vary with the size and age of the vessel, and country-specific costs of fuel and insurance, among other factors.

If purse-seine vessel time for surveys were to be provided (e.g. donated) by vessel owners, it might be that more than two vessels would be involved in the survey, each for a shorter period of time than the sea-days of the NMFS vessels. The optimal number of vessels that should participate in the survey would need to be evaluated based on target CVs and logistical constraints. Several aspects of a many-vessel survey, however, are worthy of further discussion. First, if each commercial vessel were to survey during a portion of a fishing trip, it would be important to determine the optimal allocation of survey segments from a randomised design to each vessel so as to minimise transit time from the fishing location to the survey location, taking into consideration spatial gradients in dolphin abundance (Reilly, 1990; Redfern *et al*., 2008; Forney *et al*., 2012) and the constraint that vessels should all operate at the same time of year to avoid any potential biases due to dolphin population movement. A simulation using historical commercial vessel fishing trip trajectories, in combination with historical NMFS survey tracklines, or new survey tracklines, and information on dolphin spatial distributions, could be conducted to determine how best to allocate blocks of survey segments from an ETP-wide design to individual tuna vessels in time and space. In addition, minimising heterogeneity in the data as much as is practical through survey design would be important, given concerns about perfect detection on the trackline (Barlow, 2015). Without the assumption that $g(0) = 1$, the property of 'pooling robustness' (Buckland *et al*., 2004) would no longer hold and it could not be assumed that unmodeled heterogeneity in the data would have little effect on the estimation.

Finally, because of an evasive response of dolphins to tuna vessels (e.g. Pryor and Norris, 1978; Lennert-Cody and Scott, 2005 and references therein), which varies spatially across the ETP, it would be important to have a commercial vessel survey design that allows for testing of the assumption that $g(0, \mathbf{z}) = 1$ for covariates $\mathbf{z}$, and if necessary, estimation of $g(0,\mathbf{z})$. This would require a different survey design compared to previous NMFS surveys, as well as additional equipment. Double-count survey designs, from which $g(0,\mathbf{z})$ can be estimated by mark-recapture distance methods (e.g. Borchers *et al*., 1998; Buckland *et al*., 2004), typically involve two teams of observers; for the more robust approaches, one of these teams searches at greater distance, possibly from a helicopter or using video from a drone flying ahead of the ship, while the other carries out normal search. Although commercial vessels that are suitable for a survey tend to carry a helicopter, it would first need to be ascertained whether the helicopters elicit an evasive response in dolphins, given that helicopters are used during fishing operations. In addition, observer safety may be a concern with respect to helicopter use; if helicopter use is restricted to good conditions, the data would be of limited value for quantifying $g(0,\mathbf{z})$, which is expected to be lowest in poor conditions. Drones might provide a less disruptive vehicle for detecting and tracking schools in a double-count survey design, but might also be restricted to good sighting conditions.

## AERIAL SURVEYS

Aerial line transect surveys were conducted in the ETP in the 1970s (Smith, 1981; Holt and Powers, 1982) but because of safety concerns and because the range of shore-based aircraft could not cover the entire offshore area, they were discontinued. For these reasons, surveys using observers on board manned aircraft are not considered in this review.

### Digital aerial surveys

*Manned aircraft*

Commercial manned digital aerial survey methods were developed in the United Kingdom to provide survey data on potential impacts of developing offshore wind farms. Early tests demonstrated their effectiveness for census of seaduck at one coastal site, with abundance estimates that exceeded those of traditional visual aerial survey methods (Buckland *et al*., 2012). Digital survey methods have since replaced visual aerial methods for seabirds in offshore waters of the United Kingdom, Germany, Denmark, and increasingly in the eastern United States with thousands of sorties now flown (Thaxter *et al*., 2016; Weiß *et al*., 2016; Williams *et al*., 2015). Digital video aerial survey methods have been found to give comparable results to dedicated visual aerial survey methods for harbour porpoise *Phocoena phocoena* in the United Kingdom (Williamson *et al*., 2016), and for other marine megafauna (Williams *et al*., 2015; Gordon *et al*., 2013).

Two technologies have emerged for commercial census by digital aerial survey: high resolution video and high resolution digital still imagery. In general, the video methods use bespoke camera rigs to scan a strip transect using four cameras in a comb pattern over the sea. Stills methods usually use medium-format photogrammetry cameras to sample plots (or quadrats) or transects at sea. For seabird surveys, cameras ideally collect images at a ground sample distance (GSD) of 2–3cm, and this allows species identification rates of at least 80% of all seabird species in the United Kingdom, and considerably higher rates for cetaceans. Lower resolutions of 3–5cm also achieve high identification rates for cetaceans. The higher camera resolutions are achieved while flying at 550m above sea level (a.s.l.) for digital video methods and 270–400m a.s.l. for digital stills, depending on the GSD used.

Both methods in the United Kingdom use a two-phase method for analysing digital data generated. The first phase requires a review of all material, with 10% or 20% of all material subjected to a random blind audit, and robust procedures for handling failed audits. The second phase requires all objects to be assigned to the lowest order taxon possible. Again 10–20% of all objects are subjected to a randomised blind audit, with procedures for handling failed audits. Digital stills and digital video methods have attempted to use automated methods for detection and identification of objects using machine learning methods, with varying success. While detection methods are reasonably successful in calm sea conditions, they have much poorer accuracy at higher sea states, particularly for marine mammals. Automated methods for identification of objects require considerable human intervention and oversight, negating the potential efficiency benefits of such methods. Although success so far has been low in these methods, it is likely that more sophisticated artificial intelligence algorithms will ultimately be able to replicate the undoubted accuracy of experienced human operators.

Digital aerial methods offer a number of advantages over conventional aerial survey methods:

- Because the aircraft operate at greater altitudes and have fewer crew, digital methods are considerably safer than visual aerial surveys.
- Detection rates are uniform across the whole image for digital methods, meaning that it is not necessary to account for missed detections using distance methods, and double-review methods are simpler.

- All individual animals can be counted and group sizes do not influence detection rates in digital methods, removing the need to account for group size detection bias when estimating abundance.
- A permanent record of the survey can be kept for subsequent analysis should the need arise with digital aerial methods.
- Bespoke rigs are used to angle digital cameras away from sun glare and avoid detection problems of fixed camera systems and visual aerial survey methods.
- Digital video methods are still effective at higher sea states, when compared to digital stills and visual methods, although there is some unpublished evidence for lower detection rates in video methods, mainly for sub-surface marine mammals at higher sea states.
- Identification rates are higher for digital methods.

Some issues remain for digital aerial methods when compared to other methods and the survey requirements of the ETP surveys:

- In general, digital aerial methods are more expensive than visual aerial methods but typically cheaper than dedicated ship-based methods in like-for-like surveys (Thaxter *et al.,* 2016).
- While automated data review methods are available, they are still not sufficiently efficient compared to manual review. Considerable investment is required to develop methods that will provide significant time and cost savings.
- Availability bias for diving seabirds and cetaceans in digital survey methods is acute but difficult to account for. There exists a theoretical method for measuring this bias *in situ* using digital video methods which so far is untested. This is most likely to be effective for cetacean species with relatively short dive cycles (typically 2–3 minutes or less). No method exists for measuring this bias *in situ* for digital stills methods. Generic methods can be used, based upon known dive rates where these exist, for estimating availability bias for digital survey methods (Webb *et al*., 2015).
- Although it would be possible to survey the majority of the ETP survey area, as with visual aerial survey methods, the endurance of the aircraft used for these surveys is limited and insufficient for reaching the furthest limits safely from suitable airports. While deploying helicopters from boats offshore is possible, helicopters have been found not to provide a sufficiently stable platform for digital transect-based surveys. Some aircraft are able to re-fuel mid-air, and one of the aircraft used for digital video aerial surveys has a pilot-less version which increases the endurance and safety significantly, but see below.
- While good species identification is possible using this method, it is untried for the ETP species, thus in order to estimate absolute abundance, methods might be needed to estimate the species proportions within mixed-species dolphin schools, which are commonly seen in the ETP.

Commercial aircraft could potentially be fitted with cameras to gather high-resolution images. This has the advantage of low cost relative to dedicated aerial surveys, although potentially costly certification of aircraft for installing cameras might negate this advantage. The main disadvantages are: commercial aircraft fly at a much greater altitude than dedicated survey aircraft, resulting in low-quality images; commercial aircraft usually fly much faster which would compromise the number of images or frames that can be captured; and commercial aircraft routes do not sample the ETP evenly, so that spatial modelling methods will be required to extrapolate across the whole region.

*Unmanned aircraft*

The use of Unmanned Aerial Systems (UAS), also known as drones, has proliferated in the ecological survey sector in the last decade (e.g. Anderson and Gaston, 2013; Christie *et al*., 2016; Hodgson *et al*., 2016; Marine Mammal Commission, 2016; Colefax *et al*., 2018) and, much like in manned digital aerial surveys, can be used for transect or plot sampling of marine mammal distribution and abundance. UAS are available in many forms, from small multi-copter systems that carry small video cameras that have high definition or ultra-high definition (4×) resolution and save images to flash memory cards, up to military-grade fixed wing UAS that are capable of carrying much larger payloads with higher resolution cameras and server-based image storage systems. At the smallest end of the size spectrum, the camera systems are unlikely to deliver images of sufficient quality. Most attention in the use of UAS for marine census has been given to small- to medium-sized systems, such as the AH22, that are able to carry sensors of sufficient payload to capture higher quality images or video material.

Some small- to medium-sized UAS are designed to be recovered at sea, and most would need to be deployed and recovered from the deck of a ship if they were to be used to census the entire ETP. This would elevate the cost benefit considerably by the addition of the price of a mother ship that is able to reach the more distant parts of the previous study area (Fig. 1). Part of this restriction is imposed by limited access to airspace; in Europe and the United States, aviation regulations require that UAS are flown within line of sight of an operator. A further limitation on the use of such systems is their endurance, both in the number of hours that can be surveyed in a single mission and in the storage capacity for the images. The endurance of even medium-sized systems is limited to about five hours at about 100km per hour, which would mean that a survey of the ETP would be slow, unless carried out by multiple UAS. Storage capacity also limits the duration of sorties to a few hours and also means that raw image formats cannot be stored, thus reducing image quality slightly.

Military-grade systems are able to take much larger payloads and would be able to carry the payload of a commercial digital aerial survey system on board, including multiple cameras and server-based data storage systems. This gives them considerably greater endurance. Such systems would need to take off and land at commercial or military airstrips and cannot be recovered at sea at present. The *Diamond Aviation* DA42, used by HiDef for its digital video aerial surveys, has a pilotless version used for military purposes. It would have an endurance of about 15 hours and would be licensed to carry cameras and increased data storage capacity for a wider-area survey such as the ETP. To use such as system would require negotiated access to airspace of the ETP study area. At present, cost estimates for an ETP survey with military-grade UAS systems are not available. A preliminary estimate of the cost of an ETP survey with a *FlexRotor* drone, which is a commercial/civil-use drone, is less than $2M US (Johnson *et al*., 2018). However, these drones currently have an ~2,000km range and thus would require at-sea refueling on platforms of opportunity (e.g. tuna purse-seine vessels) to cover the full ETP survey area at no additional cost. Nonetheless, these figures are encouraging with regards to the potential savings that may be possible in the future with drone surveys compared to ship-based surveys.

There are several issues that remain to be worked out for surveys with unmanned aircraft. First, as with imagery from manned aircraft, there is a lack of automated detection methods that will work in a range of weather conditions. Automated detection is possible in near calm conditions, but becomes problematic in the likely sea states typical of the ETP. Second, there may be reliability issues with UASs being lost and not re-located at sea. There is also the need for development of better international airspace management to avoid collisions or interference with commercial aircraft and purse-seiner-based helicopters.

## SATELLITE SURVEYS

Very High Resolution (VHR) satellites now have the ability to capture large areas of ocean (> 1000km$^2$ per image) at a spatial resolution of 30cm per pixel (Platonov *et al*., 2013; Stapleton *et al*., 2014). Recent work on cetaceans using lower resolution imagery (50cm) has shown the utility for counting baleen whales in optimal conditions and initial tests using 30cm imagery on humpback whales have shown a clear improvement in detection, both on the surface and beneath it (Fretwell *et al*., 2014). With 30cm satellite imagery it should be possible to identify the pattern of breaching small cetaceans in relatively calm seas, although species identification is unlikely. In calm conditions the signature of the splashes will be very bright relative to the surrounding waters, and due to the radiometric resolution of the satellites, it may also be possible to automate or semi-automate the process of finding these patches for large pods of dolphins. If agreements could be made with the satellite provider, this could be a very cost-effective way to survey large expanses of ocean to give first order abundance or presence estimates, or estimated indices of relative abundance. Other advantages are the ease of use of satellites, the ability to capture extremely large amounts of imagery in any area of ocean, the non-invasive nature and the lack of logistical set-up or flight planning effort for satellites.

However, this use of satellite technology is still developing and much testing would be needed before a workable system using satellite data could be incorporated into other survey methods. There are some comparisons to be drawn between satellites and the use of high-resolution aerial survey using digital stills; each has similar drawbacks – the need for favourable sea conditions, the problem of single instantaneous image acquisition and potential problems, and the need for automation over large areas. An additional drawback of satellite imagery is that image quality is affected by cloud cover. The main differences between the two systems are the higher resolution of the aerial imagery and the greater potential coverage from satellites.

The potential cost of the highest resolution imagery could be high for large area studies unless an agreement can be gained from the satellite provider; this is more likely either over areas with less demand for imagery (open ocean) or areas where high-profile research could be conducted. As the use of this technology is unproven for small cetacean surveys, the algorithms needed for automated or semi-automated identification still need to be constructed and proven. Manual checking over 1000's of square kilometres is difficult, although crowd-sourcing the imagery might solve this in the longer term. Species identification will be impossible with satellites as the resolution is too coarse and estimating school size could be difficult without ground truthing.

## PASSIVE ACOUSTICS

Distance sampling, adapted for acoustic data, is the most commonly used approach to estimate abundance from passive acoustic data (Heinemann *et al*., 2016). Acoustic methods may be most valuable for estimating trends in relative abundance rather than absolute abundance for ETP dolphin stocks because of the difficulty of estimating group size from acoustic data. Assuming a species can be unequivocally identified by its vocalisation repertoire, to estimate trends in relative abundance from acoustic data, there are two key requirements for acoustic distance sampling methods, as follows.

The first is that detection probability can be estimated as a function of horizontal distance from the 'cue' (e.g. vocalisation count) to the acoustic instrument. To obtain detection as a function of horizontal distance, the depth of the cue (i.e. animal or group of animals) is often assumed, and this may bias the estimated detection function. In addition, the detection probability has to be corrected for the false detection rate (i.e. vocalisations that were incorrectly assigned to the target species during data processing). Although sound-propagation modelling has been used to estimate detection range in order to estimate distance to the cue, accurate estimation of range from these models is currently challenging. This is especially true for highly directional signals like echo-location clicks. A drifting vertical hydrophone array can be used to estimate range empirically which holds more promise than model-based range estimation.

The second is that density estimation methods can be based on individual-count methods, group-count methods, or cue-count methods. Individual-count methods are typically not practical because individuals within a group cannot be discriminated acoustically. Group-count methods require an estimate of group size, and methods to estimate group size from only acoustic data currently do not exist. (Group size is often obtained from concurrently-collected visual survey data.) Methods to convert cue counts to individual density require estimates of the cue production rate (vocalisations per unit time) under environmental and social conditions that are likely to be encountered during the survey.

At present, statistical methodological challenges exist for estimating abundance from acoustic data collected with slow-moving autonomous platforms. The accuracy of estimated trends in abundance will depend on the number and location of acoustic platforms used in the survey, and whether parameters such as detection probability, vocalisation rates, and area effectively surveyed can be estimated or assumed to be constant. The number of surface drifters can be increased at relatively little cost to obtain the number of detections to achieve a desired power to detect changes in abundance. For repeated long-duration surveys, such as might be conducted for ETP dolphin species, a passive acoustic system that was integrated into a glider or float would be preferable from a data-collection perspective. Buoyancy-driven floats and gliders can collect data continuously for weeks to a few months. Floats drift with the current at a specified depth; gliders can control both vertical and horizontal position (average speed is ~0.5 knots). Techniques for categorising whistles of ETP dolphin species are being developed (Oswald *et al*., 2004; 2007) but more research is needed to reliably distinguish species. Assuming that detection distances can be measured for each acoustic detection, the remaining key uncertainties are the degree to which acoustic behaviour and group size vary over time.

Similar data processing challenges are encountered when processing passive acoustics data, as with processing of high-resolution imagery data. As with all passive acoustic systems, the large volumes of data generated require processing to remove unwanted noise, identify vocalisations of the target species and locate those vocalisations in space. This data processing must be done by skilled analysts and specialised computer software.

## COMPOSITE METHODS

There are many ways in which ship-based surveys might be combined with 'auxiliary' sources, either simultaneously or at different points in time, to obtain a 'composite' method. The use of composite methods would be an effective use of other line-transect data sources that may not require costly ship time but, as discussed above, are unlikely by themselves to provide estimates of absolute abundance in the near future due to limitations on the ability to identify species and/or estimate group size. Auxiliary source availability biases may be reduced or mitigated with technological advances; however, the biological sources of biases would remain. The large CVs on mean group size from the NMFS surveys (Gerrodette *et al*., 2008) provide further motivation to explore an index based on encounter rate. Auxiliary sources include passive acoustics, high-resolution imagery from helicopters, drones and satellites, and data collected by observers aboard tuna vessels. The possible reasons to develop auxiliary sources for composite methods include:

(1) Correct any bias in ship-based estimates. As noted above, a drone could operate ahead of a ship, providing a second platform, and data from which corrections for responsive movement and for $g(0)$ may be estimated. Drones or helicopters also could be used to check school sizes and species identifications and proportions in mixed schools, and hence estimate bias in observer group-size estimates by species.

(2) Improve precision of the ship-based estimates.

(3) Develop annual indices of relative abundance from which trends can be estimated at lower cost than for ship-based surveys.

Points (2)–(3) are discussed in more detail below.

### Improving precision

The variance on encounter rate is one of the largest components of the variance of the estimated trend for ETP dolphin species (Gerrodette *et al*., 2008). Given this, there are two ways in which precision might be improved.

First, precision might be improved by increasing the number of dolphin group sightings, $n$, (e.g. see variance decomposition eq. 3.3 of Buckland *et al*., 2004). Increasing $n$ can be done in several ways:

(a) Use high-resolution imagery from a short-range drone, operated from the survey vessel, to increase the effective area surveyed. Detections made by the drone would be added to those made by the vessel, and their location recorded as distance from the ship transect.

(b) Add subsidiary transects in the vicinity of shipboard transects using a short-range drone to increase the total transect length.

(c) Use satellite data to estimate the proportion of dolphin schools detected, $P_a$. If the ship-based survey estimator is viewed as a Horvitz-Thompson estimator (e.g. eq. 2.17 of Buckland *et al*., 2004), then dolphin group size might be estimated from the ship-based data but $P_a$ from the satellite data. Using the satellite data to estimate $P_a$ would increase $n$ because the imagery represent strip transects. Also, this would avoid the potential problem of $g(0)\neq1$ (Barlow, 2015); having to estimate $g(0)$, which has been assumed to be 1.0 (Gerrodette *et al*., 2008), would increase the variance of the estimated trend by increasing the variance of density (eq. 3.3 of Buckland *et al*., 2004).

In terms of allocation of survey effort with respect to (a)–(b), the most effective allocation to increase $n$ would be to adopt a stratified survey design and allocate proportionally more survey effort to high-density areas. For all three scenarios, experiments on estimation of availability bias for adjusting encounter rate estimates, such as those outlined in Johnson *et al*. (2018), would need to be conducted.

Second, auxiliary sources might be used to spatially 'extend' the sparse shipboard survey data such that the time series of shipboard estimates could be combined with a time series of annual auxiliary indices (see below) to improve the precision of the trend estimate. For example, surface drifters or gliders might be used to gather acoustic data, and jointly modelled with ship-based survey data or high-resolution imagery data, using a model-based approach. A similar strategy may allow utilisation of tuna vessel observer data together with research vessel or unmanned aerial survey data. Exploratory analyses using existing tuna vessel observer data and research vessel data may be useful in this regard. It is noted that the tuna vessel observer data have good spatial coverage (e.g. Lennert-Cody *et al*., 2016), and acoustic/high-resolution imagery data might have the same advantage, and thus, spatially-varying calibration against relatively sparse ship-based sightings data would in principle allow conversion to absolute density.

**Relative abundance indices based on auxiliary sources**

Because of anticipated lower costs of collecting auxiliary data in the future (Johnson *et al*., 2016), auxiliary sources could be used to develop a relative abundance index on a more frequent temporal basis (e.g. annually). This relative index could be combined with estimates from infrequent ship-based surveys, which might allow for more informed management. ETP dolphin species have low population growth rates (e.g. Reilly and Barlow, 1986), therefore the relative index would need to be precise enough to allow detection of small changes over time. Bias in the index would be tolerable as long as the bias was temporally invariant. To evaluate the assumption of temporally-invariant bias, the relative index would need to be compared periodically to a time series of ship-based survey estimates, even if the shipboard survey estimates were only conducted infrequently (e.g. every 5 years). This would only be informative, however, if the precision of the ship-based estimates were high.

Similarities in the existing abundance and the encounter rate trends for four dolphin stocks, two highly involved in the purse-seine fishery on tunas associated with dolphins (northeastern spotted and eastern spinner dolphins), one stock less involved in the fishery (short-beaked common dolphin) and one rarely involved in the fishery (striped dolphin), suggest that an encounter rate-based index may be worth further consideration (Fig. 2). Mean-scaled estimates of abundance and encounter rate show nearly identical overall trends for the northeastern spotted dolphin and the striped dolphin, and similar trends for the eastern spinner dolphin. It would be useful to conduct analyses with the existing survey data to further evaluate options for relative indices, including encounter rate of all dolphins. Relative abundance indices that might be considered are shown in Table 1. However, indices based on encounter rate require the strong assumption that group size is constant (Table 1). If encounter rate indices were to be used, to be precautionary, it might be possible to develop an index of school size that could be compared statistically among surveys to evaluate the assumption that mean group size was constant or had not changed to a meaningful extent.

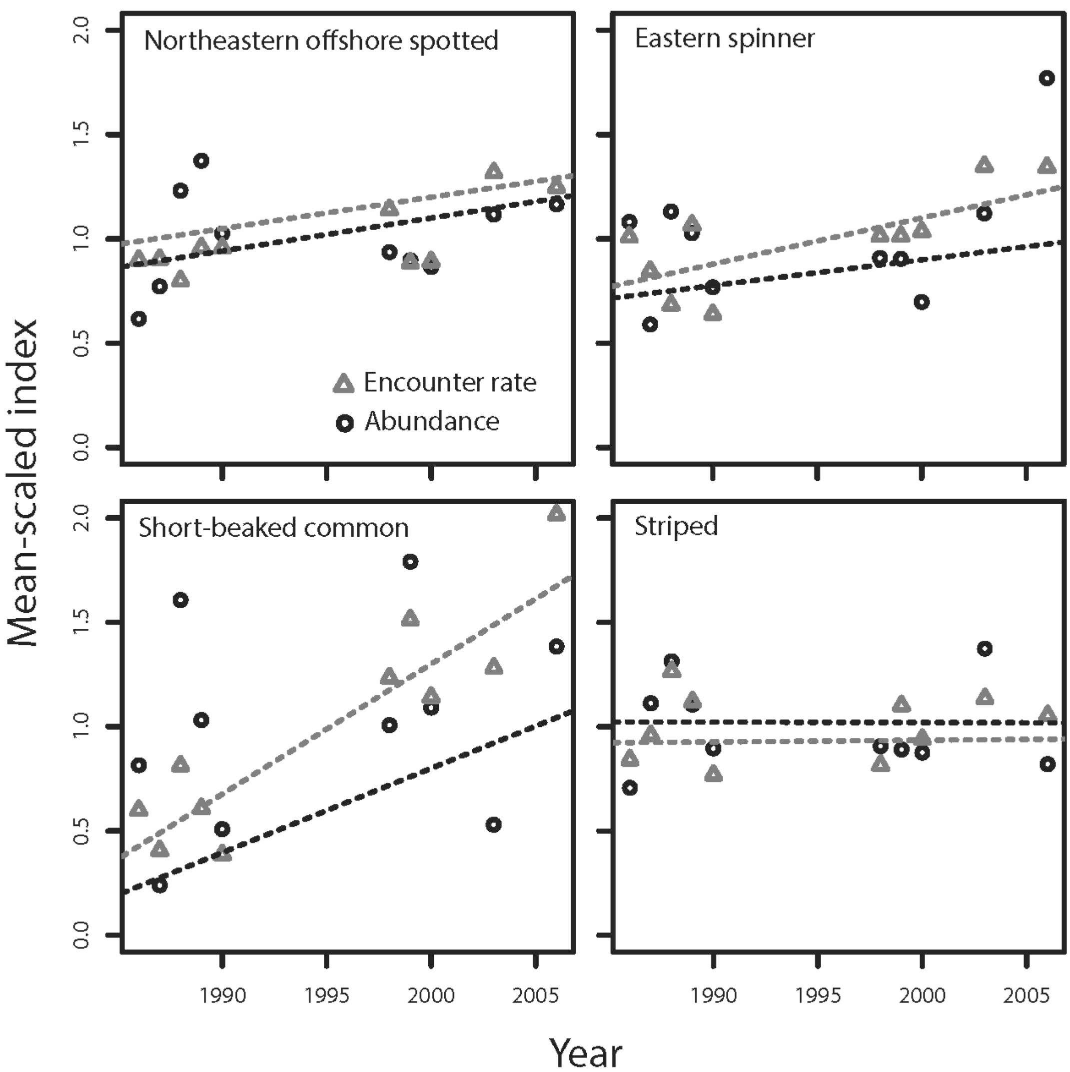


Fig. 2. Mean-scaled indices *versus* year for four dolphin stocks in the ETP. Shown on the y-axis is y/average(y), where y is either encounter rate (grey triangles) or abundance (black circles) (both from tables in Gerrodette *et al*., 2008). The dashed lines are the fitted lines obtained from weighted least squares, with weights = $1/(SE)^2$ (SE = standard error (y), also from tables in Gerrodette *et al*., 2008). The large difference in fitted lines for the short-beaked common dolphin, relative to the point patterns, is due to different weighting of the various data points; i.e. in some instances SE for encounter rate was low but the SE for abundance was high or *vice versa*.

Table 1

Types of relative abundance indices that might be considered for ETP dolphin stocks. 'Encounter rate' refers to encounter rate of dolphin schools, not individual animals. Assumptions regarding constant biases differ with the index type, as well as with the auxiliary data source (see discussion of availability biases in Johnson *et al*., 2018).

| Index type | Assumptions | Auxiliary source data |
|---|---|---|
| Encounter rate, all dolphins | Species composition of groups and group size are constant.<br>Availability biases constant. | Passive acoustics<br>High-resolution imagery (long-range drone, satellite) |
| Encounter rate, species | Group size is constant.<br>Availability biases constant. | High-resolution imagery (long-range drone) |
| Abundance, all dolphins | Species composition of groups is constant.<br>Availability biases constant. | High-resolution imagery (long-range drone) |

The similarity of existing abundance and encounter rate trends (Fig. 2) also suggest priorities for research and development for high-resolution imagery. If relative abundance indices based on encounter rate are possible, then developing methods for estimating dolphin group size might be given lower priority than developing methods to identify species. Encounter rate is based on presence/absence of dolphin schools and thus the task of identifying a dolphin school and the species within the school comes down to identifying at least one individual of each species in the school. This should simplify to some extent the problems associated with availability bias.

## DISCUSSION

To obtain future abundance estimates for ETP dolphins, the safest and most effective option would be to replicate past research vessel surveys to the extent possible. However, it is not clear that this is the best option. These ship-based surveys are costly, and precision of the abundance estimates is not high. Of course, other approaches would incur development costs, and unforeseen problems may arise. And, unless both research vessel surveys and any new methods provide unbiased estimates of abundance, estimates from a new approach are unlikely to be directly comparable with past estimates. Implementing a new approach together with a research vessel survey would allow the two approaches to be calibrated, but the cost of the exercise would be high, and unless it was repeated over several years, the calibration factor would be imprecisely estimated. A possible alternative would be to implement a less-costly approach (e.g. using drones or satellite images) with the aim of obtaining an annual index of relative abundance, together with an occasional full survey (perhaps using methods closely comparable with past research vessel surveys) to attempt to estimate absolute abundance.

Of the potential new approaches discussed in this review, perhaps the most promising in terms of cost, practicality and precision is the use of high-resolution video taken from long-range drones. Suitable drones have until recently been the preserve of the armed forces, but are now becoming commercially available. A pilot survey followed by annual surveys for perhaps four or five years would allow a new time series of abundance estimates to be generated quickly. If the drones can be flown from land rather than from a ship (which is feasible for military drones, given their range), after initial development costs, this option could have an appreciably lower cost than research vessel surveys, even after accounting for the narrower strip width of high-resolution imagery (Johnson *et al*., 2018). Satellite surveys may be a viable alternative, too, especially if resolution improves to the point that species identification becomes reliable. They would be dependent on obtaining images when sighting conditions are good over a large region, and effective software would be needed for reliable automated search of dolphin schools in vast images.

This review has focused on transect methods for fishery-independent data, however, there are other options for abundance estimation, including mark-recapture methods. Advantages and disadvantages of the use of mark-recapture methods, such as close-kin, for estimating abundance have been discussed for ETP dolphin stocks, and an outline of a pilot study using tuna vessels to assist with recaptures has been presented (Johnson *et al*., 2018). Although mark-recapture methods may be less costly than ship-based line-transect surveys (once research and development phases are completed), problems that may arise when applying these methods to ETP dolphin stocks may be expected from several sources: large population size; heterogeneous and non-independent probabilities of capture and recapture; possible errors in matching marked animals; tag loss; and, difficulty in defining the population that is being estimated, given the potential for movement in and out of the ETP. Buckland and Duff (1989) summarised the problems of estimating numbers of Antarctic minke whales by mark-recapture methods; their population size is similar to that of the main ETP dolphin stocks. The recently proposed close-kin mark-recapture methods (Bravington *et al*., 2016) may increase the number of recaptures, but a large number of tagged individuals would still be required.

The estimates of abundance are used for two main purposes in the management of dolphin stocks in the ETP, and these will determine what attributes of the abundance estimates are most important. The first is to evaluate if the stock has rebuilt from the depleted levels caused by the high levels of historic mortalities (e.g. Lo and Smith, 1986; Wade, 1995). The second is to calculate dolphin mortality limits that are used to ensure that current mortality levels are sustainable (IATTC, 2006). To evaluate the current stock status and whether the population has rebuilt, a population dynamics model is fit to the abundance estimates conditioned on the historical mortalities to reconstruct the population trajectory (Hoyle and Maunder, 2004; Wade *et al*., 2007). The population dynamics model is also used to define a reference point or rebuilding target. The current abundance estimate from the population dynamics model is compared to the reference point to determine the status of the population. The abundance estimates can be treated as indices of relative abundance and the proportionality constant (catchability) can be estimated as a parameter of the model to account for consistent biases in the estimates of abundance, but this will reduce the precision on the estimates from the population dynamics models. Therefore, the abundance estimates can be absolute or relative, but it is preferable that the abundance estimates are absolute and unbiased. However, abundance estimates are only one component of the population dynamics modelling, and the population dynamics models are based on many assumptions that are uncertain (e.g. Hoyle and Maunder, 2004), so the stock status may still be uncertain even if accurate estimates of absolute abundance are available.

The dolphin mortality limits[11] take uncertainty into consideration and lower limits are obtained when the precision of the abundance estimates is poor. Therefore, the precision of the estimates of abundance are an important consideration when choosing a method to estimate abundance. Because historic estimates of abundance have been imprecise, methods have been used to combine multiple survey estimates together to try to improve precision. This is most appropriately done using a population dynamics model since the surveys have been conducted in different years and the population dynamics model automatically takes the changes in abundance over time into account. The population dynamics model also can predict the abundance in years after the last survey estimate of abundance. However, the longer the time since an abundance estimate is available, the less reliable the management benefit of the dolphin mortality limits.

In conclusion, the following recommendations are put forward for methods for estimating abundance of ETP dolphin stock status from dedicated research surveys.

- For immediate management needs, a ship-based survey is the only reliable option. Survey methodology should:
    - Evaluate, and if necessary, adjust for imperfect detection on the trackline;
    - Consider an errors-in-variables approach to take the uncertainty of group size estimates into consideration when estimating the detection function;
    - Incorporate approaches to reduce variance, including: encounter-rate modeling using spatial distance methods, and joint modelling of the detection function with data from multiple species.
- The following pilot studies for development of relative abundance methods that might be considered in composite approaches should be conducted in tandem with any future ship-based survey:
    - Encounter rate estimation using high-resolution imagery from drones and from satellites;
    - Encounter rate estimation with passive acoustic drifters.

[11] The dolphin mortality limits are calculated as 0.1% of the estimate of the minimum abundance estimate ($N_{min}$) (IATTC, 2006), where $N_{min}$ is defined as the 20th percentile of a log-normal distribution based on an estimate of the number of animals in the stock (Barlow *et al*., 1995).

- For the longer term, it should be a top research priority to develop methods of estimating relative abundance that are less expensive than frequent ship-based surveys so that composite approaches to abundance estimation can be used.

## ACKNOWLEDGEMENTS

Special thanks to the participants of the IATTC Workshop on Methods for Monitoring the Status of Eastern Tropical Pacific Ocean Dolphin Populations, 18–20 October 2016, La Jolla, California, for insightful and productive discussion, and specifically to Megan Ferguson for her contribution to survey cost estimates produced by the break-out group on UAS surveys. Funding for the workshop was provided by the European Union and the Pacific Alliance for Sustainable Tuna. We thank Wayne L. Perryman for helpful comments on a previous draft of this manuscript, Lisa Ballance and Toby Garfield for reviewing this manuscript, and two anonymous reviewers for comments that improved this manuscript.

## REFERENCES

Anderson, K. and Gaston, K.J. 2013. Lightweight unmanned aerial vehicles will revolutionize spatial ecology. *Front Ecol. Environ.* 11: 138–146. [Available at: *https://doi.org/10.1890/120150*].

Anganuzzi, A. A. and Buckland, S.T. 1989. Reducing bias in estimated trends from dolphin abundance indices derived from tuna vessel data. *Rep. int. Whal. Commn.* 39:323–34.

Au, D. and Perryman, W. 1982. Movement and speed of dolphin schools responding to an approaching ship. *Fish. Bull.* 80:371–9.

Barlow, J. 2015. Inferring trackline detection probabilities, *g*(0), for cetaceans from apparent densities in different survey conditions. *Mar. Mamm. Sci.* 31, 923–43.

Barlow, J. Swartz, S.L., Eagle, T.C., and Wade, P.R. 1995. US marine mammal stock assessments: Guidelines for preparation, background, and a summary of the 1995 assessments. NOAA Tech. Memo. NMFS-OPR-6. 73pp.

Barlow, J., Ballance, L.T. and Forney, K.A. 2011. Effective strip widths for ship-based line-transect surveys of cetaceans. NOAA Tech. Memo. NMFS-SWFSC-484.

Borchers, D.L., Buckland, S.T., Goedhart, P.W., Clarke, E.D. and Hedley, S.L. 1998. Horvitz-Thompson estimators for double-platform line transect surveys. *Biometrics* 54: 1,221–37.

Bravington, M.V., Skaug, H.J. and Anderson, E.C. 2016. Close-kin mark-recapture. *Stat. Sci.* 31: 259–74.

Buckland, S.T. and Anganuzzi, A.A. 1988. Estimated trends in abundance of dolphins associated with tuna in the eastern tropical Pacific. *Rep. int. Whal. Commn.* 38: 411–37.

Buckland, S.T. and Duff, E.I. 1989. Analysis of the Southern Hemisphere minke whale mark-recovery data. pp.121–43. *In*: G.P. Donovan (ed.) *The Comprehensive Assessment of Whale Stocks: the Early Years*. International Whaling Commission, Cambridge.

Buckland, S.T. and Turnock, B.J. 1992. A robust line transect method. *Biometrics* 48: 901–09.

Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., Borchers, D.L. and Thomas, L. 2001. *Introduction to Distance Sampling*. Oxford University Press, Oxford.

Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., Borchers, D.L. and Thomas, L. 2004. *Advanced Distance Sampling*. Oxford University Press, Oxford.

Buckland, S.T., Burt, M.L., Rexstad, E.A., Mellor, M., Williams, A.E. and Woodward, R. 2012. Aerial surveys of seabirds: the advent of digital methods. *J. App. Ecol.* 49: 960–7.

Christie, K.S., Gilbert, S.L., Brown, C.L., Hatfield, M. and Hanson, L. 2016. Unmanned aerial systems in wildlife research: current and future applications of a transformative technology. *Front Ecol Environ*. [Available at: *https://doi.org/10.1002/fee.1281*].

Colefax, A.P., Butcher, P.A. and Kelaher, B.P. 2017. The potential for unmanned aerial vehicles (UAVs) to conduct marine fauna surveys in place of manned aircraft. *ICES J. Mar. Sci.* 75(1): 1–8.

Dizon, A.E., Perrin, W.F. and Akin, P.A. 1994. Stocks of dolphins (*Stenella* spp. and *Delphinus delphis*) in the eastern tropical Pacific: A phylogeographic classification. NOAA Technical Report NMFS 119. 20pp.

Forney, K.A., Ferguson, M.C., Becker, E.A., Fiedler, P.C., Redfern, J.V., Barlow, J. Vilchis, I.L. and Balance, L.T. 2012. Habitat-based spatial models of cetacean density in the eastern Pacific Ocean. *Endanger. Species Res.* 16: 113–33.

Fretwell, P.T., Staniland, I.J. and Forcada. J. 2014. Whales from space: counting southern right whales by satellite. *PLOS ONE* 9, e88655.

Gerrodette, T., Perryman, W. and Barlow, J. 2002. Calibrating group size estimates of dolphins in the eastern tropical Pacific Ocean. Southwest Fisheries Science Center Administrative Report LJ-02-08. [Available at: *https://swfsc.noaa.gov/uploadedfiles/divisions/prd/programs/etp_cetacean_assessment/lj_02_08.pdf* ].

Gerrodette, T. and Forcada, J. 2005. Non-recovery of two spotted and spinner dolphin populations in the eastern tropical Pacific Ocean. *Mar. Ecol. Prog. Ser.* 291:1–21.

Gerrodette, T., Watters, G., Perryman, W. and Ballance, L. 2008. Estimates of 2006 dolphin abundance in the eastern tropical Pacific, with revised estimates from 1986–2003. NOAA Technical Memorandum NMFS-SWSFC 421: 1–45.

Gerrodette, T., Perryman, W.L. and Oedekoven, C.S. 2018. Accuracy and precision of dolphin school size estimates. Mar. Mamm. Sci., Early view. [DOI: 10.1111/mms.12506].

Glennie, R., Buckland, S.T. and Thomas, L. 2015. The effect of animal movement on line transect estimates of abundance. *PLoS ONE*. 10: e0121333.

Glennie, R., Buckland, S.T., Langrock, R., Gerrodette, T., Ballance, L., Chivers, S., Scott, M. and Perrin, W.F. 2017 Incorporating animal movement into distance sampling. [Available at: *http://research_repository.st-andrews.ac.uk*]

Gordon, C., Kujawa, M., Luttrell, J., MacArthur, D., Robinson-Willmott, J. and Thaxter, C. 2013. High-resolution aerial imaging surveys of marine birds, mammals, and turtles on the US Atlantic Outer Continental Shelf — Utility assessment, methodology recommendations, and implementation tools. Final Report by Normandeau Associates to Bureau of Ocean Energy Management. OCS Study BOEM 2013-01130.

Hall, M. 1998. Ecosystem research and tuna fisheries management: Some key questions. *ICCAT Scientific Papers* 50(2):671–72.

Hammond, P. S. and Laake, J.L. 1983. Trends in estimates of abundance of dolphins (*Stenella* spp. and *Delphinus delphis*) involved in the purse-seine fishery for tunas in the eastern Pacific Ocean, 1977–1981. *Rep. int. Whal. Commn* 33:565–88.

Heinemann, D., Gedamke, J., Oleson, E., Barlow, J., Crance, J., Holt, M., Soldevilla, M. and Van Parijs, S. 2016. Report of the Joint Marine Mammal Commission-National Marine Fisheries Service Passive Acoustic Surveying Workshop, 16–17 April 2015, La Jolla, California. NOAA Tech. Memo. NMFS-F/SPO-164. 107pp.

Hodgson, J.C., Baylis, S.M., Mott, R., Herrod, A., Clarke, R.H. 2016. Precision wildlife monitoring using unmanned aerial vehicles. *Sci Rep.* 6: 22574. [Available at: *https://doi.org/10.1038/srep22574* – PMID: 26986721].

Hewitt, R.P. 1985. Reaction of dolphins to a survey vessel: effects on census data. *Fishery Bulletin* 83:187–93.

Holt, R.S. and Powers, J.E. 1982. Abundance estimation of dolphin stocks involved in the eastern tropical Pacific yellowfin tuna fishery determined from aerial, and ship surveys to 1979. NOAA Technical Memorandum, National Marine Fisheries Service, Southwest Fisheries Science Center 23. 95pp.

Hoyle, S.D. and Maunder, M.N. 2004. A Bayesian integrated population dynamics model to analyze data for protected species. *Anim. Biodivers. Conserv.* 27: 247–66.

IATTC, 1999. Inter-American Tropical Tuna Commission Special Report 11. [Available at: *http://www.iattc.org/PDFFiles2/SpecialReports/SpecialReport11.pdf*].

IATTC, 2006. Report of the Inter-American Tropical Tuna Commission Technical Workshop on Calculating Nmin for the Dolphin Stocks of the Eastern Pacific Ocean. Special Report 14. [Available at: *http://www.iattc.org/PDFFiles2/SpecialReports/IATTC-Special-Report-14ENG.pdf*].

IATTC, 2015a. Updated scientific report on the status of dolphin stocks in the eastern Pacific Ocean. [Available at: *http://www.iattc.org/PDFFiles2/Update-of-Sec-of-Commerce-letter-Apr2015.pdf*].

IATTC, 2015b. Inter-American Tropical Tuna Commission Fishery Status Report 13. [Available at: *http://www.iattc.org/FisheryStatusReportsENG.htm*].

IATTC, 2016. Report on the International Dolphin Conservation Program, Inter-American Tropical Tuna Commission, MOP, October 2016. Document MOP-34-05. [Available at: *http://www.iattc.org/Meetings/Meetings2016/Oct/Pdfs/MOP-34-05-Report-on-IDCP.pdf*].

Johnson, K.F., Punt, A.E. and Lennert-Cody, C.E. (Eds.) 2018. Report of the Inter-American Tropical Tuna Commission Workshop on Methods for Monitoring the Status of Eastern Tropical Pacific Ocean Dolphin Populations, 18–20 October 2016, SWFSC, La Jolla, California. Inter-American Tropical Tuna Commission Special Report 22. [Available at: *http://www.iattc.org/PDFFiles/SpecialReports/_English/No-22-2018-JOHNSON%20KELLI%20FAYE,%20Andre%20E.%20Punt%20and%20Cleridy%20E.%20Lennert%20Cody_Report%20of%20the%20workshop%20on%20methods%20for%20monitoring%20the%20status%20of%20EPT%20dolphin%20populations.pdf*].

Joseph, J. 1994. The tuna-dolphin controversy in the eastern Pacific Ocean: Biological, economic, and political impacts. *Ocean Development and International Law* 25:1–30.

Kinzey, D. and Gerrodette, T. 2001. Conversion factors for binocular reticles. *Mar. Mamm. Sci.* 17: 353–61.

Lennert-Cody, C. E., Buckland, S.T. and Marques, F.C. 2001. Trends in dolphin abundance estimated from fisheries data: A cautionary note. *J. Cetacean Res. Manage.* 3(3):305–19.

Lennert-Cody, C.E. and Scott, M.D. 2005. Spotted dolphin evasive response in relation to fishing effort. *Mar. Mamm. Sci.* 21: 13–28.

Lennert-Cody, C.E., Maunder, M.N., Fiedler, P.C., Minami, M., Gerrodette, T., Rusin, J., Minte-Vera, C.V., Scott, M. and Buckland, S.T. 2016. Purse-seine vessels as platforms for monitoring the population status of dolphin species in the eastern tropical Pacific Ocean: the use of fishing vessels as scientific platforms. *Fish. Res.* 178, 101–13.

Lo, N.C.H. and Smith, T.D. 1986. Incidental mortality of dolphins in the eastern tropical Pacific, 1959–1972. *Fish. Bull.* 84:27–34.

Marine Mammal Commission, 2016. Development of use of UASs by the National Marine Fisheries Service for surveying marine mammals. Marine Mammal Commission, Bethesda, MD, USA. 40pp. [Available from *http://www.mmc.gov*]

McNeely, R.L. The purse seine revolution in tuna fishing. *Pacific Fisherman* 59(7):27–58.

NMFS. 2016. Guidelines for preparing stock assessment reports pursuant to Section 117 of the Marine Mammal Protection Act. [Available at: *http://www.nmfs.noaa.gov/pr/sars/pdf/gamms2016.pdf* ].

National Research Council (NRC). 1992. *Dolphins and the Tuna Industry*. National Academy Press, Washington, D.C. 176pp.

Oedekoven, C.S., Buckland, S.T., Marshall, L. and Lennert-Cody, C.E. 2018. Design of a survey for eastern tropical Pacific dolphin stocks. Inter-American Tropical Tuna Commission Document MOP-37-02. [Available at: *http://www.iattc.org/Meetings/Meetings2018/AIDCP-37/PDFs/Docs/_English/MOP-37-02_Design%20of%20a%20survey%20for%20eastern%20tropical%20Pacific%20dolphin%20socks.pdf*]

Oswald, J.N., Rankin, S. and Barlow, J. 2004. The effect of recording and analysis bandwidth on acoustic identification of delphinid species. J. Acoustic Soc. Am. 116: 3178–85.

Oswald, J.N., Rankin, S. and Barlow, J. 2007. A tool for real-time acoustic species identification of delphinid whistles. *J. Acoust. Soc. Am.* 122: 587–95.

Perrin, W.F. 1968. The porpoise and the tuna. *Sea Frontiers* 14:166–74.

Platonov, N.G., Mordvintsev, I.N. and Rozhnov, V.V. 2013. The possibility of using high resolution satellite imagery for detection of marine mammals. *Biol. Bull.* 40: 197–205.

Pollard, J.H., Palka, D. and Buckland, S.T. 2002. Adaptive line transect sampling. *Biometrics* 58: 862–70.

Pryor, K. and Norris, K.S. 1978. The tuna-porpoise problem: behavioral aspects. *Oceanus* 21: 31–7.

Punt, A.E. 2013. Independent review of the eastern Pacific Ocean dolphin population assessment. Inter-American Tropical Tuna Commission Special Report 21. [Available at: *http://www.iattc.org/PDFFiles2/SpecialReports/IATTC-Special-Report-21.pdf* ].

Redfern, J.V., Barlow, J., Balance. L.T., Gerrodette, T. and Becker, E.A. 2008. Absence of scale dependence in dolphin-habitat models for the eastern tropical Pacific Ocean. *Mar. Ecol. Prog. Ser.* 363: 1–14.

Reilly, S.B. 1990. Seasonal changes in distribution and habitat differences among dolphins in the eastern tropical Pacific. *Mar. Ecol. Prog. Ser.* 66: 1–11.

Reilly, S. B., and Barlow, J. 1986. Rates of increase in dolphin population size. *Fish. Bull.* 84: 527–33.

Reilly, S.B., Donahue, M., Gerrodette, T., Forney, K., Wade, P., Balance, L., Forcada, J., Fiedler, P., Dizon, A., Perryman, W., Archer, F. and Edwards, E. 2005. Report of the scientific research program under the International Dolphin Conservation Program Act. NOAA Tech. Memo. NMFS-SWFSC-372. 100pp.

Schwarz, L. K., Gerrodette, T. and Archer, F.I. 2010. Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean. *J. Cetacean Res. Manage*. 11(3):253–65.

Scott, M.D. and Cattanach, K.L. 1998. Diel patterns in aggregations of pelagic dolphins and tunas in the eastern Pacific. *Mar. Mamm. Sci.* 14, 401–28.

Scott, M.D. and Chivers, S.J. 2009. Movements and diving behavior of pelagic spotted dolphins. *Mar. Mamm. Sci.* 25, 137–60.

Silva, G. 1941. Porpoise fishing. *Pacific Fisherman* 39(3):36.

Smith, T.D. 1981. Line-transect techniques for estimating density of porpoise schools. *J. Wild. Manage.* 45:650–7.

Stapleton, S., LaRue, M., Lecomte, N., Atkinson, S., Garshelis, D., Porter, C. and Atwood, T. 2014. Polar bears from space: assessing satellite imagery as a tool to track Arctic wildlife. *PLoS ONE* 9, e101513.

Thaxter, C.B., Ross-Smith, V.H. and Cook, A.S.C.P. 2016. How high do birds fly? A review of current datasets and an appraisal of current methodologies for collecting flight height data: Literature review. Report No. 666 by the British Trust for Ornithology, Thetford, Norfolk IP24 2PU, UK. [Available from *www.bto.org*].

Wade, P.R. 1995. Revised estimates of incidental of dolphins (Delphinidae) by the purse-seine tuna fishery in the eastern tropical Pacific, 1959–1972. *Fish. Bull.* 93:345–54.

Wade, P.R., Watters, G.M., Gerrodette, T. and Reilly, S.B. 2007. Depletion of spotted and spinner dolphins in the eastern tropical Pacific: modelling hypotheses for their lack of recovery. *Mar. Ecol. Prog. Ser.* 343: 1–14.

Webb, A., Irwin, C. and Elgie, M. 2015. Kincardine Offshore Wind Farm: Final report on aerial surveys from April 2013 to September 2014. Report by HiDef Aerial Surveying Limited to Pilot Offshore Wind Limited. [Available at: *http://pilot-renewables.com/pdf_docs/KOWL_ES_AppendixB_FinalAerialSurveysReport_Issued.pdf*].

Weiß, F., Büttger, H., Baer, J., Welcker, J. and Nehls, G. 2016. Erfassung von Seevögeln und Meeressäugetieren mit dem HiDef Kamerasystem aus der Luft. *Seevögel* 37: 14–21. [In German]

Williams, K.A., Connelly, E.E., Johnson, S.M. and Stenhouse, I.J. (Eds) 2015. *Baseline Wildlife Studies in Atlantic Waters Offshore of Maryland: Final Report to the Maryland Department of Natural Resources and the Maryland Energy Administration.* Report BRI 2015–17, Biodiversity Research Institute, Portland, Maine. 437pp.

Williamson, L.D., Brookes, K.L., Scott, B.E., Graham, I.M., Bradbury, G., Hammond, P.S. and Thompson, P.M. 2016. Echolocation detections and digital video surveys provide reliable estimates of the relative density of harbour porpoises. *Methods in Ecology and Evolution* 7: 762–9.

Yuan, Y., Bachl, F.E., Lindgren, F., Borchers, D. L., Illian, J. B., Buckland, S.T., Rue, H. and Gerrodette, T. 2017. Point process models for spatio-temporal distance sampling data from a large-scale survey of blue whales. *Annals of Applied Statistics* 11: 2270–97.