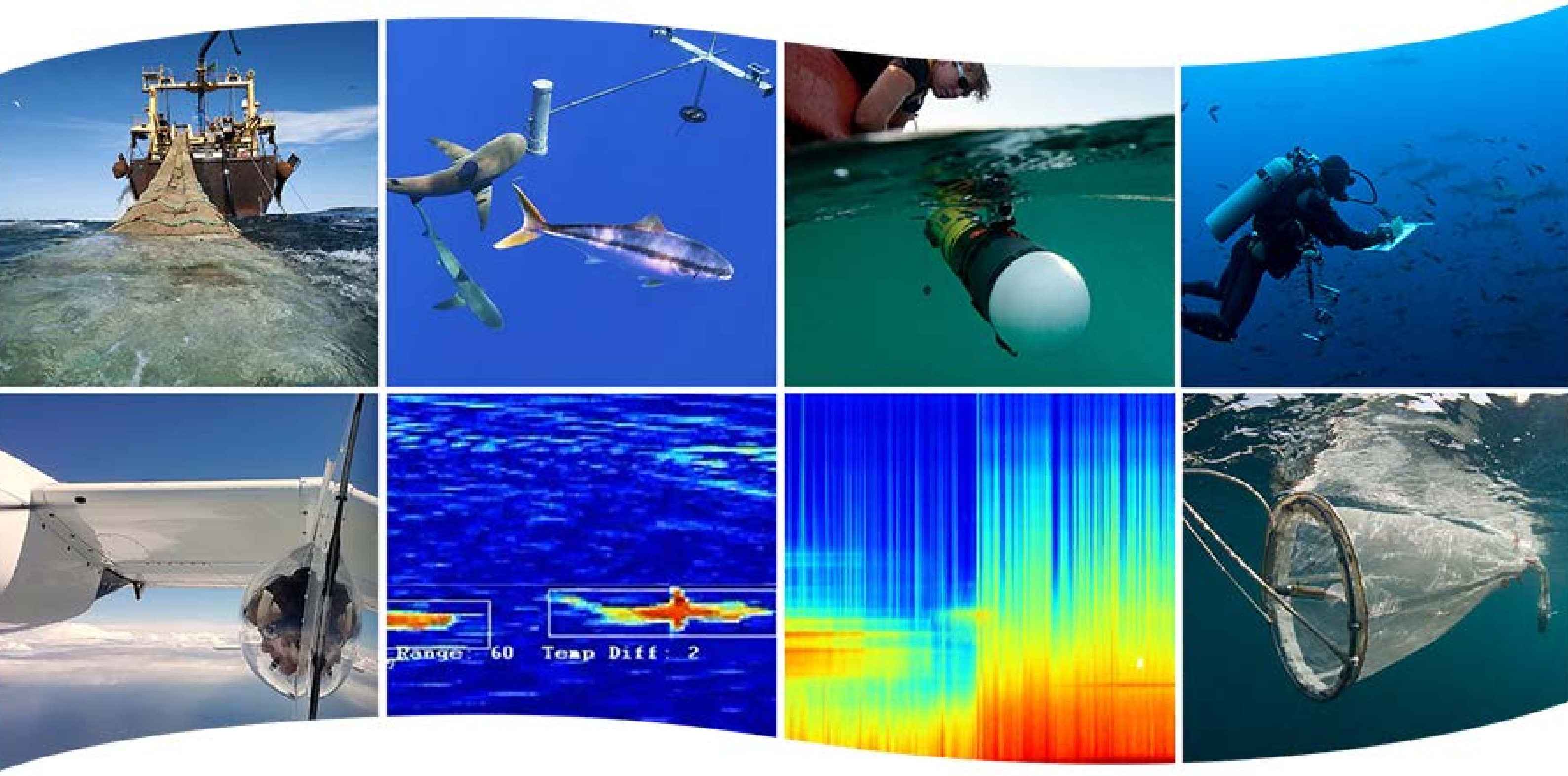
Final report

Comparative assessment of pelagic sampling methods used in marine monitoring

Phil Bouchet, Claire Phillips, Zhi Huang,
Jessica Meeuwig, Scott Foster, Rachel Przeslawski



Marine Biodiversity Hub
National Environmental Science Programme

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Geoscience Australia
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National Environmental Science Programme

Project D2: Analysis methods and software to support Standard Operating Procedures for survey design, condition assessment and trend detection.

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Important Disclaimer

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Acronyms

AMP	Australian Marine Park
AODN	Australian Ocean Data Network
ARGOS	Advanced Research and Global Observation Satellite
AUV	Autonomous Underwater Vehicle
BRUV	Baited Remote Underwater Video
CPR	Continuous Plankton Recorder
CPUE	Catch-Per-Unit-Effort
CSIRO	Commonwealth Scientific and Industrial Research Organisation
DOV	Diver-Operated Video
EBV	Essential Biodiversity Variable
eDNA	Environmental DNA (Deoxyribonucleic Acid)
EEZ	Exclusive Economic Zone
EOV	Essential Ocean Variable
FAD	Fish Aggregating Device
GBR	Great Barrier Reef
GPS	Global Positioning System
IMOS	Integrated Marine Observing System
LiDAR	Light Detection and Ranging
MCA	Marine Commonwealth Area
MPA	Marine Protected Area
OAWRS	Ocean Acoustics Waveguide Remote Sensing
OPC	Optical Plankton Counter
Radar	Radio Detection and Ranging
ROV	Remotely Operated Vehicle
SLA	Sea Level Anomaly
SST	Sea Surface Temperature
UAV	Unmanned Aerial Vehicle
USV	Unmanned Surface Vehicle
UVC	Underwater Visual Census
VPR	Video Plankton Recorder

Executive summary

Australia's Exclusive Economic Zone (EEZ) is the third largest maritime territory in the world. Monitoring its dynamics is fundamental to understanding and reporting on how the ocean is responding to human pressures and global environmental change. Increasingly stringent conservation budgets, however, are placing a strong emphasis on strategic resource allocation. Faced with mounting pressures to build accountability, managers and policy advisors must now more than ever make monitoring investment decisions that are both impactful and cost-effective. This can be challenging given the smorgasbord of modern survey tools currently available, most of which differ widely in costs, capabilities, mobilisation constraints, resolution, or sensitivity, and are evolving rapidly without always being critically evaluated or compared.

Whilst pelagic waters present fascinating opportunities for ecological investigation, their extreme horizontal, vertical and temporal patchiness, as well as the huge size range of organisms inhabiting the open ocean, also pose important methodological challenges for sampling. Early pelagic studies relied heavily on capture sampling using nets. While these remain a critical component of biological and oceanographic research today, a rapidly increasing array of innovative technologies (e.g. drifting baited videography, environmental DNA, unmanned aerial vehicles) with various degrees of autonomy and sensory capabilities is revolutionising the way we quantify biophysical processes and observe wildlife in remote habitats. Protocols for choosing optimal combinations of methods for a given region, taxonomic/indicator group, or habitat remain generally unavailable. There is thus an urgent need to synthesise and compare these methods to determine how they can best support and strengthen the empirical evidence base available for implementing marine monitoring programmes.

The aim of the present report is to provide a comparative assessment of commonly used pelagic sampling methods. We do this by undertaking a qualitative, yet comprehensive, review of the published literature to identify their potential advantages, limitations, and their relevance to monitoring efforts.

The document is divided into four main sections:

- **Section 1** offers contextual background information, and details the objectives and scope of the report.
- **Section 2** provides a succinct overview of pelagic monitoring, and describes the strengths and weaknesses of nearly 50 biological and oceanographic sampling methods. These include (i) capture methods (active and passive, e.g. pelagic trawls, pelagic longlines, light traps); (ii) hydroacoustic methods (active and passive, e.g. echosounders, passive acoustic recorders); (iii) visual, optical and thermographic methods (underwater, airborne, vessel-based, and land-based, e.g. baited remote underwater video systems, aerial visual surveys, theodolites); (iv) ocean robotics (manned and unmanned, e.g. submersibles, gliders, remotely operated vehicles); (v) satellite technologies (remote sensing, satellite photography, biotelemetry); (vi) Genomics (e.g. environmental DNA); and (vii) participatory methods (e.g. citizen science).

- **Section 3** presents the results of an online questionnaire delivered to researchers and marine practitioners across the globe, and designed to gauge general patterns of use of various pelagic sampling methods, as well as underlying drivers of method selection. Sixty-two individuals from 16 countries responded to the survey, reporting vessel-based visual surveys (39%), underwater visual census (31%), environmental DNA (31%), plankton nets (21%), or active acoustics (18%) as being among their preferred tools.
- **Section 4** gives a summary of selected published studies combining two or more pelagic sampling methods. The focus is on identifying commonalities or discrepancies in the ecological relationships inferred from multiple gear types.
- **Section 5** relates links each sampling method to its capability to measure global indicators (e.g. Essential Ocean Variables, Essential Biodiversity Variables), and provides further advice on choosing appropriate methods relative to specific monitoring objectives, target environments, and available resources.

A 'silver-bullet' approach to pelagic monitoring likely does not exist, nor is necessarily feasible. Instead, this comparative assessment provides a blueprint for guiding sampling activities in the context of pelagic monitoring efforts. Such information is essential to promoting transparency, repeatability, and standardisation across studies and institutions, so that method selection aligns with study objectives, with a clear understanding of benefits and limitations. Ultimately, robust survey designs and standard operating procedures are key factors underlying data comparability over time and space.

1. Introduction

1.1 Background

Most space on Earth is ocean. With an extent greater than 70% of the planet's surface and a volume exceeding a billion cubic kilometres (Charette & Smith 2010), the pelagic environment constitutes by far the largest of all biomes (Angel 1993). As a carbon sink and an exchanger of heat, gases, particles, and momentum with the atmosphere, it is a vital component of the climate and weather system that controls the Earth's energy budget, temperature balance, and hydrological cycle (Bigg *et al.* 2003). Importantly, it is also a vast reservoir of life, with diverse biological communities that likely outnumber most others (Robison 2004, Robison 2009), and that provide globally significant resources, services and natural capital essential to the welfare of mankind (Worm *et al.* 2006, Cardinale *et al.* 2012).

Despite their importance, pelagic waters remain sparsely explored and drastically data-deficient (Webb *et al.* 2010, Handegard *et al.* 2013, DeVaney 2016). Australia, for instance, boasts the third biggest ocean territory in the world, yet knowledge of its biodiversity values and processes is largely incomplete (Butler *et al.* 2010, Hedge 2016). Monitoring activities are fundamental to bridging these knowledge gaps, as they can generate the data necessary to assess and document trends in environmental assets in response to both human pressures and spatial management measures, ultimately allowing an ecosystem-based understanding of the country's marine estate (Hayes *et al.* 2015b).

Increasingly modest conservation budgets, however, are placing a strong emphasis on strategic resource allocation (McDonald-Madden *et al.* 2008). Faced with mounting pressures to build accountability, managers and policy advisors must now more than ever make monitoring investment decisions that are both impactful and cost-effective (McDonald-Madden *et al.* 2010). This can be challenging given the smorgasbord of modern survey tools currently available, most of which differ widely in costs, capabilities, mobilisation constraints, resolution, or sensitivity, and are evolving rapidly without always being critically evaluated or compared. In recent years, novel technologies for sampling pelagic organisms and/or habitats such as drifting videography (Bouchet & Meeuwig 2015, Bouchet *et al.* 2018b), environmental DNA (Thomsen & Willerslev 2015b), or unmanned (airborne or waterborne) vehicles (Linchant *et al.* 2015) - among many others - have emerged and are gaining traction. These techniques can supplement or even replace more traditional approaches, including midwater trawling (Sutton *et al.* 2013), visual transects (Hammond *et al.* 2013, Roberts *et al.* 2016, Bannister 2017), passive and active acoustics (Benoit-Bird & Lawson 2016), electronic telemetry (Hobday *et al.* 2009, Sims *et al.* 2009, Costa *et al.* 2010), and remote sensing (Platt & Sathyendranath 2008, Kachelriess *et al.* 2014). However, protocols for choosing optimal combinations of methods for a given region, taxonomic/ indicator group, or habitat remain generally unavailable. Additionally, the few published studies that weigh up the merits and caveats of multiple sampling gears typically do not report explicit cost estimates, thereby undermining their potential to match research and management needs (Yoklavich *et al.* 2015).

1.2 Objectives

Australia has the capacity to deploy a wide range of marine monitoring equipment, embracing new technologies as they become available, validated and practicable (Hedge 2016). The purpose of this document is to provide a critical appraisal of the suitability of various pelagic sampling methods (see **Table 1.1** for a full list) for supporting the long-term monitoring of the national network of Australian Marine Parks (AMPs). Such analysis will assist marine scientists, managers and policy agencies in choosing the most appropriate, robust, and cost-effective method for collecting empirical data on biodiversity status and trends within the Commonwealth marine estate (Katsanevakis *et al.* 2012). This work forms an output from the National Environmental Science Programme (NESP) Marine Biodiversity Hub expanded Project D2 (https://www.nespmarine.edu.au/project/project-d2-analysis-methods-and-software-support-standard-operating-procedures-survey-design), and is complemented by a similar report focused on benthic and demersal sampling (Przeslawski *et al.* 2018).

1.3 Scope

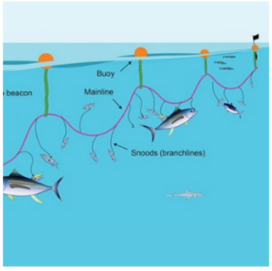
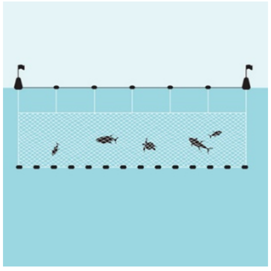
This document relates to marine surveying gear designed to acquire empirical biological data on the processes taking place in, and organisms living within, the ocean's water column, at scales ranging from micro (e.g. Billings *et al.* 2017) to macro (e.g. Bouchet & Meeuwig 2015). The report encompasses a large number of approaches to pelagic sampling, including both 'platforms' and 'sensors' (sensu Bean *et al.* 2017), as well as their combinations. These are referred to as 'methods' throughout.

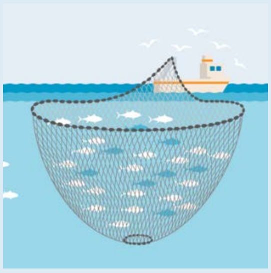
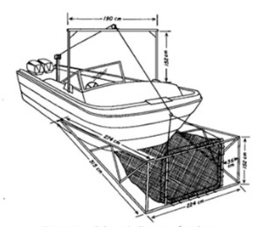
A number of reviews have assessed subsets of pelagic sampling methods (e.g. eDNA, remote sensing, baited videos, underwater visual census, tagging and telemetry, etc.), although somewhat in isolation (e.g. Hofmann & Gaines 2008, Katsanevakis *et al.* 2012, Duffy *et al.* 2013, Mallet & Pelletier 2014, Maxwell *et al.* 2014, Caldwell *et al.* 2016, Bean *et al.* 2017, Letessier *et al.* 2017, Paris *et al.* 2018). The purpose of this report is to build upon these and assess the merits and drawbacks of a large spectrum of approaches. Results will indicate whether broad-scale biodiversity patterns might prove consistent among different datasets, and indicate which combination of sampling gears may be the most robust for a given biological/oceanographic survey. The information contained herein, together with standard operating procedures (for example those in Przeslawski & Foster 2018), will contribute to the development of consistent, transparent, and harmonised strategies for surveying pelagic habitats in support of the monitoring of AMPs and other Australian waters.

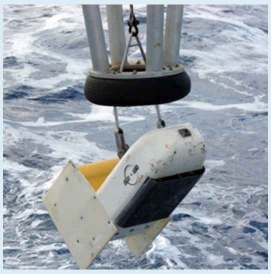
The report is divided into four further sections:

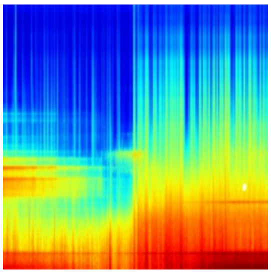
- **Section 2** presents an overview of the merits and limitations of each method, with notes on their relevance to monitoring.
- **Section 3** describes the use and perceptions of marine pelagic sampling methods via results from an online questionnaire.
- **Section 4** reviews comparative studies where multiple methods have been used simultaneously, drawing insights as to their relative performance.
- **Section 5** relates the above to marine monitoring.

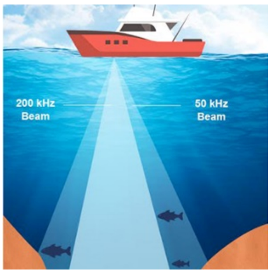
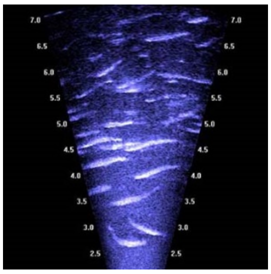
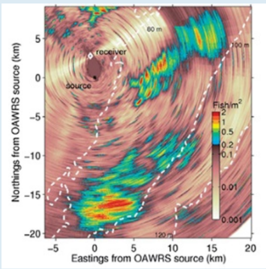
Table 1.1 List of pelagic sampling methods considered in the comparative assessment.

Capture methods			
Passive			
	**Pelagic longline** Line deployed near the surface and fitted with baited hooks attached at regular intervals.		**Pelagic (drift) gillnet** Vertical panel of fine-filament net kept at or near the surface by numerous floats and weights and moored in place by anchors.
	**Fish aggregating device (FAD)** Manmade structure built from any material (e.g. buoy, floats, driftwood) and used to attract fish.		**Light trap** Illuminated trapping device used for attracting and catching pelagic fish larvae and juveniles.

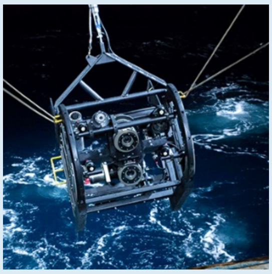
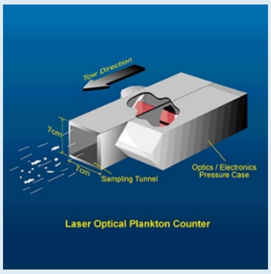
Capture methods			
Active			
	**Pelagic trawl** Net towed behind a vessel through the water column. Also known as midwater trawl.		**Pelagic purse seine** Large net used to surround a shoal of fish, with the bottom drawn together by a 'purse line' to prevent the catch escaping.
	**Plankton net** Modified trawl nets, usually funnel-shaped, designed to capture live plankton of nearly any size with vertical or horizontal tows.	 FIGURE 1.—Schematic diagram of pushnet.	**Pushnet (e.g. bow-mounted)** Scoop net with a rigid frame pushed before a vessel.

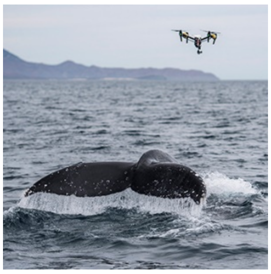
Capture methods			
Active			
	**Neuston net** Type of plankton net designed to sample neuston, i.e. minute organisms found in the surface film of the ocean's water column.		**Continuous Plankton Recorder (CPR)** Instrument designed to capture plankton samples over huge stretches of ocean.
	**Spearfishing** Fishing conducted whilst free-diving, scuba diving or snorkelling, and using powered spear guns to strike the hunted fish.		

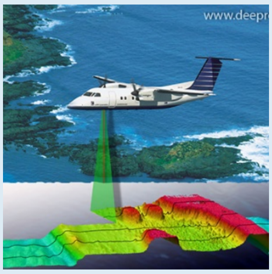
Hydroacoustic methods			
Passive			
	**Acoustic telemetry** Small sound-emitting devices that permit the remote tracking of animals in three dimensions.		**Passive acoustics recorder** Electronic recording device that acquires and stores acoustic data internally.

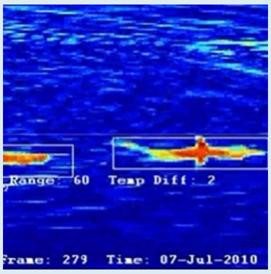
Hydroacoustic methods			
Active			
	**Active acoustics (e.g. sonar, echosounder)** Instruments sending and interpreting acoustic pulses to detect objects within the water column.		**Acoustic camera** Imaging device used to locate sound sources and interpret their signals into a visual (2D image) representation.
	**Ocean acoustics waveguide remote sensing (OAWRS)** Acoustic system capable of imaging organisms over scales > 100,000 km^2		

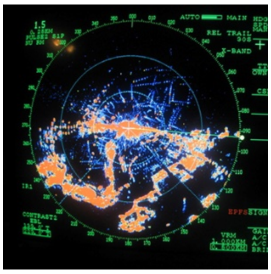
Visual, optical & thermographic methods			
Underwater			
	**Underwater visual census (UVC)** Visual transect (count) conducted by an observer equipped with SCUBA gear.		**Diver operated video (DOV)** Imagery transect conducted by an observer equipped with SCUBA gear and one or more cameras.
	**Pelagic BRUVs** Underwater video camera system deployed remotely at fixed sites and usually baited.		**Drop (drift) camera** Ballasted video camera system encased in a high pressure-rated housing and typically sunk through the water column to large depths.

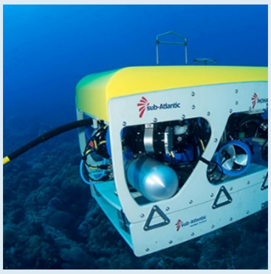
Visual, optical & thermographic methods			
Underwater			
	**Midwater towed video** Underwater camera system towed behind a moving vessel at a fixed speed and pre-determined depth.		**Optical plankton counter (OPC)** Instrument that detects, sizes, and counts particles based on their attenuation of a light beam.
	**Video plankton recorder (VPR)** Towed underwater video microscope that photographs particles and plankton in real time.		

Visual, optical & thermographic methods			
Airborne			
	**Aerial visual survey** Visual transect(s) searched by observers onboard an aircraft travelling at a chosen altitude.		**Aerial digital video** Video camera system making digital recordings of transects covered by an aircraft at a chosen altitude.
	**Aerial digital still** Hull-mounted camera taking photographic images of an area at regular intervals during a flight along pre-determined transects.		**Unmanned aerial vehicle (UAV)** Small aircraft (drone) piloted by remote control or onboard computers.

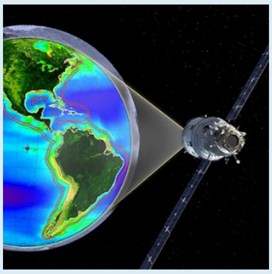
Visual, optical & thermographic methods			
Airborne			
	**Light detection and ranging (LiDAR)** Instrument that measures distances to targets by illuminating them with a pulsed laser light.		

Visual, optical & thermographic methods			
Shipboard			
	**Shipboard digital video** Video camera system making digital recordings of transects covered by a vessel at a chosen speed.		**Shipboard digital stills** Camera system taking photographic images of an area at regular intervals during vessel-based transects.
	**Shipboard visual survey** Visual transects searched by observers onboard a marine vessel travelling at a fixed speed.		**Infrared imaging** Thermographic camera device that renders infrared radiation as visible light.

Visual, optical & thermographic methods			
Land-based			
	**Theodolite** Instrument with a rotating telescope that measures horizontal and vertical angles to targets, allowing them to be tracked in space.		**Radar** System for detecting remote objects (presence, direction, speed) by sending out pulses of radio waves.

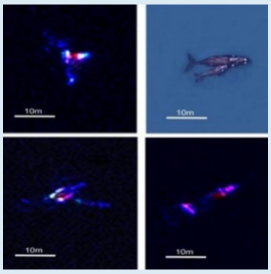
Marine robotics			
Manned			
	**Manned submersible** Crewed ship capable of submerging and operating underwater		
Unmanned			
	**Autonomous underwater vehicle (AUV)** Robot that travels underwater without requiring input from an operator.		**Remotely operated vehicle (ROV)** Unoccupied robot connected to a ship by cables that allow its remote operation and navigation.
	**ARGO float** Free-drifting profiling float that measures the temperature and salinity of the upper 2000 m of the ocean.		**Ocean glider** Autonomous winged underwater vehicle that collects ocean data using buoyancy-based propulsion.
	**Unmanned surface vehicle (USV)** Vehicle that operates on the surface of the water (watercraft) without a crew.		

Satellite technologies



Satellite remote sensing

Satellite-fitted sensors that gather data remotely by detecting the energy reflected from the Earth.



Satellite photography

Remote camera systems mounted on satellites, capable of taking photographs from space at high resolution.

Satellite telemetry

Orbiting satellites that detect and relay signals emitted from positioning transmitters attached to animals.

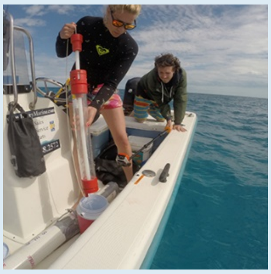
Archival data logger

Electronic device that records data over time either with built-in or external instruments and sensors.

Genomics

Biopsy

Survey technique involving the collection of a small sample of biological material (e.g. skin, muscle, fatty tissue).



Environmental DNA (eDNA)

Survey technique involving the collection of DNA fragments sourced from seawater samples.

Participatory methods

Citizen science

Engaged members of the public who contribute to science by sharing opportunistic observations of wildlife using a variety of methods.

2. Monitoring pelagic seascapes: The ‘why’, the ‘what’, and the ‘how’

Monitoring can be defined as the systematic acquisition of knowledge over time (Gerber *et al.* 2005). It typically involves the collection of empirical information about one or more indicator variables (e.g. the abundance of a threatened species, the condition and extent of important habitats) that can be interpreted to assess the state of ecosystems and draw inferences about their rates of change (Yoccoz *et al.* 2001, Jones *et al.* 2013). The importance of monitoring wildlife populations is universally acknowledged (Jones *et al.* 2013), with monitoring activities forming a central tenet of most conservation programmes (Marsh & Trenham 2008) that leverage investments in excess of 10% of the multimillion dollar budgets available to biodiversity agencies in both Australia and the United States (McDonald-Madden *et al.* 2010). Over the last decades, marine monitoring has diversified into a complex array of initiatives that operate on scales ranging from local to continental (Agnew 1997, McDonald-Madden *et al.* 2010, Borja *et al.* 2016, Bean *et al.* 2017).

2.1 Motivations for monitoring

Few monitoring programmes provide explicit statements of their underlying objectives, beyond the simple premise that additional information about a system will prove inherently useful (Yoccoz *et al.* 2001). In practice, population monitoring is routinely perceived as a rational and defensible activity in the pursuit of improved conservation outcomes (McDonald-Madden *et al.* 2010), and monitoring efforts are thus frequently conducted in response to species showing some degree of conservation concern (Marsh & Trenham 2008). In this context, monitoring plays a particularly important role in detecting early warning signals of population declines, and allowing the likely drivers of these declines to be inferred (Santini *et al.* 2017). Such knowledge is often necessary to trigger a given management action, or aid in selecting between competing management options (Gerber *et al.* 2005). Monitoring efforts can therefore provide a direct mechanism for both guiding and auditing management policies (Yoccoz *et al.* 2001), particularly in support of adaptive frameworks for state-dependent decision-making (Hauser *et al.* 2006, Jones *et al.* 2013, Brown & Williams 2015). There are, of course, additional reasons to monitor (**Figure 2.1**), including legislative obligations, public engagement, education and awareness raising, or the hope for serendipitous discoveries (McDonald-Madden *et al.* 2010, Possingham *et al.* 2012).

In an era where ocean life is rapidly becoming ‘under siege’ (Coll *et al.* 2012), with pervasive species losses occurring as a consequence of human activities (Butchart *et al.* 2010, Stokstad 2010, Selig *et al.* 2014, McCauley *et al.* 2015), much marine monitoring has focused on evaluating and understanding the effectiveness of marine protected areas (MPAs) (Pomeroy *et al.* 2005). By promoting resilience to overfishing (Aburto-Oropeza *et al.* 2011, García-Rubies *et al.* 2013), disease outbreaks (Mellin *et al.* 2016), global environmental change (Micheli *et al.* 2012), and natural perturbations (Olds *et al.* 2014), MPAs can often provide a buffer against biodiversity erosion (Sala & Giakoumi 2017). They have thus recently proliferated, particularly in offshore waters (Singleton & Roberts 2014).

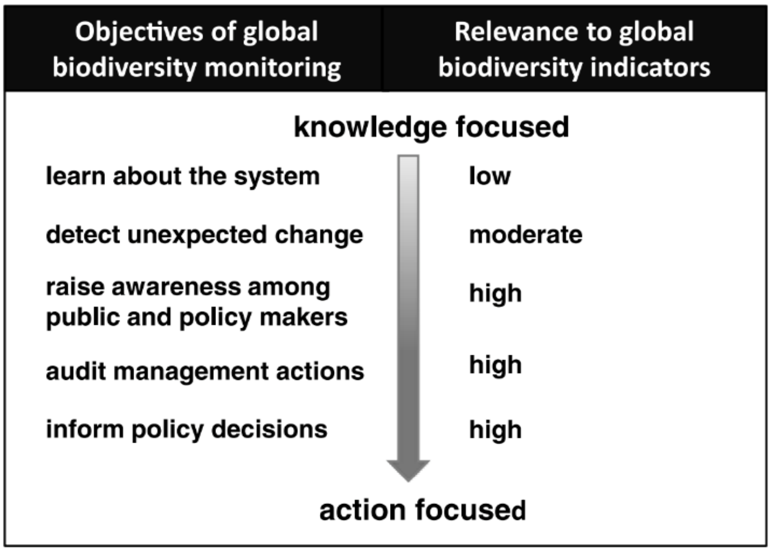


Figure 2.1 Motivations for monitoring biodiversity. Figure reproduced with permission from Wiley (Source: Jones *et al.* 2011).

Despite their high profile and increasing geographic coverage (Letessier *et al.* 2017), contention remains about the capacity of MPAs to yield optimal and consistent conservation benefits (Klein *et al.* 2015, Davies *et al.* 2017, O'Leary *et al.* 2018). Differences in MPA performance stem in part from shortfalls in staffing and equitable access to infrastructure (Gill *et al.* 2017, Worm 2017), but also from fundamental limitations in our understanding of biological patterns throughout deep pelagic environments (Fraschetti *et al.* 2002, Leathwick *et al.* 2008, Webb *et al.* 2010). Though evidence exists that MPAs can be successfully implemented in the absence of perfect ecological knowledge, evaluations of MPA efficacy constitute a necessary and promising approach to improving science-driven policy-making (Fox *et al.* 2012). Such evaluations cannot, however, be considered without rigorous and comprehensive monitoring data (Edgar 2011, Bourlat *et al.* 2013).

Monitoring is notoriously costly and labour-intensive (Borja & Elliott 2013), and in the wake of a global economic crisis, every penny spent on monitoring is potentially a penny not allocated to on-the-ground action. As a result, an expanding body of literature discusses 'value-of-information' analysis, namely the trade-off between spending limited funds on direct management action, and gaining new information in an attempt to improve management performance in the future (Grantham *et al.* 2008, Underwood *et al.* 2008, Hermoso *et al.* 2013, Runting *et al.* 2013, Canessa *et al.* 2015, Hermoso *et al.* 2015, Maxwell *et al.* 2015, Williams & Johnson 2015). The value of monitoring inevitably rises with increasing uncertainty around available prior information (e.g. previous abundance estimate) (Hauser *et al.* 2006). Monitoring efforts should accordingly be driven by proximate information needs (Proença et al. 2016), such that good monitoring must rest on a clear rationale for gathering data in the first place (McDonald-Madden et al. 2010). To be most useful, monitoring should also be sustained through time, although this is often made difficult by a lack of stable, continuous and long-term funding and staffing schemes.

2.2 Defining monitoring variables

Protracted (and usually unsettled) arguments over exactly what to monitor remain a key obstacle to efficient monitoring worldwide (Pereira *et al.* 2013). ‘Laundry list’ approaches (i.e. where practically everything is monitored) are often discouraged as they lack focus and tractability, wrongly favour quantity over quality, and ultimately disregard the day-to-day realities of operating and financing credible monitoring programmes (Lindenmayer & Likens 2010). Accordingly, there has been growing interest in the targeting of ‘indicator’ species (Carignan & Villard 2002). For instance Lindenmayer & Likens (2010) reported that over 55 major taxonomic groups, from viruses to virtually all higher vertebrates, have been used in this capacity (Siddig *et al.* 2016). Surrogate relationships, however, are not always well resolved, making it a challenge to appraise the motivation behind a given choice of indicators (Lindenmayer & Likens 2009). Because charismatic species typically receive greater attention (Jones *et al.* 2013), taxonomic bias is also a concern, as it will limit relevance other organisms and life stages (Bourlat *et al.* 2013). Critically, the data necessary to document early warning signals of ecosystem change remain largely scattered and scant. Most indicators are hence only narrowly applicable and lack robustness due to incomplete spatial, temporal and taxonomic coverage (Schmeller *et al.* 2017b).

In response to this and the need to regularly deliver consistent information on the state of the world’s natural communities, the Group on Earth Observations Biodiversity Observation Network (GEOBON, http://geobon.org/) developed the conceptual Essential Biodiversity Variables (EBVs) framework (Pereira *et al.* 2013). EBVs represent a unifying set of complementary metrics that allow a multidimensional and integrated view of trends in biodiversity over time, from fine to large scales (Proença *et al.* 2016). They act as an intermediate layer between raw biological observations and derived indicators (Brummitt *et al.* 2017, Schmeller *et al.* 2017a), and were conceived to help the strategic prioritisation and harmonisation of monitoring initiatives. Despite the emergence of the Global Ocean Observing System’s (GOOS, http://www.goosocean.org/) Essential Ocean Variables (EOVs, **Figure 2.2**) (Fischer & Grimes 2012, Constable *et al.* 2016, Miloslavich *et al.* 2018), defining which parameters to measure in the sea remains generally more complex than on land as marine (pelagic) systems exhibit much higher heterogeneity and patchiness (Kavanaugh *et al.* 2016). As Hayes *et al.* (2015a) point out: “We find ourselves in a catch-22: we don’t understand marine ecosystems sufficiently well to know what we must measure, yet without appropriate long-term measurements our chances of improving our understanding are small”. Equally important is the recognition that different stakeholders are prone to holding contrasting views on the suite of candidate variables that ought to be operationalized (Schmeller *et al.* 2017a). Rather than ‘what should be monitored?’, a more germane philosophy may therefore be to ask ‘what is the crucial question?’ or ‘why do we want to monitor it?’ (Lindenmayer & Likens 2009, Hayes *et al.* 2015a).

In general, the monitoring of MPAs has conventionally revolved around rejecting the null hypothesis that closures (spatial and/or temporal) have no impact on population variables (e.g. density, biomass), community variables (e.g. composition, structure), or fisheries variables (e.g. catch per unit effort) for harvested species (Gerber *et al.* 2005). A key underlying assumption, in this context, is that a positive ratio (i.e. inside vs. outside, or after vs. before) will be a reflection of MPA success (Claudet *et al.* 2008, Lester *et al.* 2009, Moffitt *et al.* 2013).

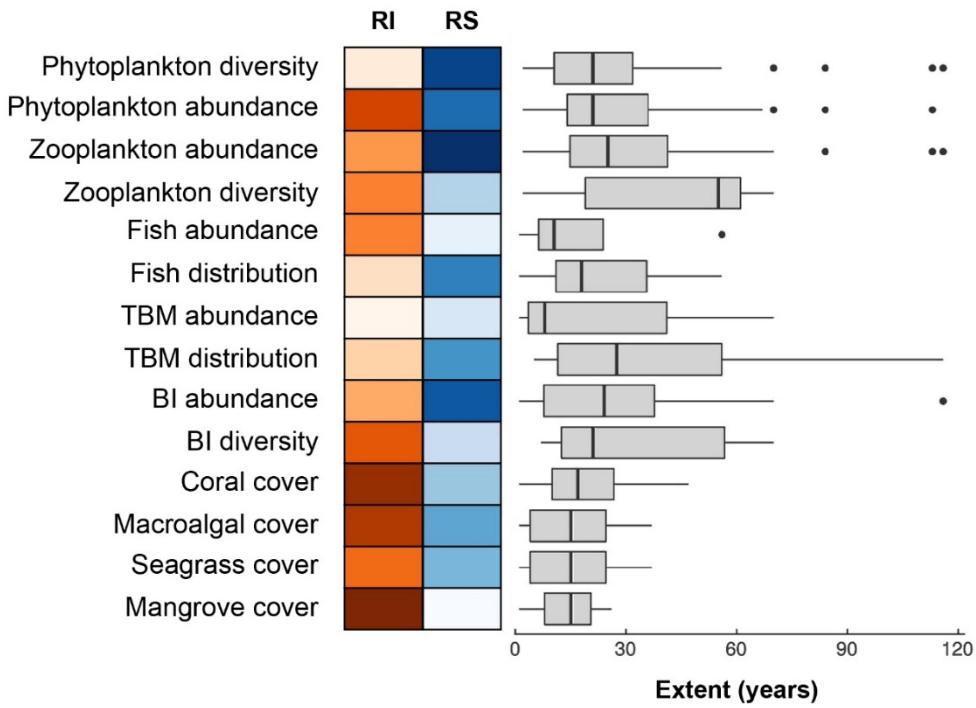


Figure 2.2 List of biological Essential Ocean Variables (EOVs) and their relative impact (RI) and relative scalability (RS). Relative impact scores are calculated based on the perceived capacity of each variable to address a range of societal drivers and pressures. Relative scalability scores consider the spatial cover and temporal extent of available observations, with highest feasibility for expansion to global coverage. Each is shown using a colour scale that ranges from low (light) to high (dark) values. Extent: Temporal extent of 104 observing programmes measuring each variable. TBM: Sea turtles, seabirds and marine mammals. BI: Benthic invertebrates. Figure adapted from Miloslavich *et al.* (2018) under a Creative Commons Attribution 4.0 International license (CC BY 4.0).

Monitoring activities also allow performance to be tracked through time, such that baseline surveys conducted upon MPA designation can provide a snapshot of ecological communities and oceanographic processes that may be used as a benchmark for detecting future changes, particularly over the longer term (Barrett *et al.* 2007). The nature and type of information being collected for monitoring is, therefore, a crucial aspect to consider (Gerber *et al.* 2005). As a rule, data must be systematic, scalable, and taxonomically representative, so that they can be meaningfully compared, for instance across sites or years (Proença *et al.* 2016). While socio-economic factors have received increasing attention in recent MPA studies (e.g. Rodríguez-Rodríguez *et al.* 2015), the majority of monitoring efforts continue to be based on physical and biological metrics only. Species abundance is one example of the latter, and has great appeal because fluctuations in numbers of organisms may signal the decline or recovery of a threatened species, or the spread of an invasive one. Abundance is also arguably one of the most available types of monitoring data (Marsh & Trenham 2008), and has accordingly been proposed as a priority EBV (Schmeller *et al.* 2017b). Variability in abundance, however, can easily be confounded by both environmental and demographic stochasticity (e.g. extreme mortality events, Frederiksen *et al.* 2008) and biotic interactions (e.g. competitive release, Barley *et al.* 2017), making abundance measures insufficient on their own (Santini *et al.* 2017).

Richness-based indices are also relatively common state variables in biodiversity monitoring programmes (Yoccoz *et al.* 2001), and have the advantage of only necessitating presence data, which often prove easier and more cost-efficient to gather. As a summation of counts, however, species richness forgoes information on species identity (e.g. endemic and invasive species cannot be differentiated), and only responds to local extinctions, colonisations, and migrations. It may, therefore, be inadequate or misleading as an indicator of biodiversity change if considered without additional information (Santini *et al.* 2017), particularly where preferential sampling is likely to result in unknown biases in species detectability.

Clearly, no single metric can provide a full picture of all relevant facets of biodiversity, and understanding the sensitivity of alternative metrics to change is crucial to aligning monitoring protocols with both management needs and the main objectives of a given study (Santini *et al.* 2017). Additional complications arise within MPA networks, as biodiversity patterns in one reserve may be affected by processes of larval dispersal and adult fish movement between it and other nearby spatial closures (White *et al.* 2011).

2.3 Approaches to pelagic monitoring

Some of the difficulties in measuring MPA effectiveness should be overcome by the use of rigorous sampling methods and designs that pay heed to issues of scale (**Figure 2.3**), power, sampling frequency, and control site selection (Fraschetti *et al.* 2002). Foster *et al.* (2018) give a comprehensive overview of key several statistical considerations for marine monitoring, and we thus do not delve into these here. Rather, we focus on describing the array of sampling approaches currently available (**Table 1.1**), and their main benefits/limitations for monitoring pelagic waters. Topical reviews are listed at the end of each individual section (where appropriate), as a source of additional information to which readers can refer.

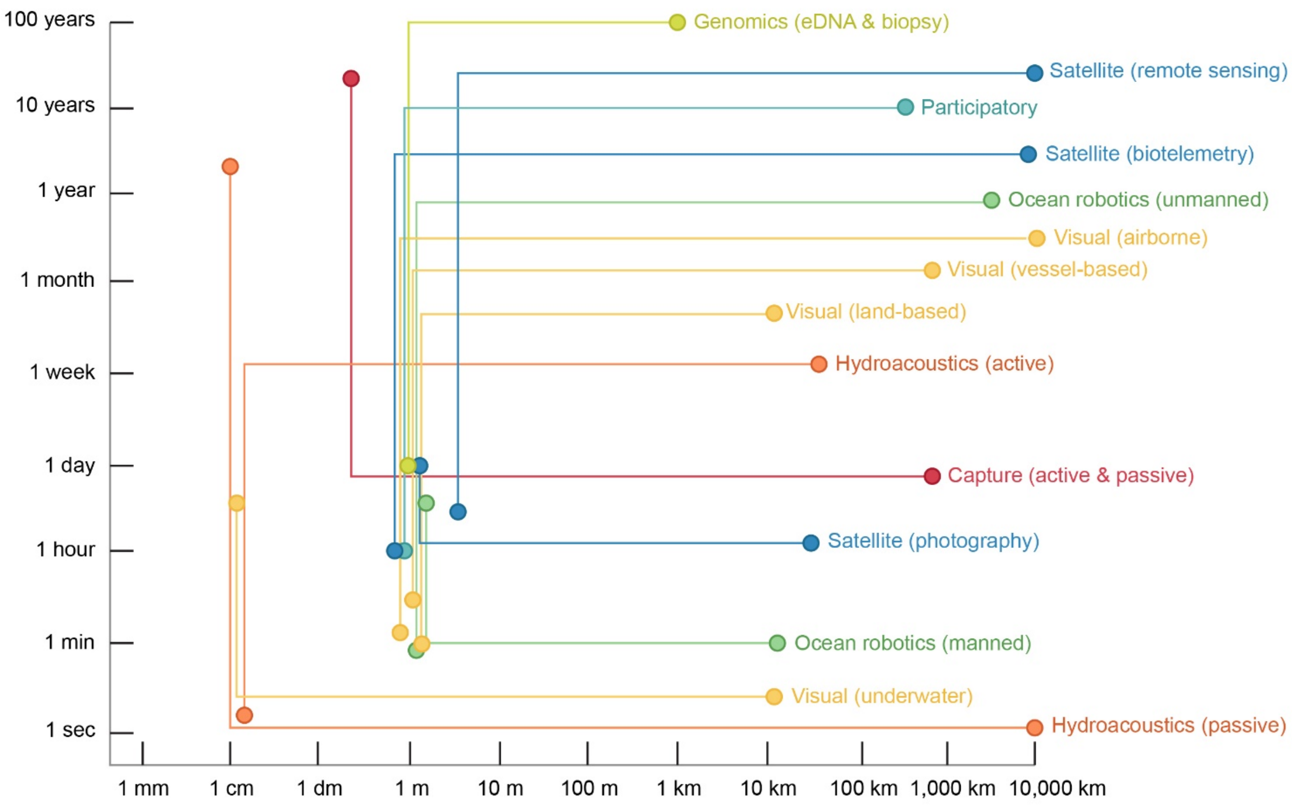


Figure 2.3 Time-space diagram showing the approximate temporal and spatial scales of pelagic sampling methods used in marine monitoring. Methods are colour-coded by class, as per **Section 2.3**.

2.3.1 Capture methods

For decades, traditional approaches to monitoring marine populations revolved around various forms of capture sampling (Jech *et al.* 2009). Capture methods return biological specimens (live or dead) and comprise a wide range of gears, including trawls, longlines, gillnets, purse seines, or traps (**Table 1.1**), which can be deployed passively or actively, and are designed to target different taxonomic groups, bathomes and habitats.

2.3.1.1 Passive capture

Passive methods relate to devices that are typically handled without mechanised assistance and are not actively moved or operated by humans (excluding deployment and retrieval) (Hubert *et al.* 2012). Because of this, passive gears rely on the movements and behaviour of animals for capture, which typically occurs by entrapment (i.e. using enclosures with tapered or funnel-shaped openings that hinder escape after entrance), entanglement (i.e. ensnaring in webbing or mesh), or angling (i.e. baited hooks) (Hubert *et al.* 2012). Passive sampling has long been one of the preferred tools for appraising commercial fisheries, with pelagic longlines, for example, being the most widespread gear in the open ocean (Baum *et al.* 2003, Watson & Kerstetter 2006). Likewise, gill and trammel nets are routinely used in a variety of marine and estuarine habitats (Hanan *et al.* 1993, Gray *et al.* 2005).

Benefits

Passive gears are generally inexpensive, simple in design and construction, and their operation commonly demands little specialised training. They can be used to yield insights into species relative abundance, geographic distributions, size compositions, sex ratios, or reproductive strategies (e.g. Stevens & McLoughlin 1991, Santana-Garcon *et al.* 2014a, Ohshimo *et al.* 2016). A distinct advantage of passive methods is their ability to acquire specimens that can be visually identified and sampled for taxonomic, genetic, and molecular analysis (Lavery & Shaklee 1989, Stewart *et al.* 2016). Light trapping, for instance, has been used extensively for the collection of laboratory and museum specimens (Costello *et al.* 2017). Effort is also relatively straightforward to control for, as it can be expressed as a standardised measure of soak time, for any given gear configuration (Hubert *et al.* 2012).

Limitations

The use of passive methods, however, is generally restricted to areas free of obstructions, snags, and floating debris, as well as locations with minimal turbidity and current (Costello *et al.* 2017). Importantly, all passive methods are selective for certain species, size classes, and/or sexes (Revill *et al.* 2007), such that a quantitative understanding of selectivity at each step of the capture sequence (from gear encounter, to capture, and eventually retention) is required for correct interpretation of the resulting data. For instance, estimates of growth rate, population size, or body condition can be biased as a consequence of the over/under-representation of some animals relative to others (Hubert *et al.* 2012). If it exists, prior knowledge of gear selectivity (e.g. from fishers' experience) can be harnessed to increase sampling efficiency in single-species assessments, yet this information is rarely available. As with other capture methods, the assumption that the catch-per-unit-effort (CPUE) of passive gears is proportional to true species density has also been insufficiently verified, with many factors (e.g. season, temperature, time of day, turbidity, currents, schooling behaviour, crepuscular activity, mesh size, net hanging ratio, soak time, fleet behaviour and information sharing) ultimately affecting capture efficiency (Ward *et al.* 2004, Ward 2008, Costello *et al.* 2017). Catches may therefore not accumulate at uniform rates, and may be limited by gear saturation if the probability of capture at any given time point depends on the number of animals previously caught (Prchalová *et al.* 2011). Similarly, species, sex and ontogeny-related responses to light traps can occur, making them more useful for some organisms than others (Costello *et al.* 2017). Further issues arise with bait-reliant gears, as rates of bait loss are seldom quantified (Ward & Myers 2007), and bait release tends to decrease exponentially over time. Pelagic species (e.g. striped marlin, spearfish, or bigeye tuna) are additionally susceptible to capture on the sinking (setting) and rising (retrieving) of hooks on longlines, a trait which confounds estimates of capture efficiency if setting and retrieving times vary. The types and sizes of hooks, baits, and lures can also affect selectivity (Løkkeborg & Bjordal 1992, Piovano *et al.* 2010). For obvious reasons, both direct mortality and sub-lethal effects (e.g. physical stress) represent significant concerns for the use of capture methods within MPAs, particularly where threatened or endangered species are found (Letessier *et al.* 2017). Bycatch is another major issue, both during use (Gilman *et al.* 2006, Bull 2007, Brothers *et al.* 2010) and following abandonment or accidental loss at sea (Uhlmann & Broadhurst 2015, Wilcox *et al.* 2015). Lastly, passive gears can contribute to the unintended spread of invasive alien species in sensitive habitats (Bax *et al.* 2003).

<u>Selected topical reviews</u>

- Hubert WA, Pope KL, Dettmers JM (2012). Passive capture techniques. *In*: Fisheries Techniques, 3rd Edition, pp. 223-265. Zale AV, Parrish DL, Sutton TM (Eds). Bethesda, Maryland, American Fisheries Society, 1009 p.
- Gabriel O, Lange K, Dahm E, Wendt T (2008). Fish catching methods of the world. Wiley-Blackwell, Oxford, UK, 536 p.

2.3.1.2 Active capture

The use of active capture sampling is widespread in oceanic waters, as it has long formed a staple of monitoring activities for commercially important stocks of marine species (Costello *et al.* 2017). The majority of active capture methods (e.g. pelagic trawling) rely on human or mechanical power to move sampling gear (i.e. nets) through the water. Nets can vary hugely in size, shape and mesh dimensions, each of which will be tailored to the specific characteristics of the target taxa, from small-sized, slow-moving plankton, to large-bodied, fast-moving vertebrates (Templado *et al.* 2010). Breath-holding or SCUBA spearfishing is a predominantly recreational activity, with an increasing focus on pelagic species in the last few decades (Young *et al.* 2015b).

<u>Benefits</u>

Active capture methods share many of the advantages of their passive counterparts, including the direct identification of specimens brought on-board, and the ability to obtain empirical measurements of individual-level traits such as body length, body mass, age, or stomach content composition (Wienerroither *et al.* 2009, Heino *et al.* 2010). Those small-sized, 'fragile' organisms that tend to sustain substantial physical damage in the net's codend (e.g. Ctenophores) can still be identified using *in situ* video systems (e.g. Underwood *et al.* 2014). Pelagic trawls can be towed across large volumes of water, and are therefore attractive for animals that distribute too sparsely to be efficiently sampled with other methods (e.g. optical devices). Once difficult and labour-intensive to operate, nets can today be fitted on most vessels without special facilities, and be controlled remotely using autonomous, multiple-layer systems that receive commands via acoustic or conducting cables (Oozeki *et al.* 2012). Plankton nets can also often be deployed from platforms of opportunity, maximising coverage at little extra cost. Indeed, the Continuous Plankton Recorder (CPR) programme (Edwards *et al.* 2010) (**Figure 2.4**) owes much of its success to the simple and robust design of its plankton collection device, which enables it to be towed behind a wide range of vessels on their normal trading routes and at their conventional operating speeds, unaccompanied by research staff. The 'child' AusCPR survey run in Australia as a facility of the Integrated Marine Observing System (IMOS, Hill *et al.* 2010) has built on this success, contributing insights into plankton abundance, biomass and composition at continental scales (Davies *et al.* 2016).

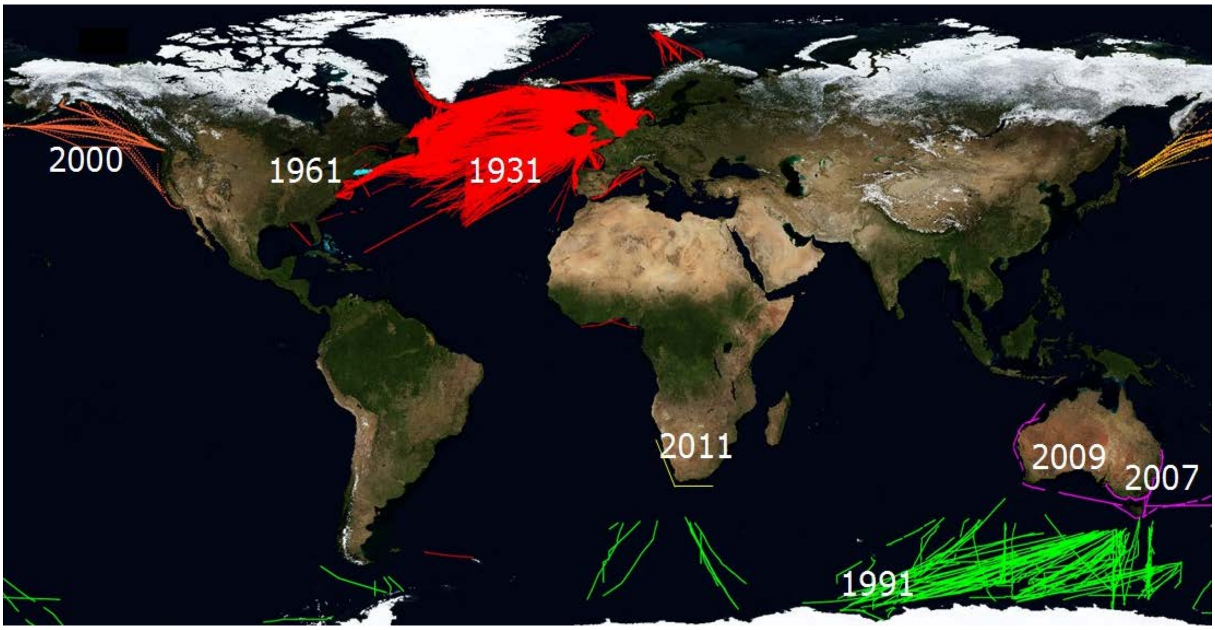


Figure 2.4 Historical coverage of Continuous Plankton Recorder (CPR) surveys worldwide. Transects are colour coded by their respective regional programme. The start year is also shown for each survey. Source: Global Alliance of Continuous Plankton Recorder Surveys, https://www.cprsurvey.org/.

Limitations

Active capture methods are limited by persistent bias and heterogeneity in gear efficiency and catchability (Oozeki et al. 2012). Pelagic trawls, for instance, are well known to be species and size-selective (Hylen *et al.* 1995). Ultimately, trawl size has to be traded off against mesh size, with finer-meshed nets requiring lower tow speeds that may be insufficient to capture animals showing avoidance behaviour (Suuronen *et al.* 1997). The percentage of organisms escaping or being retained by the net may therefore differ from haul to haul, biasing comparisons between studies (Misund *et al.* 1999, Heino *et al.* 2010). Importantly, mid-water capture sampling retains little of the context upon which each specimen is collected (e.g. lack of spatial distribution data due to all species being amassed in a single codend, vertical bias if species have differential spread through water column). Some animals/species may also exhibit a higher probability of entanglement in the forenet, without ever entering the codend (Kashkin & Parin 1983).

Selected topical reviews

- Gabriel O, Lange K, Dahm E, Wendt T (2008). Fish catching methods of the world. Wiley-Blackwell, Oxford, UK, 536 p.
- Valdemarsen JW (2001). Technological trends in capture fisheries. *Ocean & Coastal Management*, 44(9-10): 635-651.

2.3.2 Hydroacoustic methods

The limited penetration of light in seawater makes optical tools inappropriate for pelagic monitoring over large areas and depth ranges. Sound, on the other hand, propagates easily below the ocean surface, and the development of hydroacoustic technologies that can efficiently detect and locate pelagic biota based on the emission and reception of sound waves has therefore evolved rapidly (Horne 2000, Martignac *et al.* 2015, Benoit-Bird & Lawson 2016) (**Figure 2.5**). Active methods analyse the scattering of sound pulses generated by one or more transducers, whereas passive methods infer the distribution and behaviour of soniferous species by 'eavesdropping' on their natural vocalisations (Mann *et al.* 2008). The former approach has been extensively used to support biomass estimations (Scalabrin & Massé 1993, Letessier *et al.* 2016, Proud *et al.* 2018) and stock assessments in fisheries research (Amin & Nugroho 1990, Honkalehto *et al.* 2011, Hashim *et al.* 2017). Whether active or passive, hydroacoustic methods generally share the advantages of allowing rapid, non-invasive, remote sampling of pelagic communities with high spatio-temporal resolution and coverage (Benoit-Bird & Lawson 2016). Although their performance can be hampered by adverse sea states (Knudsen 2009), acoustic instruments can typically be used in conditions where many other methods would prove unviable (e.g. darkness, turbid waters). Importantly, truly synoptic observations of multiple taxa and/or physical processes can often be gained through acoustics (Baran *et al.* 2017), allowing quantitative measures of baseline conditions and assessments of the state of wildlife populations over time (Kracker 2007, Lammers *et al.* 2008). For these reasons, acoustic methods have been proposed as useful tools for evaluating the effects of pelagic MPAs (Egerton *et al.* 2018).

2.3.2.1 Passive acoustics

Passive acoustic methods allow the near-continuous detection and monitoring of both biological activity and man-made noise in marine environments (Marley *et al.* 2017c). Standard instrumentation typically comprises a hydrophone that converts sounds into a voltage that can be recorded and analysed (Mann *et al.* 2008). Pelagic hydrophones can either be fixed (i.e. moored to the seafloor, or roped to a buoy; Mellinger *et al.* 2007, Sousa-Lima *et al.* 2013), mobile (e.g. towed behind a ship, fitted on an ocean glider; Baumgartner *et al.* 2013, Wall *et al.* 2017), or even miniaturised within attachable tags deployed on individual animals (Johnson & Tyack 2003). A mobile approach grants larger geographic coverage, while a fixed one delivers longer time series that may span up to weeks or even years (Curtis *et al.* 1999, Mann *et al.* 2016). Passive acoustic receivers vary greatly in capabilities and costs, from small, hand-deployable units to sophisticated systems deployed from large vessels (Sousa-Lima *et al.* 2013). Passive acoustic methods are particularly widespread in marine mammal studies due to the complex array of vocalisations produced by whales and dolphins (Di Sciara & Gordon 1997).

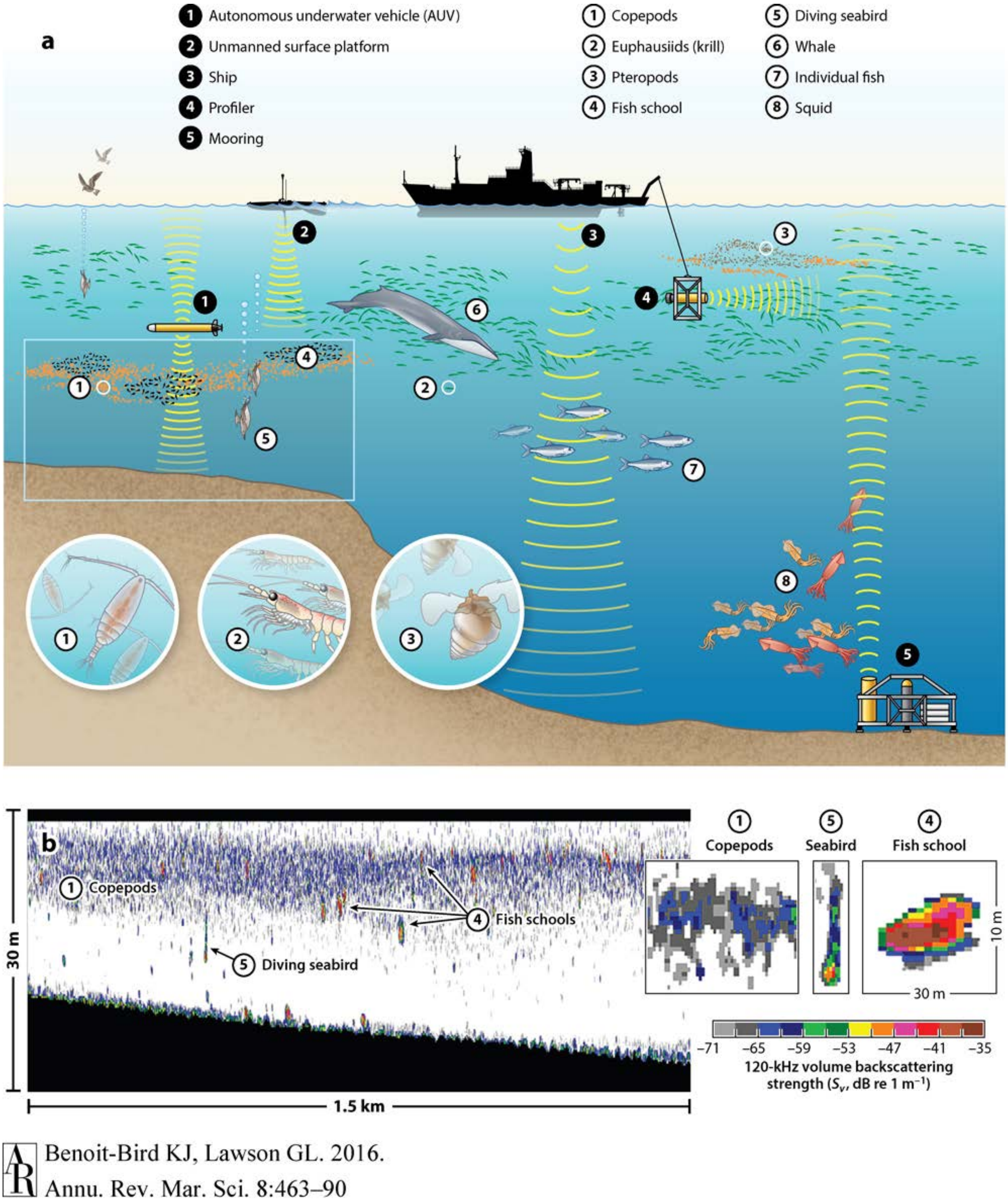


Figure 2.5 (a) Subset of acoustic methods used to study pelagic organisms, illustrating varying capabilities for target detection, depths of operation, degrees of autonomy, and sizes. **(b)** Example echogram (corresponding to the shaded rectangle in panel a), showing data from a 120-kHz echosounder inside an autonomous underwater vehicle. Both the quantitative backscatter data and the morphometrics of echo distributions can be used to identify key targets, including the bubble streams left behind by diving seabirds, schools of fish, and layers of mesozooplankton. Figure reproduced with permission from Annual Reviews (Source: Benoit-Bird & Lawson 2016).

Passive acoustic data can yield insights into the distribution patterns and seasonal occurrence of key species (Verfuß *et al.* 2007), as well as their migratory movements (Comeau *et al.* 2002) and responses to global anthropogenic change (Rogers *et al.* 2013), with direct implications for spatial conservation planning and for MPA monitoring (Casale *et al.* 2016, Merchant *et al.* 2016, Sánchez-Gendriz & Padovese 2016, Heenehan *et al.* 2017). Recognition of the negative effects of noise on marine wildlife (Gordon *et al.* 2003, Kight & Swaddle 2011) is also now encouraging explicit consideration of anthropogenic soundscapes in conservation planning and protected area designation processes (Williams *et al.* 2015a). As a result, applications of passive acoustics to the characterisation of noise impacts from human activities (e.g. vessel traffic, pile-driving and port construction activities, seismic exploration, military sonar exercise, renewable energy developments) are booming (Bailey *et al.* 2010, Ou *et al.* 2011, Merchant *et al.* 2014).

Furthermore, hydrophone arrays (consisting of anywhere between two and hundreds of units) allow sound sources to be accurately localised, a key requirement for producing estimates of absolute species density (Marques *et al.* 2013). The potential for sound signatures ('soundscapes') to act as indicators of biodiversity is also an active area of research (Parks *et al.* 2014, Pieretti *et al.* 2017).

Benefits

Key advantages of passive acoustic methods include their cost-effectiveness, their autonomous and non-obtrusive nature, and their ability to operate at night and in poor weather, offering a valuable alternative for monitoring biodiversity when traditional (e.g. visual) surveys are impractical or impossible (Staaterman *et al.* 2017). Because acoustic data can be collected over a wide range of habitats and depths for long periods of time, passive acoustic methods are attractive for mapping pelagic species' ranges and distributions year-round (Wall 2014). Passive acoustic technologies applied to mobile autonomous underwater platforms (e.g. gliders) benefit from minimum environmental impact, covertness, and the availability of several functionalities, including automated noise detection and classification algorithms (Tesei et al. 2015). Some models can dive below 1,000 m, a depth range at which some deep-diving cetaceans such as sperm and beaked whales typically forage and vocalize (Mellinger et al. 2017). In addition, acoustic recordings are permanent data records that can be archived for future use and re-analysed if spurious results appear or new processing techniques develop (Rogers *et al.* 2013, Simard *et al.* 2015). Acoustic data are largely independent of collection error and inter-observer bias, and readily collect information on multiple species, which may be useful for monitoring long-term changes in community composition (Rogers *et al.* 2013). For example, fish aggregating devices (FADs) are increasingly being fitted with passive acoustic sensors for pelagic monitoring purposes (Dagorn *et al.* 2007, Gandilhon *et al.* 2010, Govinden *et al.* 2013, Moreno *et al.* 2016).

Limitations

The greatest impediment to using passive acoustics lies in a limited understanding of the behavioural context of sound production (and its variability across locations, seasons, time of day, sex and age classes, etc.) for numerous organisms (Amorim 2006, Mann *et al.* 2016, Lewis & Širović 2017). Some species will be inherently more amenable to acoustic surveys than others, depending on the frequency, source level, and directionality of their vocalisations (Zimmer 2011). In many cases, knowledge of the relationship between calling rates and animal density remains tenuous, particularly when the proportion of the population available for detection individual recorders is unknown. Because of this, acoustic data are often reduced to rough proxies of abundance (Mellinger *et al.* 2007). Most passive acoustic devices also store data internally, and must therefore be physically recovered before analysis can begin. Manual data processing remains a labour-intensive exercise, and despite significant advances, automated tools need to be consistently tested for accuracy (e.g. false positives/negatives), with mixed results obtained so far (Bittle & Duncan 2013). In particular, species discrimination can be hampered by significant overlap in the spectral and temporal characteristics of concurrent calls, combined with a wide range of call types across different taxa (Rankin *et al.* 2017). Other bottlenecks exist around data accessibility, as the large volumes of information generated by passive acoustic detectors are challenging to make available in easy to use, distributable formats. Detection probability and receiver performance are also a function of background noise, with acoustic interferences such as masking potentially hampering species identification and group size estimation (Clark *et al.* 2009). Data from implantable acoustic tags are generally constrained to the small scales of their associated receiver arrays, which may not be sufficient relative to monitoring objectives (with some exceptions; Hoenner *et al.* 2018, Bruce *et al.* In press). This method is also invasive, requiring both animal capture and handling (Mulcahy 2003). Lastly, the ocean is harsh on acoustic gear. Corrosion, fouling, and damage from currents, tides, or storms, can all affect the longevity and efficiency of acoustic instruments (Dudzinski *et al.* 2011).

<u>*Selected topical reviews*</u>

- Baumgartner MF, Stafford KM, Latha G (2018). Near real-time underwater passive acoustic monitoring of natural and anthropogenic sounds. *In*: Observing the Oceans in Real Time, pp. 203-226. Venkatesan R, Tandon A, D'Asaro E, Atmanand MA (Eds). Springer International Publishing, 323 p.
- Gannon DP. 2008. Passive acoustic techniques in fisheries science: A review and prospectus. *Transactions of the American Fisheries Society*, 137(2): 638-656.
- Horne JK. 2000. Acoustic approaches to remote species identification: A review. *Fisheries oceanography*, 9(4): 356-371.
- Mann DA, Hawkins AD, Jech JM (2008). Active and passive acoustics to locate and study fish. *In*: Fish Bioacoustics, pp.279-309. Webb JF, Fay RR, Popper AN (Eds), Springer, 318 p.
- Sousa-Lima RS, Norris TF, Oswald JN, Fernandes DP (2013). A review and inventory of fixed autonomous recorders for passive acoustic monitoring of marine mammals. *Aquatic Mammals*, 39(1): 23-53.
- Van Parijs S, Clark C, Sousa-Lima R, Parks S, Rankin S, Risch D, Van Opzeeland I (2009). Management and research applications of real-time and archival passive acoustic sensors over varying temporal and spatial scales. *Marine Ecology Progress Series*, 395: 21-36.

2.3.2.2 Active acoustics

A variety of active acoustic systems is available, ranging from fixed moorings to autonomous platforms, lightweight towed or pole-attached gears, or larger hull-mounted arrays (Bean *et al.* 2017) (**Figure 2.5**). In its most basic form, however, active acoustic sampling usually involves the use of downward-facing echosounders that ensonify the water column by transmitting regular sound pulses ('pings'), and measuring the returning echoes after the sound has scattered off the seabed or any other physical and biological targets (e.g. plankton swarms or fish schools; Kracker 2007, Kaiser & Attrill 2011). The strength of this backscatter is frequency-dependent, such that it can be used to estimate the relative composition and size distribution of species assemblages, based on their known acoustic properties (Bjerkeng *et al.* 1991, Thompson & Love 1996, Logerwell & Wilson 2004).

Benefits

Pelagic organisms are well-suited to acoustic sampling as numerous species display highly aggregative behaviour (e.g. schooling) and patchy distributions that may be inefficiently captured with alternative techniques (e.g. trawling) (Bean *et al.* 2017). A key benefit of active acoustic methods lies in their fine spatial resolution and their ability to collect data on multiple species simultaneously and nearly continuously from a moving vessel (or other platform) (Benoit-Bird & Lawson 2016). Data acquisition is typically fast, and information recorded digitally with immediate availability. This allows unique insights into numerous aspects of pelagic ecology (Benoit-Bird & Lawson 2016), including trophodynamics, ecosystem structure (Benoit-Bird & McManus 2012), or animal distributions and movements across scales spanning kilometres to entire ocean basins (Kaltenberg & Benoit-Bird 2013, Trenkel & Berger 2013). For example, echosounders have helped map previously undocumented cod breeding regions, and determine the timing of their associated spawning migrations (Rose 1993). Likewise, autonomous underwater vehicles fitted with acoustic probes have allowed inaccessible parts of the Antarctic ice pack to be surveyed, revealing the importance of the ice edge to krill (Brierley *et al.* 2002). Amplitude mixing of multiple acoustic frequencies can also support stronger species identification capacity (Kloser *et al.* 2002), and investigations of animal movement behaviour are made possible by the tracking of individuals with split-beam transducers (Handegard *et al.* 2005). Increasing integration of acoustics systems as standard equipment on commercial and fishing industry vessels will afford exponentially growing opportunities to monitor pelagic habitats beyond the reach of traditional research cruises (Benoit-Bird & Lawson 2016).

Limitations

The main limitation of hydroacoustic methods relates to their usually poor taxonomic resolution. The received acoustic signal is a combined function of species morphometry (e.g. body shape, length, width), anatomy (presence/absence and size a gas-filled swim bladder), physiology (e.g. gonad production, lipid content, gut fullness), and behaviour (e.g. schooling) (Jørgensen 2003, Brehmer *et al.* 2007). The acoustic scattering properties (target strength) of many taxa also remain largely undescribed (but see Lee & Shin 2005), making species identification difficult without secondary information. Furthermore, organisms must be separated enough to be properly discriminated, making abundance/biomass estimation problematic when animal density is high (e.g. compact shoals of small fish). Hull-mounted technologies must be appropriately calibrated for reliable abundance estimation and often provide inadequate sampling at boundaries such as the sea surface or near the seafloor (but see Baran *et al.* 2017), with a limited range of available vessels that are fit-for-purpose. Temperature-salinity profiles are required for accurate quantifications of sound propagation and absorption rates (Pyć *et al.* 2016) but can be difficult to acquire *in situ*, particularly from commercial vessels. Additional uncertainty can arise from animal avoidance behaviour, and echogram data can be corrupted by parasite signals from bubbles or drifting debris (Brehmer *et al.* 2006). Acoustic emissions ultimately contribute to marine noise pollution, an issue increasingly recognised as a significant and pervasive threat to ocean ecosystems and wildlife (Simmonds *et al.* 2014, Williams *et al.* 2015b, Kunc *et al.* 2016). Lastly, the start-up costs for hydroacoustic equipment may be an obstacle to their adoption in MPA studies (Egerton *et al.* 2018), with active acoustics gear being generally power-hungry and generating voluminous and inherently complex datasets that demand specialised knowledge and advanced computational tools for analysis.

<u>*Selected topical reviews*</u>

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2.3.3 Visual, optical & thermographic methods

Visual, optical and thermographic methods encompass a large array of instruments and techniques designed to function in air or underwater, using either the naked eye or some form of photographic/video/heat signature recorder.

2.3.3.1 Underwater methods

Whether diver-reliant or diver-free, underwater visual techniques have been used for decades in marine ecology studies (Murphy & Jenkins 2010). Prominent examples include:

- Underwater visual census (UVC) and diver-operated video (DOV), which require a swimmer, snorkeler or SCUBA diver to record observations along a predefined survey route (Fontes *et al.* 2014, Juhel *et al.* 2018);

- Baited remote underwater video systems (BRUVS), which consist of a frame that supports a container filled with bait to attract animals into the field of view of a camera (or camera pair) (Letessier *et al.* 2013, Santana-Garcon *et al.* 2014c);
- Towed underwater video transects, where recording equipment is towed at a constant depth behind a moving vessel (Riegl *et al.* 2001);
- Drop ('drift') cameras, designed as autonomous, deep-submergence, buoyancy-compensated systems (Berkenpas *et al.* 2013).

Important technological advances (e.g. camera resolution, sensors, battery life, and information storage) have propelled these methods to the forefront of pelagic science in recent years, with significant investments being made towards developing standard operating procedures that can facilitate their use in pelagic monitoring globally (Whitmarsh *et al.* 2017, Bouchet *et al.* 2018b).

Benefits

Underwater visual methods have become popular as non-lethal and cost-effective means of observing and measuring whole assemblages of pelagic biota in habitats that otherwise could not be easily sampled (Boldt *et al.* 2018). They can provide permanent, high-definition archives of the data, yield insights into animal behaviour (Santana-Garcon *et al.* 2014b, Kempster *et al.* 2016) outside of laboratory settings, and quantify species-environment relationships with sufficient spatio-temporal replication to support the development of predictive statistical models (Schmiing *et al.* 2013, Bouchet & Meeuwig 2015, Gonzáles-Andrés *et al.* 2016). Many aspects of the life histories of pelagic taxa remain largely unknown, and visual methods can be a powerful way of filling these knowledge gaps, for example by documenting biologically important areas like spawning (Fukuba *et al.* 2015) or nursery grounds (Meeuwig JJ, unpublished data). Archived video footage can be shared and analysed independently, ensuring data traceability and enabling both discussions about species identifications and the cross-validation of subsequent analyses. Stereo-systems (i.e. fitted with camera pairs) are advantageous in making measurements in three-dimensional space possible, such that animal body lengths can be accurately computed based on epipolar geometry (Letessier *et al.* 2015). Remote systems have the added advantage of reducing biases related to gear or diver avoidance, and baited systems benefit from increased predator encounter rates and detection probabilities, without precluding the sampling of prey or herbivorous species (Bouchet *et al.* 2018b). Most underwater visual methods are quick to deploy and straightforward to operate, making efficient use of boat and researcher time.

Limitations

Diver-based methods are usually constrained to clear, shallow waters (i.e. < 30 m) where SCUBA activities can be performed safely (e.g. in tropical Australia, outside the range of saltwater crocodiles), and can yield substantially biased observations if animals respond to human presence (Dickens *et al.* 2011, Lindfield *et al.* 2014). UVC and DOV are also influenced by variability in the divers' swimming speeds and taxonomic skills, resulting in errors that are unlikely to be noticed or cannot be verified without video imagery - although this can be addressed with comprehensive training and quality control (e.g. Reef Life Survey) (Stuart-Smith *et al.* 2017).

Midwater towed video systems are generally inappropriate for pelagic sampling (Assis *et al.* 2007), and few examples of their use of in pelagic habitats accordingly exist (but see Riegl *et al.* 2001). To date, there has been little effort to measure rates of bait release and plume dispersal, undermining our understanding of the effective range of attraction available to baited instruments under various ocean conditions (Heagney *et al.* 2007). For camera-based methods, further issues can arise with restricted fields of view, as screen saturation will lead to underestimates of animal abundance at high population densities (Kilfoil *et al.* 2017). Although seldom quantified or addressed, visibility remains a pervasive problem that could inflate the prevalence of false negatives, particularly for shy animals that do not approach the sampling equipment (**Figure 2.6**). Lastly, the labour costs and time investment necessary to process and annotate video imagery are often seen as substantial shortcomings – although progress in the development of deep learning and computer vision algorithms is earmarked to automate this process in the future (Salman *et al.* 2016, Westling *et al.* 2016, Shafait *et al.* 2017).

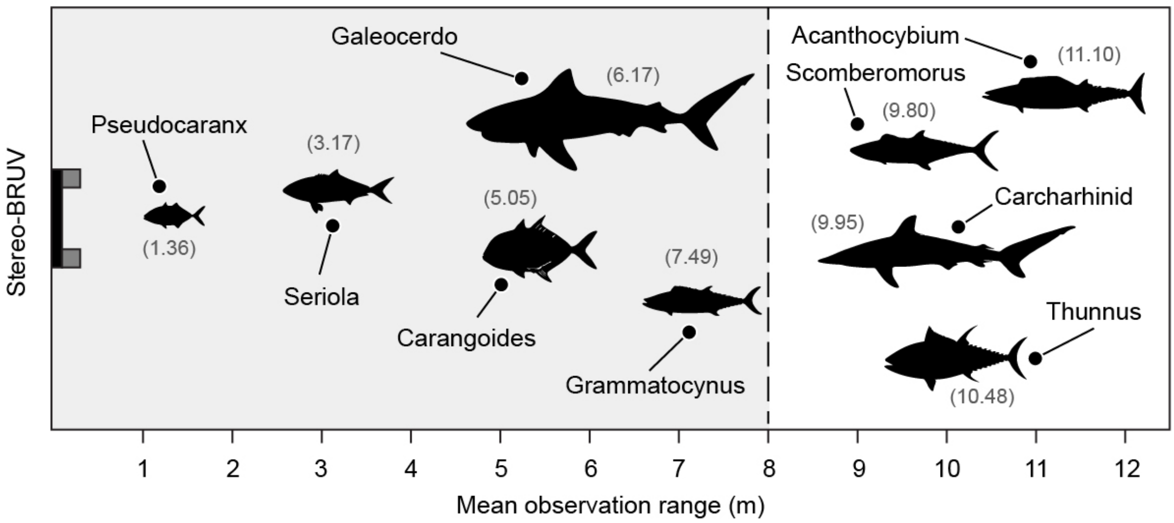

Figure 2.6 Mean observation ranges for a range of example pelagic genera recorded on stereo-BRUVS in the Houtman Abrolhos Islands, Western Australia (mean values shown in grey). The shaded area shows the range over which fish lengths can be accurately measured with a typical stereo-BRUV setup. Data source: Santana-Garcon *et al.* (2014d).

<u>*Selected topical reviews*</u>

- Cappo M, Harvey E, Malcolm H, Speare P (2003). Potential of video techniques to monitor diversity, abundance and size of fish in studies of marine protected areas. *Proceedings of the World Congress on Aquatic Protected Areas proceedings*, Cairns, Australia, 455-464.
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▪ Struthers DP, Danylchuk AJ, Wilson AD, Cooke SJ (2015). Action cameras: Bringing aquatic and fisheries research into view. *Fisheries*, 40(10): 502-512.

2.3.3.2 Airborne methods

Airborne methods (e.g. transect surveys from a manned aircraft or helicopter) are critical tools for estimating the distribution, abundance and health of wildlife species, particularly over large spatiotemporal scales (Roberts *et al.* 2016, Laran *et al.* 2017). In marine systems, most applications tend to target air-breathing megavertebrates that are regularly visible at the surface (Sleeman *et al.* 2007), such as whales (Salgado Kent *et al.* 2012), dolphins and porpoises (Allen *et al.* 2017), seabirds (Buckland *et al.* 2012, Pettex *et al.* 2017), dugongs (Pollock *et al.* 2006), and sea turtles (Lauriano *et al.* 2011). However, pelagic fishes like tuna (Royer *et al.* 2004), swordfish (Lauriano *et al.* 2017), sunfish (Grémillet *et al.* 2017), capelin (Naumenko 2002) or anchovy (Davison *et al.* 2017), as well as elasmobranchs including white sharks (Dicken & Booth 2013), whale sharks (Rowat *et al.* 2009) and manta rays (Martin *et al.* 2016), can all also be successfully surveyed from the air. Spotter planes indeed boast a long tradition of use in fisheries to assist locating schools of tunas (Bauer *et al.* 2015), sardines (Kaplan *et al.* 2016) and other pelagic fishes. In Australia, multiple NERP/NESP projects have relied on airborne methods for marine monitoring (Hagihara *et al.* 2016, Bannister 2017, Bouchet *et al.* 2018c).

Of all available airborne methods, unmanned aerial vehicles (UAVs) have experienced a remarkable uptake in environmental research over the last decade. Also known as 'drones', remotely piloted aerial systems (RPAS) or unmanned aerial systems (UAS), these instruments are light-weight, portable platforms piloted remotely from the ground, and allowing surveys of remote, hard-to-reach areas within small time windows. Their potential as a survey tool in pelagic habitats is therefore quickly gaining attention, especially as continuing progress in their design and engineering (e.g. component miniaturization, lithium batteries, high-resolution sensors) is rapidly making them more versatile and affordable on the civilian market (Colefax *et al.* 2017) (**Table 2.1**). There are abundant designs for UAVs. The key distinction in terms of their capability and ease of operation is their physical size and power, which limits their payload carrying capacity, operating altitude, and range (Anderson & Gaston 2013).

<u>Benefits</u>

Manned aircraft are often deemed to provide the greatest return on investment per sample, as they can operate for long durations and cover wider stretches of ocean more quickly than UAVs. However, UAVs may be a more efficient approach at finer spatial scales (e.g. few square kilometres, or in isolated locations) (**Table 2.1**), with potential for increased sighting rates (Hodgson *et al.* 2013). The altitude and speed specifications of a given flight can often be selected to maximise the detectability of target fauna. When operated from an airplane, light detection and ranging (LiDAR) systems also enable rapid surveys of the distribution and abundance of dense-schooling fish stocks in shallow coastal waters, within an operational depth range of 30-40 m, depending on water clarity (Churnside *et al.* 2003, Carrera *et al.* 2006, Churnside *et al.* 2017). Digital cameras delivering stills and video feeds can be used to enhance encounter rates, although usually within a narrower search swath located immediately beneath the plane (Buckland et al. 2012).

Table 2.1 Comparison of traditional and Unmanned Aerial System (UAS) surveys of Steller sea lion (*Eumetopias jubatus*) haul outs and rookeries in Alaska. Surveys were conducted by the National Oceanic and Atmospheric Administration. Table reproduced with permission from Wiley (Source: Christie *et al.* 2016).

	Manned aerial surveys	UAS surveys
Purpose of surveys	Estimate the abundance of Steller sea lions in the inner Aleutians	Estimate the abundance of Steller sea lions in the outer Aleutians
Cost (per day)	$4700 per day including fuel and pilot, or $400 per site	$3000 per day based on the cost of vessel support, or $1700 per site
Type of aircraft	NOAA Twin Otter	APH-22 hexacopter
Distance/area surveyed	2500 km of coastline, including the Gulf of Alaska and part of Aleutians; 210 sites surveyed	400 km of coastline along the western Aleutian chain, 30 sites surveyed; maximum distance from the vessel was 634 m, longest flight was 16 minutes
% animals detected	100% hauled-out animals	100% hauled-out animals
Data collected	Quantitative imagery, animal counts	Quantitative imagery, animal counts, individual identification
Number of personnel	6	2
Observed effect on animal	Slight and variable, 5% of adults moved toward water	Very low to none, 0.3% adults moved toward water
Advantages	(1) surveyed up to 50 sites per day (2) high-quality images (3) cost per site low	(1) surveyed remote sites with no airfields (2) extremely low disturbance (3) very high-quality images (flew at altitude of 45 m) (4) less subject to flight restrictions due to weather conditions (5) Biologists can double as pilots
Disadvantages	(1) requires good weather at primary and alternate airfields (minimum of 750-ft ceilings) (2) relatively noisy (3) may only fly on half (or less) of days available (4) requires a runway for takeoff/landing (5) imagery has lower resolution (flight altitude: 150-305 m) (6) requires flight crew of 3 plus 3 observers	(1) can only survey a few sites (1-3) per day (2) requires costly vessel for use as a transport (3) cannot fly in high winds (wind speed must be less than 25 knots on the ground) (4) must stay within line-of-sight and 0.8 km of observer

Images are then available for independent verification and automated animal detection (Seymour et al. 2017). In many cases, multiple taxa are recorded simultaneously during a given survey (e.g. Laran *et al.* 2017), such that opportunities for collaboration are fostered and the total survey effort needed per species group is reduced (Bouchet *et al.* 2018c).

UAVs can also be fitted with an array of sophisticated sensors (Gonzalez *et al.* 2016), processors, and samplers (e.g. petri dishes; Pirotta *et al.* 2017), allowing the non-invasive collection of biological material and the generation of real-time, high-quality imagery with considerable gains in accuracy and precision over traditional counts for some species (Hodgson *et al.* 2016). Furthermore, multirotor UAVs frequently boast vertical take-off and landing capabilities, obviating the need for additional landing equipment and making them suitable for launching and retrieving from small vessels (Anderson & Gaston 2013). Although still uncommon, some consumer-level fixed-wing UAVs are manufactured to be launched by hand or small catapults, which may be practical from small vessel platforms also. Finally, as aviation accidents account for over two-thirds of fatalities among wildlife biologists (Sasse 2003), the improved mission safety conferred by UAVs is indisputably one of their biggest assets.

Limitations

Aerial surveys are logistically difficult to implement, particularly in developing countries, and incur high costs from aircraft hire and staffing, sometimes exceeding US $1,000 per survey hour. Financial support from external donors is thus often necessary (Linchant et al. 2015), but can be unpredictable, jeopardising long-term sampling coverage and making monitoring initiatives challenging (Bauer *et al.* 2015). Aircraft and flight plans may also change if extended delays occur between successive campaigns, which complicates comparisons between datasets and studies (Linchant *et al.* 2015). Importantly, traditional airborne methods are contingent on good visibility (e.g. clear weather, sufficient daylight, calm seas) and are ineffective at capturing organisms that stay submerged for long periods, such that only a fraction of those individuals present is actually recorded ('availability bias'; Panigada *et al.* 2017). While long-endurance, fixed-wing UAV platforms exist, these are often either military-based craft or custom-built, at substantially higher expenditure. The majority of available UAVs is therefore only useable over limited ranges (i.e. within line-of-sight), at slow speeds, and under small payloads. Additionally, stringent and country-specific civil aviation regulations and complex permitting processes can limit their adoption for scientific applications (Anderson & Gaston 2013, Christie *et al.* 2016). The capacity of UAVs to eliminate inter-observer biases is generally offset by longer manual data post-processing times, and potentially higher risks of species disturbance (Pomeroy et al. 2015).

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- Anderson K, Gaston KJ (2013). Lightweight unmanned aerial vehicles will revolutionize spatial ecology. *Frontiers in Ecology and the Environment*, 11(3): 138-146.
- Buckland ST, Burt ML, Rexstad EA, Mellor M, Williams AE, Woodward R (2012). Aerial surveys of seabirds: The advent of digital methods. *Journal of Applied Ecology*, 49(4): 960-967.

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- Hodgson AJ, Marsh H, Delean S, Marcus L (2007). Is attempting to change marine mammal behaviour a generic solution to the bycatch problem? A dugong case study. *Animal Conservation*, 10(2): 263-273.
- Kemper G, Weidauer A, Coppack T (2016). Monitoring seabirds and marine mammals by georeferenced aerial photography. *International Archives of the Photogrammetry, Remote Sensing & Spatial Information Sciences*, 41: 689-694.
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2.3.3.3 Shipboard methods

Shipboard methods continue to advance our collective understanding of the distributions and interactions of pelagic organisms across many trophic levels (Kavanaugh *et al.* 2016). Shipboard visual surveys typically employ human observers tasked with detecting and identifying species using the naked eye or with the aid of binoculars (Aragones *et al.* 1997). However, emerging technologies like infrared thermography are creating new opportunities for non-invasive marine monitoring in place of human personnel (Horton *et al.* 2017).

Benefits

Large vessels can provide ideal physical platforms for conducting a variety of monitoring activities, and examples of multidisciplinary research cruises accordingly abound (e.g. Nichol *et al.* 2013, Doray *et al.* 2017). Vessel-based still photography or video allows semi-quantitative assessments of body condition and exposure to human threats (e.g. entanglement in fishing gear) in large air-breathing megafauna, with a level of detail usually not attainable from other methods such as manned airplanes (Hunt *et al.* 2013). Thermographic imaging devices can also be useful for detecting warm-blooded animals (e.g. cetaceans) at a range of distances in both low- and high-latitude environments (Horton *et al.* 2017).

Limitations

Shipboard surveys are arguably among the most costly monitoring methods available, and are largely constrained by slow speeds (i.e. long travel times to and from sampling sites), finite fuel capacities, and weather conditions (typically, Beaufort sea states < 4-5). Survey areas must additionally be free of navigational hazards, with sufficient depth to allow safe operations and crossing. Responsive species movement prior to detection (i.e. attraction to, or avoidance of, the vessel) is difficult to predict but can generate substantial bias in estimates of abundance if it occurs (Palka & Hammond 2001). Thermal imaging is contingent upon animals surfacing and revealing parts of their bodies (Verfuss *et al.* 2018). As a rule, detections will rise with bigger body or group sizes, and more energetic/frequent surfacing behaviour (**Table 2.2**). Cameras with wider thermal infrared frequency bands are more capable of picking up heat cues but remain expensive, particularly so for high-sensitivity models fitted with cryogenically-cooled detectors or large focal lengths suited to long-range applications (Horton *et al.* 2017).

Table 2.2 Factors expected to increase (✓) or decrease (×) the detection performance of shipboard thermal infrared imaging vs. visual observers. Empty cells denote conditions where no such influence is expected.

Factor	Thermal imaging	Visual observer
Animal-dependent		
Surface behaviour	✓	✓
Skin pigmentation and colouring		✓
Body size	✓	✓
Strength of exhalation (air-breathers)	✓	✓
Movement in relation to vessel	✓	✓
Position relative to water surface		✓
Group size	✓	✓
Environmental		
Aerosols	×	
Fog	×	×
Glare	×	×
Light level	×	✓
Rain	×	×
Sea state	×	×
Snow	×	×
Presence of non-targets at the surface	×	×
Water temperature (but see Horton *et al.* 2017)	×	

Infrared detectors also demand a direct line-of-sight to the target(s), and can lose functionality through interaction with sea spray. The data streams generated by infrared imaging systems are voluminous, posing further challenges with data handling, analysis and signal processing. Data accessibility is another critical impediment, as many shipboard surveys are undertaken for commercial purposes (e.g. marine faunal observers collecting visual data during seismic exploration activities). The resulting data are frequently subject to confidentiality agreements that prevent their public release.

Selected topical reviews

- Hunt KE, Moore MJ, Rolland RM, Kellar NM, Hall AJ, Kershaw J, Raverty SA, Davis CE, Yeates LC, Fauquier DA (2013). Overcoming the challenges of studying conservation physiology in large whales: A review of available methods. *Conservation Physiology*, 1(1): cot006.
- Verfuss UK, Gillespie D, Gordon J, Marques TA, Miller B, Plunkett R, Theriault JA, Tollit DJ, Zitterbart DP, Hubert P (2018). Comparing methods suitable for monitoring marine mammals in low visibility conditions during seismic surveys. *Marine Pollution Bulletin*, 126: 1-18.

2.3.3.4 Land-based methods

Land-based methods are cost-effective approaches to monitoring coastal areas (Giacoma et al. 2013), and encompass both theodolites, high-frequency radars and high-resolution panoramic camera systems (Gigapan, not included in this report) (Lynch *et al.* 2015). The former were first introduced in the 1970s, and have been a major feature of cetacean research ever since (Hoyt & Hvenegaard 2002). Theodolites are surveying instruments that measure horizontal angles from some arbitrarily selected reference point (e.g. lighthouse, islet, mountain peak, headland), and vertical angles to a gravity-referenced ground-level vector. Such measurements can be mapped into geographical coordinates, provided the height of the theodolite station (above the sea surface) and the position of the reference point are accurately known (Pryor & Norris 1998). Successive, time-stamped theodolites fixes can be used to characterise animal movement trajectories in relation habitat variables, environmental cues, and anthropogenic stressors (Würsig et al. 1991). This method has been used in numerous locations around the globe to obtain insights into patterns of species occupancy (Marley et al. 2017a), residency (Wood 1998) seasonal habitat use (Tyne et al. 2015) (**Figure 2.7**), and overlap with human activities (Williams et al. 2002, Piwetz et al. 2012, Baş et al. 2015, Marley et al. 2017b).

High-frequency radio detection and ranging (radar) systems rely on the in-air propagation of electromagnetic waves to locate targets (Verfuss et al. 2018). They are commonly used to measure oceanographic parameters such as wave height, wave direction, and ocean current velocity, sometimes over hundreds of kilometres at hourly or daily resolutions (Bean et al. 2017). Radar applications include oil spill monitoring, search and rescue efforts by coast guards, ship traffic management and navigational safety, tsunami detection, and the monitoring of coastal upwelling, eddies, storm events, fish larval transport, and harmful algal blooms (Paduan & Washburn 2013, Wyatt 2014).

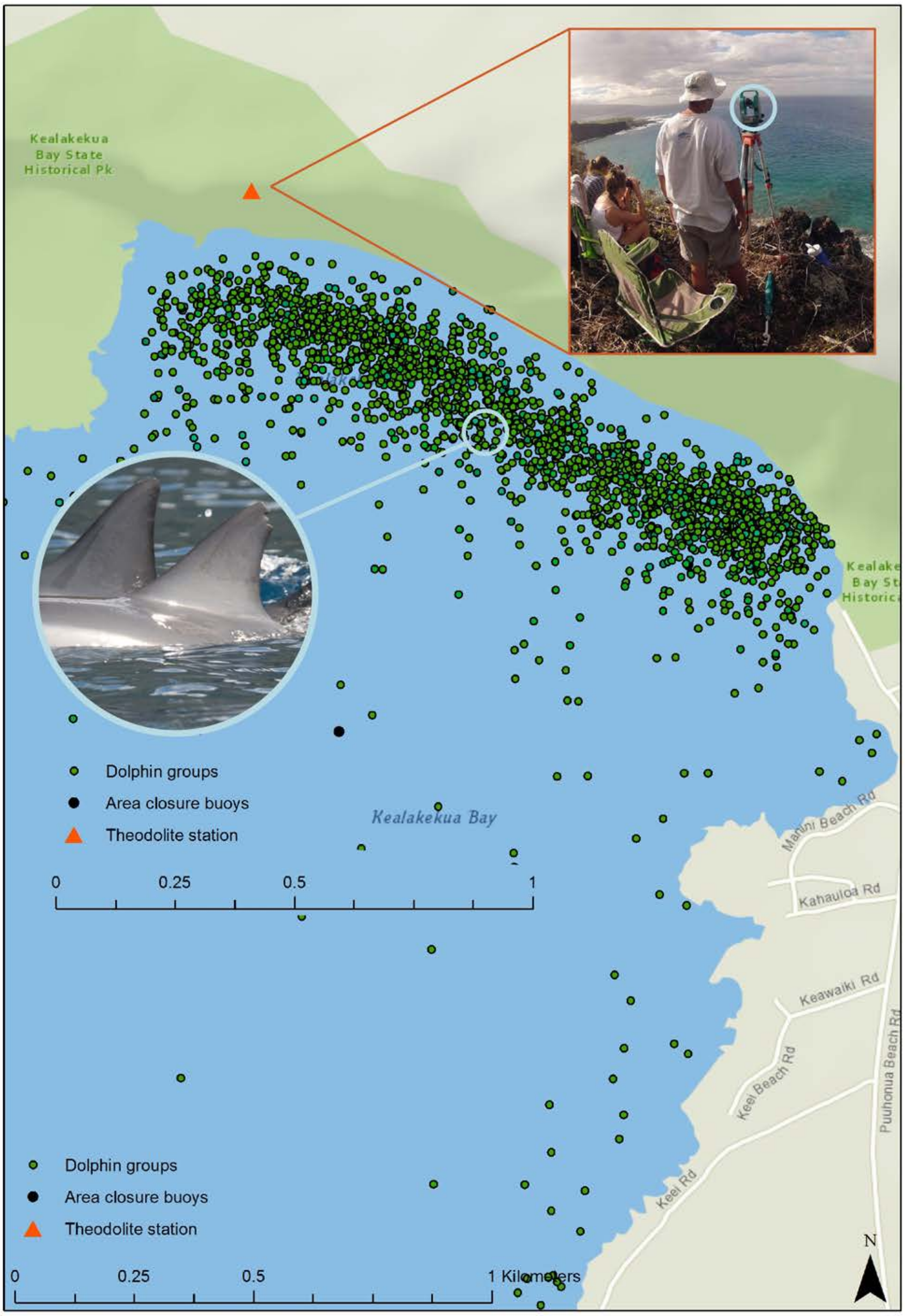

Figure 2.7 Land-based observations of spinner dolphin (*Stenella longirostris*) groups in Kealakekua Bay, Hawaii. Insets show the cliff-top theodolite station manned by three observers (orange), and a close-up view of individual dolphins (blue), akin to what would be seen through the theodolite's viewfinder. Figure adapted from material developed by the Murdoch University Cetacean Research Unit and Duke University. Photo credits: Julian Tyne, MUCRU and Demi Fox, Lenfest.

They have also been used to infer connectivity between marine protected areas within networks (Zelenke et al. 2009), and are prominent in ornithology, as they can detect and track birds approaching or departing from breeding colonies and roosting areas (Harmata *et al.* 1999, Gauthreaux Jr & Belser 2003, Lilliendahl *et al.* 2003).

Benefits

Land-based methods are non-invasive, enabling the monitoring of free-ranging organisms from a distance, without risks of observer-induced disturbance and with high levels of accuracy (Hastie *et al.* 2004). Permanent records of fine-scale animal behaviour can be obtained by combining them with simple video camera setups (Hastie *et al.* 2004). Theodolite surveys are also generally cheap compared to alternatives, and while some theodolite devices can be valued upwards of AUD $10,000, the initial investment can easily be attenuated with appropriate maintenance and cleaning, guaranteeing their proper functioning over several decades (Morete *et al.* 2018). Under budget constraints, theodolites stations can often be rented or borrowed from other academic departments (e.g., Geology, Geography). Training is straightforward and can be completed within a matter of hours, without requirements for specialised skills. While theodolites must be protected from rain and humidity, radars can gather data almost independently of weather conditions. Similarly, marine surveillance radars are relatively inexpensive, available off-the-shelf, require little modification, are easy to operate and maintain (Wyatt 2014).

Limitations

Land-based methods tend to have limited geographical coverage in the vicinity of coastal locations with high relief, and are normally constrained to relatively conspicuous species that regularly come to the surface within sight of land (Würsig et al. 1991). Measurement quality degrades rapidly with decreasing station height and increasing proximity of the targets to the horizon. Imprecise positioning can also be caused by failures to take tidal fluctuations into consideration, by inclement weather (e.g. sea state, swell), or by discrepancies in the visual acuity of different observers (Giacoma et al. 2013). Theodolite readings can be time-consuming in areas of high species density, reducing data collection to the positions of small groups or subgroups rather than individuals, forfeiting information on spatial group structure (Hastie et al. 2003). Unless animals show distinctive markings or coloration patterns, individual identification can be difficult, raising issues surrounding double counting.

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2.3.4 Ocean robotics

Robots are powerful tools for accessing environments too dangerous or too remote for human exploration (Bellingham & Rajan 2007), and ongoing collaborations between engineers, physicists and biologists are therefore essential to supporting pelagic sampling globally and into the future (Jech et al. 2009). The development of autonomous and remotely operated underwater vehicles (e.g. Argo floats, ROVs, gliders etc.) has seen an exponential increase in the second half of the 20th century, with numerous applications in research, industry, and the military. For instance, ROVs are increasingly used to conduct pelagic biodiversity assessments around oil and gas platforms during operation and decommissioning phases (Claisse *et al.* 2014, Macreadie *et al.* 2018). Today, ocean robotics are transforming from an activity previously focused on time-capped deployments overseen by experts, to round-the-clock operations led by multidiscpinary teams (Bellingham & Rajan 2007).

2.3.4.1 Manned

Typically used for deep-sea exploration (Heirtzler & Grassle 1976), manned submersibles place humans into the water column, where they can guide data collection and monitoring activities directly (Bergman 2012). Submersibles are typically untethered, and therefore decoupled from vessel platforms, which provides considerable flexibility in selecting and capturing biological specimens.

Benefits

Manned submersibles achieve a level of precision and selectivity that remains unsurpassed by any other method of biological sampling (Kelley *et al.* 2016). They allow observations of species in their natural environment, with highly accurate vertical and horizontal geo-positioning (Hunt 1997) and the ability to instantly react to the observed environment. Submersibles can also collect targeted specimens in pristine condition, enabling critical shipboard and lab experimentation. They can be outfitted with a variety of sensors for measuring physico-chemical parameters.

Limitations

Manned submersibles come in an array of sizes, personnel capacities, and depth ratings, but all require a trained pilot. Their biggest drawbacks are their high costs, slow speeds (i.e. 1-2 knots), and limited dive times. With rare exceptions (e.g. acrylic spheres), most submersibles are also designed for benthic surveying and manufactured with downward angled viewports that will limit sampling efficiency in the midwater (Hunt 1997). Poor water clarity and dim light present major hurdles, although these can be overcome to some degree by combining acoustic sonars and personal observations. Importantly, researchers must pay close attention to survey designs to obtain abundance estimates from remotely operated vehicles. This is due to the propensity of pilots to 'chase' individuals, which has the potential to bias inferences of population size. Such bias can be avoided by using adaptive sampling protocols and specialised data analysis (see Thompson & Seber 1996). Lastly, sample volumes (i.e. holding capacity) are limited, and usually smaller than with other methods.

2.3.4.2 Unmanned

Unmanned methods encompass autonomous underwater vehicles (AUVs), remotely operated vehicles (ROVs), Argo floats, ocean gliders, and unmanned surface vehicles (USVs). An AUV is a marine craft pre-programmed to conduct a variety of unattended underwater missions without constant supervision or monitoring by a human operator, whereas an ROV requires instructions delivered through a cabling system or via acoustics (Roberts & Sutton 2006). Gliders are autonomous, buoyancy-driven vehicles that oscillate through the water column, relying upon large wings to translate vertical into horizontal motion (Rudnick *et al.* 2004). Wave gliders harvest the abundant energy contained in ocean waves for (nearly limitless) propulsion (Wiggins *et al.* 2010). Cousins to those are Argo profiling floats, which regulate their buoyancy to surface periodically, transmit oceanographic data via satellite, and return to depth (Roemmich *et al.* 2004). The global array of Argo floats consists of more than 3,000 free-drifting units that have been monitoring the upper 2,000 m of the world's oceans for many years, and are now being redesigned to achieve an operational range of up to 4,000 m (Le Reste *et al.* 2016).

<u>*Benefits*</u>

Unmanned methods can complement conventional forms of sampling by providing long-term, fine-resolution coverage of areas that are impractical or too expensive to survey with large vessels (Jech *et al.* 2009, Sousa *et al.* 2016), with no constraints from weather conditions or sea states. For example, ROVs routinely observe large pelagic vertebrates swimming in the mesopelagic and upper bathypelagic zones (**Figure 2.8**) (Smolowitz *et al.* 2015). In fact, serendipitous encounters with previously undocumented species are relatively common on ROVs, making them a valuable platform for catalysing new scientific discoveries (Macreadie *et al.* 2018). Some instruments can remain unattended for several weeks to months, offering an unsurpassed level of autonomy (Suberg et al. 2014). In addition, autonomous underwater vehicles have greatly matured over the past 10 years, with significant advances in the technologies required for reliable deployment, mission control, performance and recovery (Fernandes et al. 2003). Vehicles carrying appropriate sensors can simultaneously monitor a range of physical and biological parameters (Suberg et al. 2014). For instance, modern gliders can be fitted with cameras (Dodge et al. 2018), mobile tracking systems (Clark et al. 2013), or acoustic loggers/echosounders (Klinck et al. 2012, Meyer-Gutbrod et al. 2012, Baumgartner et al. 2013) to monitor megafauna (e.g. sea turtles, whales, sharks, fishes) in virtually real-time and ground-truth other surveys. Regular surface communications via satellite allow the movements of unmanned instruments to be controlled remotely, with near-instant data relays (Suberg et al. 2014). Unmanned infrastructure is also inherently scalable, making coordinated 'fleets' of instruments appealing and cost-efficient for establishing broad-scale ocean monitoring networks (Leonard et al. 2010, Schofield et al. 2010). Such an approach maximises the likelihood of capturing intermittent, localised phenomena (e.g. plankton blooms, upwelling events) (Rudnick et al. 2004). One feature that distinguishes gliders from other unmanned methods is their slow speed, which reduces drag and permits longer-duration operations.

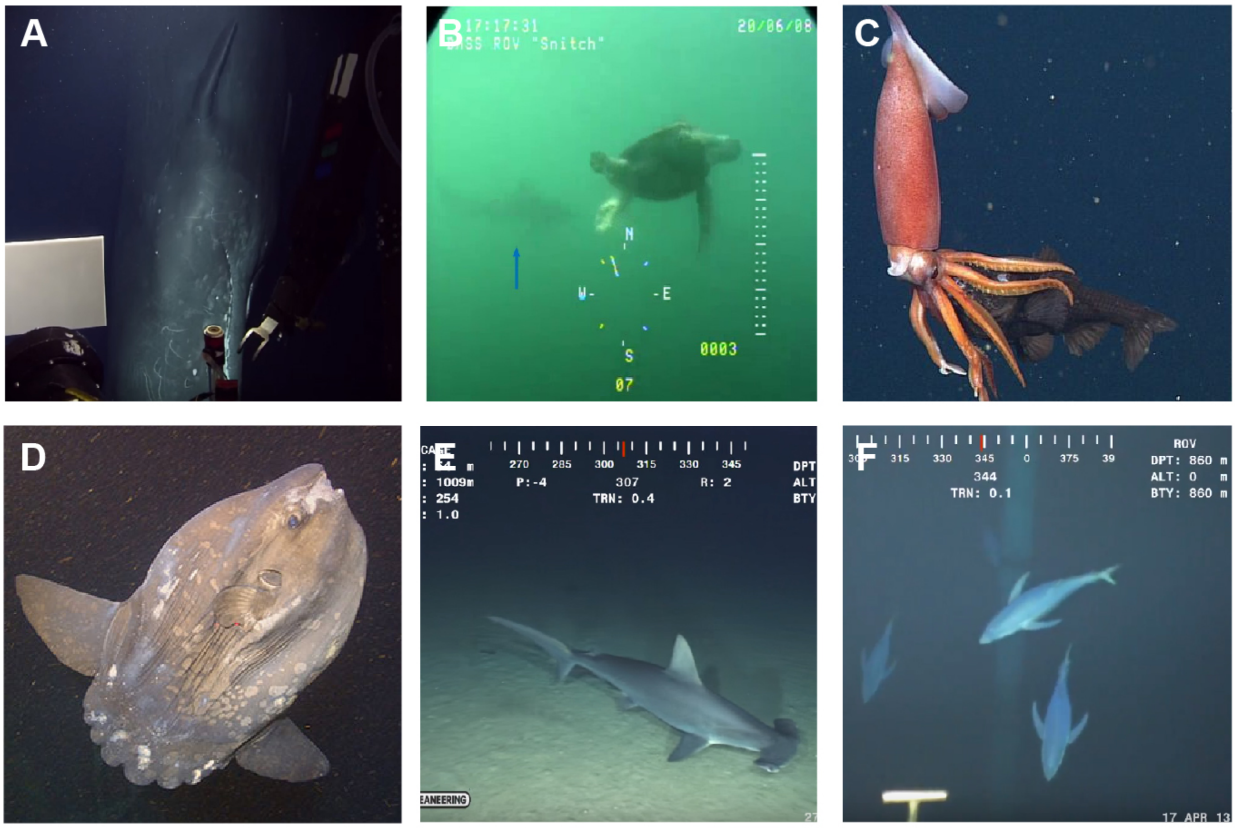

Figure 2.8 Examples of pelagic wildlife observed by remotely operated vehicles (ROVs). **(A)** Sperm whale, *Physeter macrocephalus* (source: E/V Nautilus). **(B)** Loggerhead sea turtle, *Caretta caretta* (reproduced from Smolowitz *et al.* (2015) under a Creative Commons Attribution-NonCommercial-NoDerivatives 4.0 International license CC BY-NC-ND 4.0). **(C)** Pelagic squid**,** *Gonatus sp.* feeding on a bathylagid fish, *Bathylagidae* (reproduced from Choy *et al.* (2017) under a Creative Commons Attribution 4.0 International license CC BY 4.0). **(D)** Ocean sunfish, *Mola mola* (Image courtesy of Deepwater Canyons 2013, NOAA-OER/BOEM/USGS). **(E)** Scalloped hammerhead shark, *Sphyrna lewini* (source: The SERPENT Project). **(F)** Tunas, *Thunnus sp.* (source: The SERPENT Project).

Limitations

Limitations in energy storage, power consumption, and information payloads are a fundamental caveat of, and driver in, the design of unmanned instruments (Fernandes & Brierley 1999, Bellingham & Rajan 2007). Each mission therefore reflects a fine compromise between battery life, sampling duration, sampling frequency, and data quality (Willcox *et al.* 2001). Most vehicles are large and slow-moving, making them prone to drift in areas of strong currents (Davis *et al.* 2009), with potentially significant displacement across dive cycles. Although not all need to surface and send messages after each dive, failure to do so introduces uncertainty in physical/biological measurements and wastes energy expanded towards maintaining buoyancy

(Rudnick *et al.* 2004). Another recurring problem is the need for sensor calibration and data validation (e.g. through net sampling or seawater collection), particularly over long-term surveys (Suberg et al. 2014). Bio-fouling can affect this process, by causing sensor drift in a non-linear fashion that is difficult to reconcile with calibration procedures at the beginning and end of a mission (Suberg et al. 2014). Fixed moorings and profiling floats provide extensive time-series, but the former only collect information in one point location and the latter can be difficult to control spatially. Flight control and the alignment of trajectories from multiple vehicles within a fleet remain an outstanding operational challenge. Instruments carrying passive acoustic sensors are generally too slow moving to use target motion analysis as a means of localising animals in the water column.

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2.3.5 Satellite technologies

2.3.5.1 Satellite remote sensing

Recognition of the utility of making measurements of the world's oceans from space came as early as the 1970s, when polar-orbiting satellites were earmarked as potential tools for estimating productivity in offshore fisheries at large spatial scales (Yentsch 1973). Since then, the rapid development of optical, altimetric, and radiometric satellite sensors with varying resolutions (from sub-metre to >1,000 m) and acquisition frequencies (from sub-hourly to fortnightly, **Table 2.3**) has revolutionised our ability to monitor both coastal and pelagic systems (Andréfouët *et al.* 2008, Platt & Sathyendranath 2008, Horning 2010, Klemas 2012). A maturing array of remote-sensed data products is now available, including gridded maps of sea surface temperature (SST), chlorophyll-a concentration, water turbidity and ocean topography (e.g. sea level anomaly (SLA), sea surface height) (Brando & Dekker 2003, Hu et al. 2004, Polovina & Howell 2005, Chen et al. 2007, Platt et al. 2008). When measured in concert, these variables can offer synoptic snapshots of prominent oceanographic features such as boundary currents, eddies, frontal zones, and upwelling cells (Prata & Wells 1990, Moore *et al.* 2007, Sousa *et al.* 2008, Klemas 2012, Foster *et al.* 2014, Huang & Feng 2015, Huang 2017, Leplastrier & Huang 2017). Such features are often associated with local or regional biodiversity hotspots (e.g. Ward *et al.* 2006, Tetley *et al.* 2008, Gill *et al.* 2011), and provide a blueprint for classifying the pelagic ocean into seascape units useful for monitoring and management (Gohin *et al.* 2008, Hardman-Mountford *et al.* 2008, Costello 2009, Kavanaugh *et al.* 2014, Kavanaugh *et al.* 2016). Remotely-sensed products are also commonly used as inputs to species distribution models for numerous pelagic species, including fishes, reptiles, and mammals (e.g. Oliveira & Stratoudakis 2008, Panigada *et al.* 2008, Valavanis *et al.* 2008, Bouchet *et al.* 2017, Thums *et al.* 2017).

Table 2.3 Commonly used and currently operational satellite platforms for remotely sensing pelagic ecosystems. MODIS: Moderate Resolution Imaging Spectroradiometer. AVHRR: Advanced Very High Resolution Radiometer. SLA: Sea level anomaly. SST: Sea surface temperature. Ocean Colour variables include Chlorophyll-a (Chl-a), total suspended sediment (TSS), coloured dissolved organic matter (CDOM), euphotic depth, K490 (downwelling diffuse attenuation coefficient at 490 nm, i.e. a turbidity parameter), primary production (PP), etc.

Instrument	Sensor type	Key variables	Resolution	Revisit time	Archive length
MODIS Terra-MODIS (AM) Aqua-MODIS (PM)	Optical Radiometry	Ocean colour variables, SST	250 m, 500 m, 1 km	2 x per day	1999-present
AVHRR	Radiometry	SST	~ 1 km	4 x per day	1979-present
Jason-2, 3	Altimetry	SLA	25 km	10 days	2001-present
Himarwari-8	Optical, Radiometry	Chl-a, SST	2 km	10 mins	2014-present
Worldview 2, 3, 4 (Commercial)	Optical	Chl-a, TSS	0.5-2 m	1-10 days	2007-present
Sentinel 2	Optical	Chl-a, TSS	10-60 m	10 days	2015-present
Sentinel 3	Optical, Radiometry, Altimetry	Ocean Colour variables, SST, SLA, TSS	1-2 km	1-2 days	2016-present
Landsat 5, 7, 8	Optical	Ocean colour variables	25 m	~ 16 days	1987-present
Planet (Commercial)	Optical	Maritime monitoring	0.7-5 m	1 day	2009-present

Benefits

A key advantage of satellite remote sensing is that it provides a cost-effective tool for mapping environmental and biophysical data over broad spatial scales, something that remains impractical with *in situ* methods (Andréfouët et al. 2008). The SeaWiFS, MODIS-Aqua, and VIIRS sensors, for instance, have provided an extended time series of worldwide, near-daily ocean colour observations since 1997, offering important information to quantify lower trophic level pelagic dynamics at scales from 1 km to global. The periodic, repeat coverage of satellite-based remote sensing is useful for monitoring change and understanding trends (Geller *et al.* 2017), particularly where sites can be revisited on a regular basis throughout the lifetime of the satellite. The high temporal frequency of some remotely sensed data (e.g. Himarwari-8) can also help capture dynamic, rapidly developing phenomena such as upwelling events, phytoplankton blooms and eddy movements (Nezlin *et al.* 2012, Ramanantsoa *et al.* 2018). Importantly, remote sensing datasets are permanent records that offer repeatable, standardised and verifiable information. Many are (or will soon become) cheap or freely available operational products (Game *et al.* 2009), facilitating access at different scales and resolutions. Single images are taken unobtrusively and can be analysed and interpreted for different purposes, without any permitting or fieldwork requirements.

A commonly used method for obtaining satellite remote-sensed data for the Australian marine estate is via the Integrated Marine Observing System (IMOS, http://imos.org.au/) (Hill *et al.* 2010) and its data retrieval portal, the Australian Ocean Data Network (AODN, https://portal.aodn.org.au/) (Proctor *et al.* 2012). Digital Earth Australia (http://www.ga.gov.au/dea) is expanding its capability in the marine realm, including products useful to intertidal mapping (Sagar *et al.* 2017) and mangrove monitoring (Rogers *et al.* 2017). Remote sensing products, including SST from AVHRR, SLA from satellite altimeters such as Jason 2 and 3 and ocean colour products from Modis Aqua (including Chl a, net primary production and the picoplankton and nanoplankton fractions) can be accessed in near real time via IMOS and AODN. In this way, Australian pelagic ecosystems can be investigated and characterised. For example, IMOS products were used in the study of a recent heat wave that resulted in a number of biological impacts across the Tasman Sea, including disease outbreaks for oysters, above average abalone mortality, and reduced performance of cultured salmon (Oliver *et al.* 2017). In that study, remotely sensed and observed SSTs were combined to produce a time series that showed that the summer of 2015/16 was much warmer than in recent years. The IMOS OceanCurrent gridded SLA product was used to support the attribution of a strengthened East Australian Current as a cause of the heatwave through increased transportation of warm waters into the south Tasman Sea (Oliver *et al.* 2017).

Similarly, the Great Barrier Reef (GBR) Marine Park Authority incorporates satellite remote sensing into their monitoring activities via the eReefs portal (http://www.ereefs.org.au/). Ocean colour products from MODIS Aqua are optimised for the region and automatically feed into annual GBR report card assessments of reef health as part of an ongoing monitoring and management programme for the reef (Queensland Government 2015). Inshore water quality is monitored via the relative concentrations of both remotely sensed Chl-a and total suspended sediment, and is compared against water quality guidelines (Great Barrier Reef Marine Park Authority, 2010). The eReefs research project demonstrates how remote sensing can be incorporated into marine monitoring and management programs, such as those required by Australia's extensive network of AMPs.

Limitations

Satellite remote sensing of pelagic habitats faces a number of unique challenges. The first is that open ocean environments are profoundly dynamic, with significant changes occurring in sub-daily time steps (e.g. tides, mobile oceanographic features, diel migrations) that cannot be adequately captured by spaceborne instruments. Second, remote sensing observations are generally constrained to surface conditions, disregarding the biophysical interactions and processes that take place throughout the entire water column. Gaps in spectral time-series are also common (e.g. owing to cloud cover), and may complicate image processing and subsequent analyses. In tropical regions, for example, high levels of reflectance and cloud formation during the wet season often result in seasonally biased datasets due to the limited accessibility of glare-free and cloud-free imagery. Third, pelagic species can be highly mobile and may respond to physical ocean conditions with varying time lags, meaning that the time scales required to characterise their distribution and population dynamics must match those of the data collection cycles (Mannocci *et al.* 2017). Although species such as sea turtles, sharks and cetaceans can be directly observed using very high-resolution sensors (<1 m), such data are expensive and often impractical to collect, particularly over large areas (Geller *et al.* 2017). Fourth, while their outputs are inexpensive, the costs and labour requirements associated with developing, building, testing and launching remote sensing satellites (with risks ensuing from single point failures) are prohibitive. It is also noteworthy that: (i) the sun-synchronous nature of orbiting spectral satellites such as MODIS and Landsat means that data are only acquired during daylight hours, and (ii) contrary to shallow coastal habitats, the deep ocean cannot be readily mapped using space-based optical systems, instead requiring submerged active sensors such as multi-beam sonar, buoy-based instrumentation, or gliders.

Importantly, remote sensing instruments should be routinely calibrated and validated against *in situ* measurements to ensure that they are working to their maximum potential and that they maintain satisfactory accuracy (e.g. minimal classification errors). The Bluelink (http://wp.csiro.au/bluelink/) and OceanMAPS (http://wp.csiro.au/bluelink/global/oceanmaps/) collaborations, variously between the CSIRO, Bureau of Meteorology and the Royal Australian Navy, are good illustrations of how these data types can be merged to complement and value-add to one another. OceanMAPS leverages the Bluelink data assimilation system to offer an ocean forecasting service that combines ocean modelling with field-based SST and salinity measurements from floats and moorings with remote sensing altimetry and SST inputs (Brassington et al. 2012). The result is a near-real time ocean forecasting system for Australia that offers global, regional and littoral insights into mesoscale circulation and dynamics.

Lastly, users of remote sensing data should be mindful of the corrections and algorithms applied to data products. In particular, atmospheric corrections are subject to continual improvements to counter the effects of unwanted radiance on detected signals. Around 90% of the detected top-of-atmosphere signal in marine environments is corrupted by atmospheric (e.g. aerosols and gases) and water surface (e.g. sun and sky glint) interferences (Emberton *et al.* 2016). These effects must be accounted for to determine the desired water-leaving radiance. The effects of aerosols are generally lower over the open ocean than over coastal and inland waters, with atmospheric corrections for the former considered to be robust (IOCCG 2010). Further research, however, is needed to quantify these effects in turbid or strongly absorbing environments (Fan *et al.* 2017).

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2.3.5.2 Satellite photography

Although satellite remote sensing is primarily used in habitat monitoring applications, the direct detection of marine animals from space has the potential to inform MPA management in many ocean areas (Pettorelli *et al.* 2016). A number of recent studies have successfully identified and censused megavertebrates such as penguins (Fretwell & Trathan 2009, Fretwell *et al.* 2012, Lynch *et al.* 2012), seals (LaRue *et al.* 2011, McMahon *et al.* 2014), polar bears (Stapleton *et al.* 2014), and whales (Platonov *et al.* 2013, Fretwell *et al.* 2014) based on commercial, high-resolution imagery.

Benefits

The use of satellite images presents several advantages compared to traditional survey methods: it removes any effects ensuing from observer presence, and allows the observation of vast geographic areas that would otherwise be extremely difficult to survey due to their size, remoteness, and often treacherous terrain/conditions (McMahon *et al.* 2014). Over the last decade, the spatio-temporal accuracy of satellite products has also drastically improved, and the costs of acquiring imagery has considerably decreased. The Worldview2 satellite, for example, boasts an on-the-ground pixel size of 50 cm in the panchromatic and 2 m in its eight colour spectral bands (Fretwell *et al.* 2014). While most of the highest-resolution datasets cannot yet be accessed freely, an increasing number of private companies are making subsets of their image banks available for research purposes at no cost (e.g. DigitalGlobe Foundation, http://foundation.digitalglobe.com/application/). Even though manual processing is labour-intensive, time-consuming, and subject to considerable error (Laliberte & Ripple 2003), the development of data-mining algorithms holds promise for automating image classification and pattern recognition (Vukelic et al. 2018).

Limitations

First and foremost, the use of satellite imagery is constrained by the weather. Under ideal conditions, high-resolution photographs are capable of discerning individuals or even differentiating among groups of sympatric species (Lynch *et al.* 2012). Pervasive cloud cover and high Beaufort Sea states, however, can easily complicate image interpretation and compromise object detection (LaRue *et al.* 2011). Subsurface rocks in shallow areas, seabird flocks, surface bubbles, vessels, and behavioural displays (e.g. tail slapping, rolling, or blowing in baleen whales) present further obstacles to correct classification (Fretwell *et al.* 2014). Counts of land-associated taxa such as pinnipeds may also be precluded when shadows are produced by low sun angles over complex terrains. In this context, image interpretation of unknown locations should be approached with caution, particularly if local topographic heterogeneity is high and/or analysts inexperienced. Even when cloud-free snapshots are obtained, visibility will be contingent upon successfully matching the timing of the imagery to the phenology of the target species (Lynch *et al.* 2012). This is complicated by the complex diving behaviour of many pelagic animals, which spend significant lengths of time underwater where they are effectively unavailable for detection (Rogers *et al.* 2013). One critical yet unresolved factor is thus to quantify the water column penetration properties of different satellite sensors and how these vary as a function of turbidity and surface roughness, two properties that typically change over short time spans and even spatially within a single image (Fretwell *et al.* 2014). An additional challenge lies in determining whether image counts are commensurate with true population sizes. This is commonly assumed to be the case, but can only be verified by synchronous ground-truthing, particularly as detectability is influenced by animal density (Lynch *et al.* 2012). Finally, significant progress has been made towards building automated tools for image processing, but these require further testing across a range of taxonomic groups and sea conditions (Fretwell *et al.* 2014).

2.3.5.3 Satellite biotelemetry

This is a golden age for species movement studies (Hays *et al.* 2016). The rapid sophistication and miniaturisation of animal-borne satellite tags in the last 30 years has transformed our ability to document the behaviour of marine organisms over previously unimaginable spatio-temporal scales (Priede & French 1991, Hart & Hyrenbach 2009). For instance, with the compilation of global satellite telemetry datasets (e.g. Tagging of Pacific Predators, TOPP http://gtopp.org/;

Marine Megafauna Movement Analytical Program, MMMAP https://mmmap.wordpress.com/; Global Procellariiform Tracking Database GPTD http://www.seabirdtracking.org/) have come unprecedented ecological insights into critical habitats, migration pathways, drivers of fine and large-scale movements, linkages between ocean features and multispecies hotspots, patterns of species niche partitioning, and spatial overlap between species' distributions and political boundaries, as well as human threats (Block *et al.* 2011, Le Corre *et al.* 2012, Lascelles *et al.* 2016, Queiroz *et al.* 2016, Rodríguez *et al.* 2017, Sequeira *et al.* 2018). An increasing body of literature also demonstrates the value of satellite telemetry data in supporting the assessment and monitoring of pelagic MPAs (Maxwell *et al.* 2011, Scott *et al.* 2012, Young *et al.* 2015a, Maxwell *et al.* 2016, White *et al.* 2017) (**Figure 2.9**). Continued demand for tracking devices has thus resulted in the development of a variety of biologging technologies (Thomas *et al.* 2012), from transmitters that relay data to orbiting satellites on a near-continuous basis (e.g. ARGOS, Fastloc GPS) to self-contained devices that log and archive data internally (Priede & French 1991, Cooke 2004).

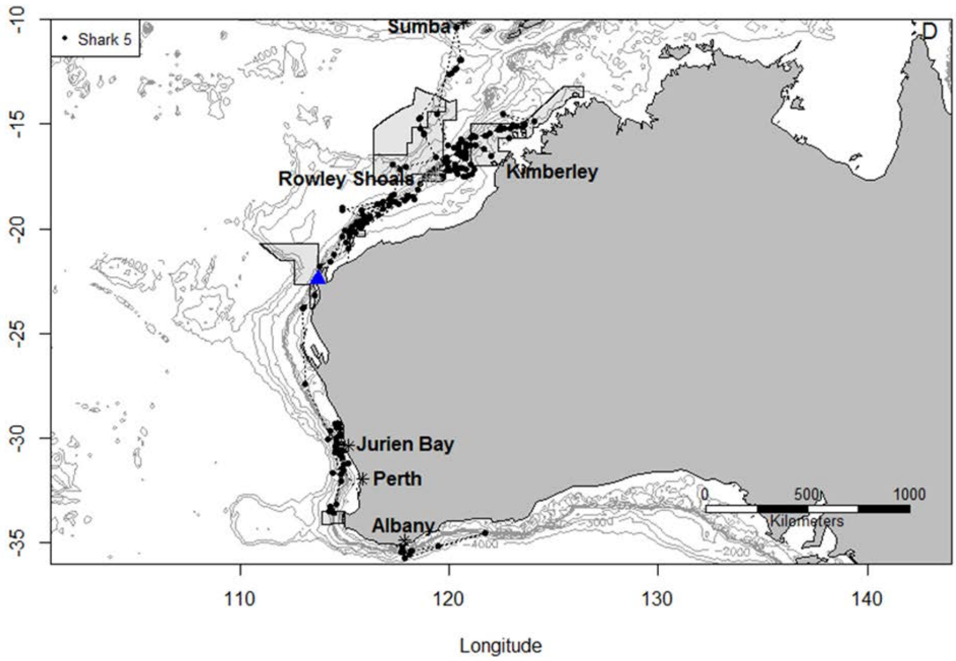

Figure 2.9 Location uplinks received from a tiger shark (*Galeocerdo cuvier*) satellite tagged at Ningaloo Reef, northwestern Western Australia (blue triangle). Grey polygons indicate Commonwealth Marine Reserves. Reproduced from Ferreira *et al.* (2015) under a Creative Commons Attribution 4.0 International license (CC BY 4.0).

__Benefits__

Satellite tags provide a wealth of individual at-sea location information that can be downloaded remotely without the need to recover the tag. The advent of new generations of GPS methods (TrackTag, Fastloc) have catalysed considerable improvements (i.e. 10 to 100-fold) in data precision compared with traditional ARGOS systems, opening up exciting possibilities to track micro-scale patterns of space use (Rutz & Hays 2009). Some instruments can also function for multiple years, enabling the study of animal movements and migrations in real time over prolonged, uninterrupted periods. For instance, geolocation tags that infer latitude/longitude from changes in ambient light levels benefit from minimal power consumption and can last for up to a decade (although this comes at the expense of spatial resolution) (Burger & Shaffer 2008). Externally or internally-mounted temperature, speed, acoustic, and/or pressure sensors offer a wide spectrum of additional functionalities, including the monitoring of numerous physiological and behavioural variables (e.g. energy expenditure, activity budgets, rare behavioural events like prey captures).

Limitations

Above all else, biotelemetry methods are costly and logistically challenging to implement, with usually only a small number of tag units available for any given project. For instance, fees to the ARGOS delivery system can exceed $20 per day per tag, putting a burden on studies that demand high replication. Issues surrounding less-than-ideal sample sizes are therefore widespread, and can constrain the ability to obtain statistically meaningful and biologically relevant results (Ropert-Coudert & Wilson 2005). As a rule, the species most amenable to biotelemetry methods are large-bodied, accessible and easy to capture (e.g. land-breeders), available in appreciable numbers, and of lesser conservation concern (Ropert-Coudert *et al.* 2009).

Importantly, the process of tag attachment is usually invasive, and can result in both physical injuries and short- and long-term physiological impacts. Negative effects may include tissue degradation, bacterial infection, higher parasitic loading, changes in movement behaviour and diving efficiency, increased energetic demands from hydrodynamic drag, heightened stress levels, reduced growth and survival of offspring, reduced colony attendance, higher mortality rates, and lower foraging success where instrument specifications (e.g. colour) affect predator-prey interactions (Hawkins, 2004). These effects, however, can vary as a function of tag size, shape, and placement along the animals' bodies. Tag fitting protocols must also be tailored to the target species (e.g. glue on fur in seals, muscle puncture in sharks). The risk of data loss is another important caveat, as cessations of signal transmission commonly occur following physical damage (e.g. antenna breakage), bio-fouling, battery exhaustion, salt-water switch failure, animal mortality, and premature tag detachment (Hays et al. 2007). Despite emergent synergies in the way different taxa cross ocean basins (Sequeira *et al.* 2018), movement data can also show a staggering amount of heterogeneity among individuals, seasons, and populations, which complicates statistical inference (Rutz & Hays 2009).

Another long-standing Achilles' heel of satellite biotelemetry methods is the minimum time required to synchronise transmissions with satellite overpasses, to ensure successful data uplinks. Many air-breathing vertebrates (e.g. whales, turtles) only spend brief time intervals at the surface, which can be insufficient to generate and relay locations. Pop-up archival tags that do not require long-distance signal reception have been developed for animals that stay continuously submerged (e.g. fishes, sharks), yet must be physically retrieved (Whitmore et al. 2016). Lastly, several sources of errors can influence positional accuracy (Hammerschlag et al. 2011), and erroneous inferences can be made when focus is placed on small spatial scales relative to satellite location error, resulting, for instance, in protective boundaries such as MPAs being placed ineffectually.

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2.3.6 Marine genomics

2.3.6.1 Biopsy (tissue) sampling

Remote biopsy techniques have been routinely used to collect tissue samples from free-ranging marine animals non-lethally, and are particularly prominent in cetacean research (Noren & Mocklin 2012) (**Figure 2.10**). Such samples contain a wide variety of invaluable physiological information, and can provide insights into prey preferences, foraging ecology, contaminant loads, hormone balance, gene expression, disease state, and numerous other metabolic processes (Hunt *et al.* 2013). Emerging close-kin mark-recapture techniques capitalise on advances in molecular genetics to affordably and reliably identify parent-offspring pairs, and have been successfully applied to gain absolute abundance estimates for white shark (Hillary *et al.* 2018) and tuna (Bravington *et al.* 2016) populations from tissue samples. More generally, biopsies are also increasingly recommended as an alternative to capture sampling for threatened populations of marine fishes and sharks (Hussey *et al.* 2012) (but see Heupel & Simpfendorfer 2010 for a contrasting viewpoint). For example, successful efforts have been made to develop fish muscle punch techniques either akin to those employed in marine mammal studies (Bridges *et al.* 2001, Kinney *et al.* 2011) or modified to function underwater (Evans 2008).

Figure 2.10 Bird's eye-view of remote dart biopsy sampling of Australian snubfin dolphins (*Orcaella* heinsohni) in Cygnet Bay, Western Australia. Photo credits: Marine Quintin, Murdoch University Cetacean Research Unit.

<u>*Benefits*</u>

In most cases, biopsy sampling guarantees the acquisition of sufficient high-quality genetic material for reliable nuclear and mitochondrial analyses (Krützen *et al.* 2002). Although boat and personnel costs per sample remain typically high, sample processing can prove substantially cheaper and less time-consuming than striving to extract genetic material from alternative sources of poorer quality (e.g. scats) (Parsons *et al.* 2003). Costs can be further reduced for species of commercial or recreational interest through direct access to specimens landed at the docks (Evers *et al.* 2008). When obtained in tandem with visual identification techniques, biopsy samples can also be linked to individual-based data, giving insights into social structure, individual space use, behavioural dynamics and population demography (Tezanos-Pinto & Baker 2012). Biopsying effort is often opportunistic, and can be coupled with other types of surveys.

Limitations

A strong disadvantage of biopsying is that it is invasive (i.e. results in physical lesions, and for some species like fishes, requires capture, handling, and sometimes anaesthesia/surgery), which restricts sampling to the size and age classes that can be ethically targeted under existing permitting restrictions (Kellar *et al.* 2013). Despite limited investigations of post-sampling recovery, the wounds incurred from biopsies appear to be generally minor in whales and dolphins, with reports of rapid healing (Krützen *et al.* 2002) and mild behavioural responses of short duration (Kowarski *et al.* 2014). However, at least one mortality event following dart penetration has been previously documented (Bearzi 2000). Some evidence exists that behavioural biases can be introduced where differential individual behaviours lead to disproportionate sampling of say, either males or females (Kellar *et al.* 2013). As a rule, success in acquiring biopsy samples will be a function of: (i) the experience and training of the research team (e.g. competency in archery/shooting and boat handling); (ii) the shooting range and angle of impact; (iii) the deployment method (e.g. power setting on delivery device); and (iv) animal behaviour (e.g. swimming speed, activity state) (Noren & Mocklin 2012). These factors may vary between species, populations, and study sites.

Selected topical reviews

- Hunt KE, Moore MJ, Rolland RM, Kellar NM, Hall AJ, Kershaw J, Raverty SA, Davis CE, Yeates LC and Fauquier DA (2013). Overcoming the challenges of studying conservation physiology in large whales: A review of available methods. *Conservation Physiology*, 1(1): cot006.
- Noren DP, Mocklin JA (2012). Review of cetacean biopsy techniques: Factors contributing to successful sample collection and physiological and behavioral impacts. *Marine Mammal Science,* 28(1): 154-199.

2.3.6.2 Environmental DNA (eDNA)

Marine monitoring is heavily reliant on accurate taxonomic identification, yet most traditional approaches to this either demand increasingly rare specialist knowledge (Wheeler *et al.* 2004), or protocols that inflict damage or disturbance to species of interest, encroaching on animal welfare (Rees *et al.* 2014). The discovery that organisms can be studied non-invasively by retrieving the genetic material they release naturally in the environment has thus been a major scientific breakthrough with immense potential for the conservation of biological diversity (Goldberg *et al.* 2015, Thomsen & Willerslev 2015b, Thomsen & Willerslev 2015a) (though see Will *et al.* 2005 for some counter points). Molecules derived from skin, saliva, mucous, hairs, sperm, eggs, faeces, urine, blood, root, leaves, fruit, pollen, and rotting bodies are collectively referred to as environmental DNA (eDNA) (Bohmann *et al.* 2014). Interest in their analysis has recently skyrocketed and eDNA approaches are now being applied to the monitoring of numerous ecosystems, including aquatic (Thomsen *et al.* 2012a, Thomsen *et al.* 2012b, Pilliod *et al.* 2013, Boussarie *et al.* 2018).

Benefits

eDNA presents considerable advantages for marine monitoring (Thomsen *et al.* 2012a), in great part due to its ability to profile entire biological communities with only a single standardised sample (Port *et al.* 2016, Yamamoto *et al.* 2017). Indeed, rapid advances in high-throughput next-generation sequencing have made comprehensive biodiversity surveys possible with minimal effort and expense (Thomsen & Willerslev 2015b). This renders eDNA a potent tool for elucidating mechanistic, ecosystem-wide, and evolutionary processes (Bohmann *et al.* 2014), as well as supporting biogeographic studies at the population level (Sigsgaard *et al.* 2017). A further advantage is that eDNA techniques are not impacted by phenotypic plasticity, allowing the successful differentiation of sister species that would otherwise be too morphologically similar to be confidently told apart (Johnson *et al.* 2017). High detection accuracy and sensitivity make eDNA a valuable complement to conventional monitoring methods (Baker *et al.* 2018) (**Figure 2.11**), allowing improved surveillance of cryptic (Baker *et al.* 2018, Boussarie *et al.* 2018), introduced/invasive (Kim *et al.* 2018), or vulnerable and endangered species at low densities (Kelly *et al.* 2014a, Weltz *et al.* 2017). This is particularly true where rates of false absences can be minimised by using multiple samples from the same locality, multiple extractions per sample and/or multiple polymerase chain reactions per extraction) (Ficetola *et al.* 2015).

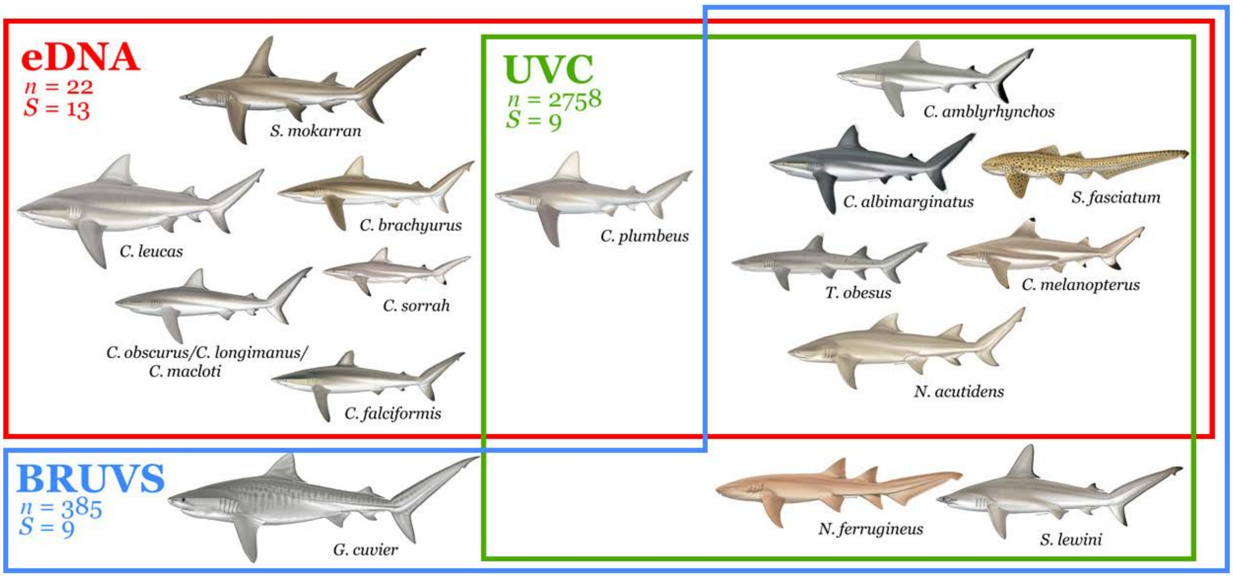

Figure 2.11 Venn diagram showing the species detected by different sampling methods, namely eDNA (n = 22 samples, S = 13 species), underwater visual census (UVC, n = 2758 samples, S = 9 species), and baited remote underwater video systems (BRUVS, n = 385 samples, S = 9 species). Scientific drawings courtesy of M. Dando. Figure adapted from Boussarie *et al.* (2018) under a Creative Commons Attribution NonCommercial License 4.0 (CC BY-NC).

Limitations

The transport, dispersion, and dilution of free-floating eDNA fragments within dynamic marine ecosystems remains a significant obstacle to making accurate and defensible inferences about the distribution of target organisms using eDNA techniques (Thomsen *et al.* 2016). For instance, evidence exists that the strongest eDNA signals may only be found in samples taken at sites situated in close proximity (i.e. a few metres) to where the animals were (Foote *et al.* 2012). This issue is compounded by the fact that decay rates can vary by several orders of magnitude (from hours to weeks) (Thomsen *et al.* 2012a), and may be affected by a myriad of biological and physico-chemical factors including microbial activity, pH, oxygen concentration, and temperature (Barnes *et al.* 2014, Strickler *et al.* 2015, Salter 2018). Incorporating models of environmental dynamics and eDNA degradation into genetic surveys is thus a critical step toward independent validation (Kelly *et al.* 2014b).

Additional uncertainties around shedding rates (Sassoubre *et al.* 2016) and DNA provenance complicate the assessment of false positive/false negatives. For example, it is impossible to distinguish between live animals and dead specimens leaking material into the water column. Mobile pelagic predators such as fishes, seabirds or mammals may also re-distribute DNA from prey items across sites through excretory processes (e.g. defecation) (Thomsen *et al.* 2012a), biasing analytical results. From a practical perspective, the high salinity of seawater may render DNA extraction inefficient by inhibiting sequence amplification by polymerase chain reaction. Current eDNA technologies also consist mainly of do-it-yourself solutions, and the lack of purpose-built sampling equipment limits the efficiency and harmonisation of eDNA studies (though this is changing, see Thomas *et al.* 2018). Another pitfall of the eDNA approach is the failure of short primers to resolve some taxonomic groups with sufficient resolution, hindering the biological interpretation of field data (Thomsen *et al.* 2016). Lastly, despite increasing support for the idea that eDNA concentration can be positively correlated with ‘true’ population biomass or abundance, evidence for the generality of this principle at different spatio-temporal scales remains elusive (Takahara *et al.* 2012, Kelly *et al.* 2014a, Thomsen *et al.* 2016, Yamamoto *et al.* 2016). Assuming that eDNA offers a reasonable representation of animal density in an area, this relationship is not necessarily retained in the final data, as the number of sequences recovered per taxon is a function of the original (yet unknown) number of sequences in the water. The latter can be expected to be taxon-, age class-, season-, and site-specific. The former will be influenced by heterogeneity in (1) the volume of the water samples, (2) the amount of eDNA retained on the filters, (3) the efficiency of the subsequent DNA extraction process, (4) the affinity of the chosen primers for each taxon present, and (5) any errors arising during amplification or sequencing.

Selected topical reviews

- Bourlat SJ, Borja A, Gilbert J, Taylor MI, Davies N, Weisberg SB, Griffith JF, Lettieri T, Field D, Benzie J, Glöckner FO, Rodríguez-Ezpeleta N, Faith DP, Bean TP, Obst M (2013). Genomics in marine monitoring: New opportunities for assessing marine health status. *Marine Pollution Bulletin*, 74: 19-31.

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- Deiner K, Bik HM, Mächler E, Seymour M, Lacoursière-Roussel A, Altermatt F, Creer S, Bista I, Lodge DM, de Vere N, Pfrender ME, Bernatchez L (2017). Environmental DNA metabarcoding: Transforming how we survey animal and plant communities. *Molecular Ecology*, 26(21): 5872-5895.
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- Rees HC, Maddison BC, Middleditch DJ, Patmore JRM, Gough KC (2014). The detection of aquatic animal species using environmental DNA - A review of eDNA as a survey tool in ecology. *Journal of Applied Ecology*, 51: 1450-1459.
- Thomsen PF, Willerslev E (2015). Environmental DNA - An emerging tool in conservation for monitoring past and present biodiversity. *Biological Conservation*, 183: 4-18.
- Trivedi S, Aloufi AA, Ansari AA, Ghosh SK (2016). Role of DNA barcoding in marine biodiversity assessment and conservation: An update. *Saudi Journal of Biological Sciences*, 23(2): 161-171.

2.3.7 Participatory methods

The fields of citizen science and crowdsourcing have been burgeoning rapidly, with thousands of non-specialist volunteers now actively contributing to, and participating in, evidence-based marine research projects designed to underpin policy (Thiel *et al.* 2014, Hyder *et al.* 2015) and support responses to time-sensitive conservation problems (Scyphers *et al.* 2015). Insights into ocean ecosystems are also increasingly being gained using 'conservation culturomics', a new branch of social science that extracts biological knowledge from user-generated content shared online across various social media platforms (Di Minin *et al.* 2015).

<u>*Benefits*</u>

A major benefit of participatory methods is their cost-effectiveness (Goffredo *et al.* 2010). Citizen science initiatives indeed offer opportunities to collect extensive datasets on the large-scale distribution and temporal dynamics of endangered species, fisheries resources, harmful species, litter, and pollution, all at minimal expense (e.g. free labour and fund raising). This is, in many ways, a win-win situation, where scientists improve the spatio-temporal scope of their studies without sacrificing local context, and stakeholders and participants gain new skills, leading to better acceptance of the information they helped collate themselves (Starr 2010). Low-cost programmes can also be sustained over the long-term using local expertise and financing, promoting public engagement, and maximising the likelihood of detecting environmental and biological change (e.g. species invasions) (Goffredo *et al.* 2010). In Australia, for example, the Range Extension Database and Mapping (Redmap, http://www.redmap.org.au/) project started as a Tasmanian initiative designed to document observations of uncommon marine species (Pecl *et al.* 2014). It is now a national programme with data used in numerous studies on climate change and range shifts (Last *et al.* 2011, Lenanton *et al.* 2017). Similarly, the Reef Life Survey (https://reeflifesurvey.com/) involves an extensive international network of highly trained volunteers and professional biologists. The programme's global coverage has facilitated analyses of patterns in reef fish functional diversity around the globe and has supported the most comprehensive empirical assessment of key features for successful MPA design and management (Edgar & Stuart-Smith 2014).

Limitations

Overall, citizen science activities tend to be concentrated around populated centres and easily accessible habitats in close proximity to shore, leaving pervasive gaps in oceanic waters (Thiel *et al.* 2014). Since most volunteers have variable educational backgrounds and lack formal training (Goffredo *et al.* 2010), a common perception is that data collected by non-professionals are of fundamentally low quality, and must only be used where no other alternatives exist. As a rule, even simple methodologies must be adapted to the capacities of participants, with training effort typically rising with the complexity of the required tasks. Asking volunteers to self-fund their travels to survey sites and adhere to strict protocols may help achieve more uniform data collection but risks making research projects less attractive, thereby compromising participation rates and ultimately sample sizes. Additional issues relate to species misidentifications, rounding errors, independent validation, and biased (and/or unquantified) sampling effort (Vianna *et al.* 2014, Ward *et al.* 2015).

Selected topical reviews

- Di Minin E, Tenkanen H, Toivonen T (2015). Prospects and challenges for social media data in conservation science. *Frontiers in Environmental Science*, 3: art63.
- Hyder K, Townhill B, Anderson LG, Delany J, Pinnegar JK (2015). Can citizen science contribute to the evidence-base that underpins marine policy? *Marine Policy*, 59: 112-120.
- Thiel M, Penna-Díaz MA, Luna-Jorquera G, Salas S, Sellanes J, Stotz W (2014). Citizen scientists and marine research: Volunteer participants, their contributions, and projection for the future. *Oceanography and Marine Biology: An Annual Review*, 52: 257-314.

3. User perceptions of pelagic sampling methods

Participatory approaches designed to integrate the perspectives of a diverse array of end-users, stakeholders, and researchers, are key to building a collective understanding of the current state of scientific knowledge, and leveraging capacity across various sectors to maximise knowledge grain and scientific progress (Fox *et al.* 2012). With this in mind, an online questionnaire on users' experience with, and perceptions of, a wide range of pelagic sampling equipment and methods was administered on SurveyMonkey. The main aim was to better understand which method(s), if any, were preferentially used and why (Caldwell *et al.* 2016). We sought input from a diverse array of scientists, technicians, undergraduate and postgraduate students, environmental consultants, engineers and managers.

3.1 Online questionnaire

The questionnaire was released on July 12, 2017 (https://www.surveymonkey.com/r/nespd2-pelagic) and run for a total of 50 days, under approval from CSIRO's Social Science Human Research Ethics Committee (077/17) and in accordance with the National Statement on Ethical Conduct in Human Research (2007). All NESP researchers were invited to contribute. Additional respondents from an array of relevant academic, governmental, non-governmental and industry organisations worldwide (Caldwell *et al.* 2016) were solicited by promoting the questionnaire across professional listserves, online fora, conference events, and social media channels, including:

- The Australian Marine Science Association (AMSA) (https://www.amsa.asn.au/) weekly e-news bulletin;
- Printed flyers distributed at the annual conference of the Australian Society for Fish Biology (ASFB) (http://www.asfb.org.au/) in Albany, WA (22-24 July, 2017);
- The University of Western Australia's Oceans Institute (http://www.oceans.uwa.edu.au/) weekly newsletter;
- The University of Victoria's MARMAM (https://lists.uvic.ca/mailman/listinfo/marmam) mailing list;
- The University of Queensland's Spatial.Ecology (http://lists.science.uq.edu.au/mailman/listinfo/spatial.ecology) mailing list;
- The Society for Conservation Biology (SCB) Marine Section (https://conbio.org/groups/sections/marine) mailing list;
- The International Network of Next-Generation Ecologists (INNGE) (http://innge.net/) mailing list;
- The University of Florida Sea Turtle Biology and Conservation (CTURTLE) (https://lists.ufl.edu/cgi-bin/wa?A0=CTURTLE) mailing list;
- The Marine Ecosystems and Management (MEAM) (https://meam.openchannels.org/) info service on ocean planning and ecosystem-based management;

- The European Cetacean Society (ECS) (http://www.europeancetaceansociety.eu/) mailing list;
- The University of Bangor Marine Biology (MARBIO) (https://listserv.bangor.ac.uk/mailman/listinfo/marbio) mailing list;
- The University of Konstanz Animal Movement (animove) (http://animove.org/) mailing list;
- NOAA's Coral Reef Conservation Program's (Coral) (http://coral.aoml.noaa.gov/mailman/listinfo/coral-list) mailing list;
- Colleagues of PJB at the WA Department of Biodiversity, Conservation and Attractions (https://www.dbca.wa.gov.au/) (DBCA, formerly Department of Parks and Wildlife);
- Numerous posts on social media (e.g. Twitter).

Participants were also encouraged to share the survey through their own networks to help reach as large a section of the international scientific community as possible.

The survey consisted of 42 questions designed to capture information relating to respondents':

- Identity and particulars (e.g. country, job affiliation, career stage);
- Primary sampling locations;
- Awareness of various pelagic methods;
- Frequency and history of use of various pelagic methods;
- Main objectives in delivering monitoring programmes;
- Opinions on what they saw as advantages and drawbacks of various pelagic methods;
- Patterns of platform ownership and operation;
- Individual perceptions of costs, risks, and technical requirements;
- Drivers of method choice;
- Opinion on the need for, and value of, developing of standard operating procedures to monitor pelagic environments on a national scale.

The survey was built using advanced piping and branching features, such that some questions were automatically bypassed or updated (where relevant) as a function of answers provided on previous pages. For instance, out of an initial list of 47 pelagic methods, only a maximum of five which respondents identified as using/having used the most were allowed to be carried forward as answer options throughout the survey. A full copy of the questions is included in Appendix 1.

3.2 Summary of responses

The median survey completion time was 23 minutes. Ten respondents exited the survey within two minutes of commencement and were excluded from further analysis. With one exception, these people either had no prior experience in marine sampling or, conversely had spent a large amount of time in the field (20+ expeditions, Appendix 1). Only a small number of questions were made compulsory. While this granted participants some level of flexibility and allowed them to skip any aspects of the survey they felt uncomfortable with or unqualified to address, it also means that response rates typically varied per question. The results presented below must therefore be interpreted accordingly (i.e. all rates are calculated based on the number of responses received for each specific question).

3.2.1 Demographics

Sixty-two individuals from 16 countries completed the survey in full. The majority of respondents were senior/project scientists (n=17, 27%), followed by PhD students (n=16, 26%), consultants (n=8, 13%) and postdoctoral researchers (n=6, 10%). Half (n=35) of respondents were affiliated with a university institution, with the rest working in the government sector (n=10, 16%), private companies (n=8, 13%) and non-profit organisations (n=8, 13%). Response rates were greatest in Australia (n=29, 47%), followed by the United States (n=10, 16%) and the United Kingdom (n=6, 10%), though several individuals from Africa, Asia and Europe also contributed (**Figure 3.1**). Respondents generally showed high levels of familiarity with marine sampling, with 68%

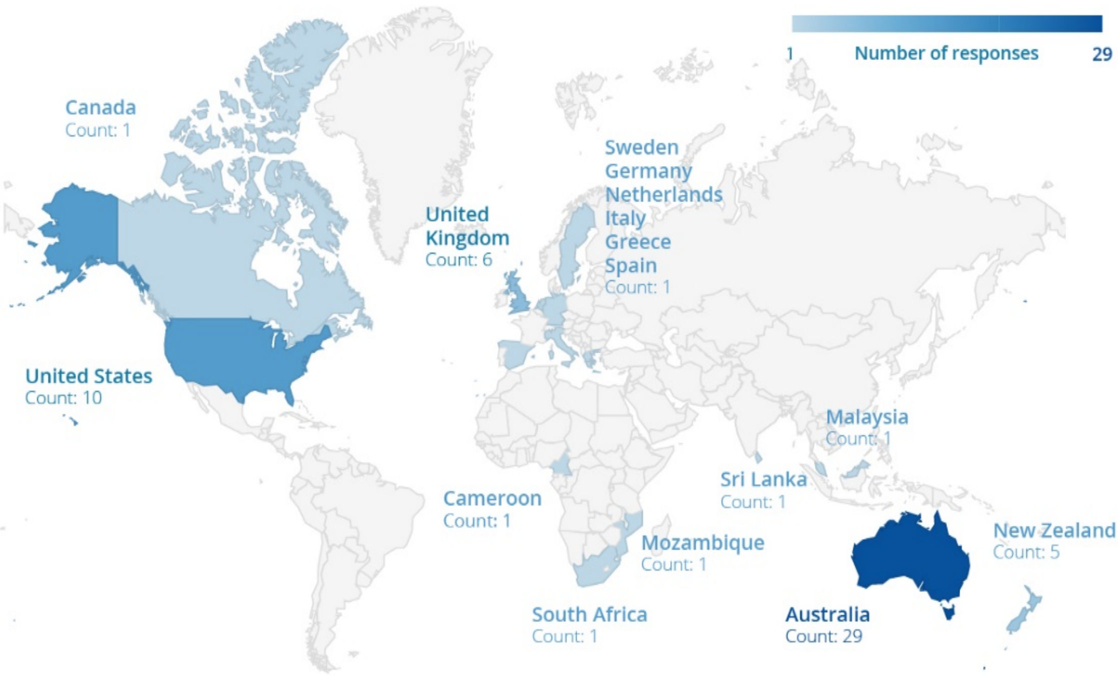

Figure 3.1 Worldwide distribution of responses received in the NESP D2 online pelagic questionnaire.

(n=42) having undertaken at least five field expeditions, and more than a third (n=22, 35%) having completed 20 or more.

3.2.2 Fieldwork

Close to half (n=29, 47%) of respondents said the bulk of their field activities took place overseas, whilst another 31% (n=19) worked mostly in Australia, and 23% (n=14) did both. Respondents' affiliations were generally good indicators of the geographical extent of their sampling work, insofar as a large proportion of Australians conducted monitoring in Australian waters (n=17, 59%) and the majority of overseas respondents focused their efforts on overseas sites (n=27, 82%). Two respondents based in New Zealand and the United States indicated they worked solely in Australia.

Marine fieldwork spanned all environments. Most participants (n=63, 71%) usually sampled multiple habitats, with a preference towards shallower bathomes (coastal and intertidal, n=38, 61%; shelf <50m, n=33, 53%; shelf 50-200m, n=30, 48%). In Australia, the North-west, South-west and Temperate east bioregions (**Figure 3.2**) were best represented, with 53% (n=17), 38% (n=12) and 38% (n=12) of respondents respectively.

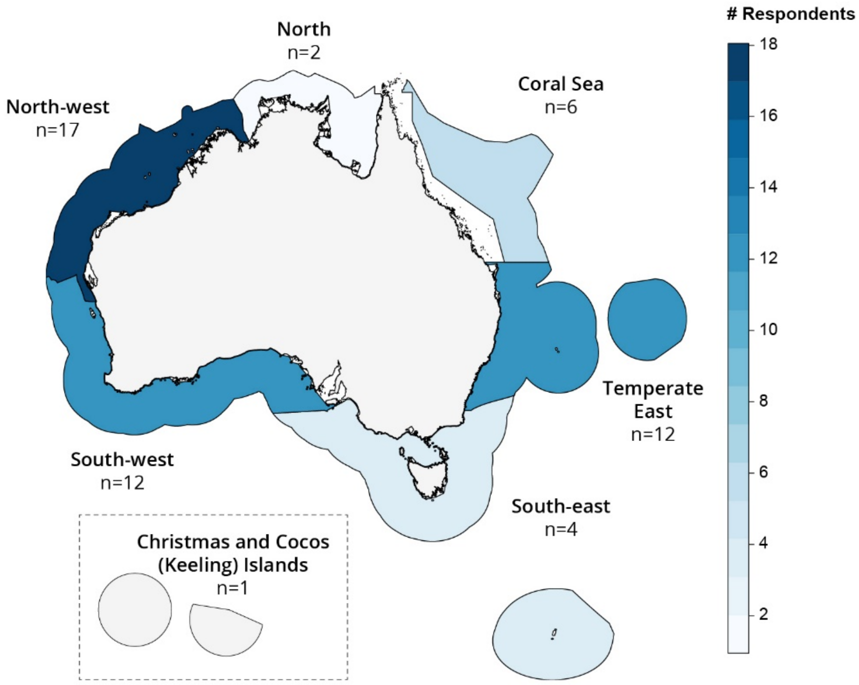

Figure 3.2 Geographic distribution of respondents' sampling activities within Australian waters.

Slightly under two-thirds (n=19, 59%) of respondents only worked in a single bioregion. The northern Indian Ocean, Southwest Pacific and North Atlantic Current were the three most sampled pelagic provinces (**Figure 3.3**). The Leeuwin Current, Gulf Stream, southern Indian Ocean, Mediterranean and Eastern Tropical Pacific followed in close second. Little monitoring occurred in polar regions, and no respondents reported working in the Somali Current, Humboldt Current, and Kuroshio-Oyashio Current provinces.

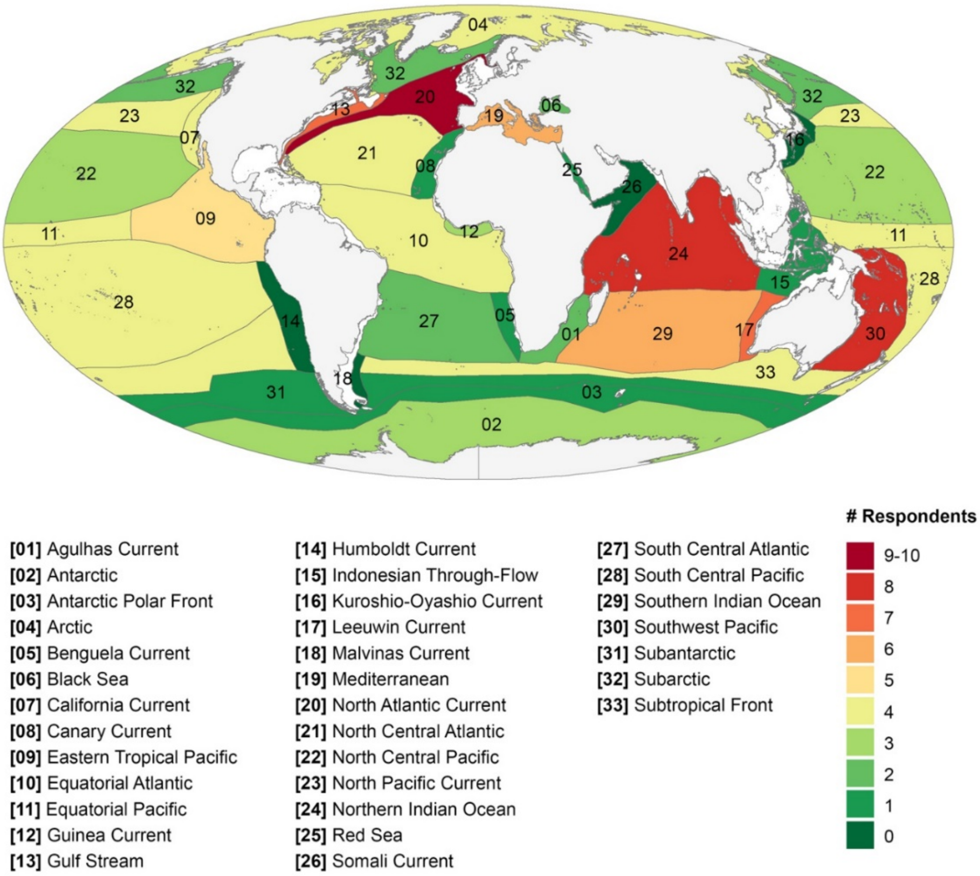

Figure 3.3 Geographic distribution of respondents' sampling activities across Global Open Ocean and Deep Seabed (GOODS) pelagic provinces.

3.2.3 Method use and familiarity

On average, respondents were aware of more than half (n=26) of the pelagic methods listed in the survey, and showed greatest familiarity with citizen science (n=54, 87%), UVC (n=52, 84%), visual vessel-based surveys and biopsy sampling (n=51, 82%), as well as water sampling (e.g. eDNA, n=50, 81%) and plankton nets, remote sensing, and satellite telemetry (n=49, 79%) (**Figure 3.4**). Of these, the following methods were frequently used by at least 10 respondents: vessel-based visual surveys (n=24, 38%), water sampling (e.g. eDNA) and UVC (n=19, 31%), citizen science (n=17, 27%), land-based surveys (e.g. theodolite tracking, n=14, 23%), satellite remote sensing and plankton nets (n=13, 21%), active acoustics (e.g. echosounders, n=11, 18%) and lastly satellite telemetry and passive acoustics (n=10, 16%).

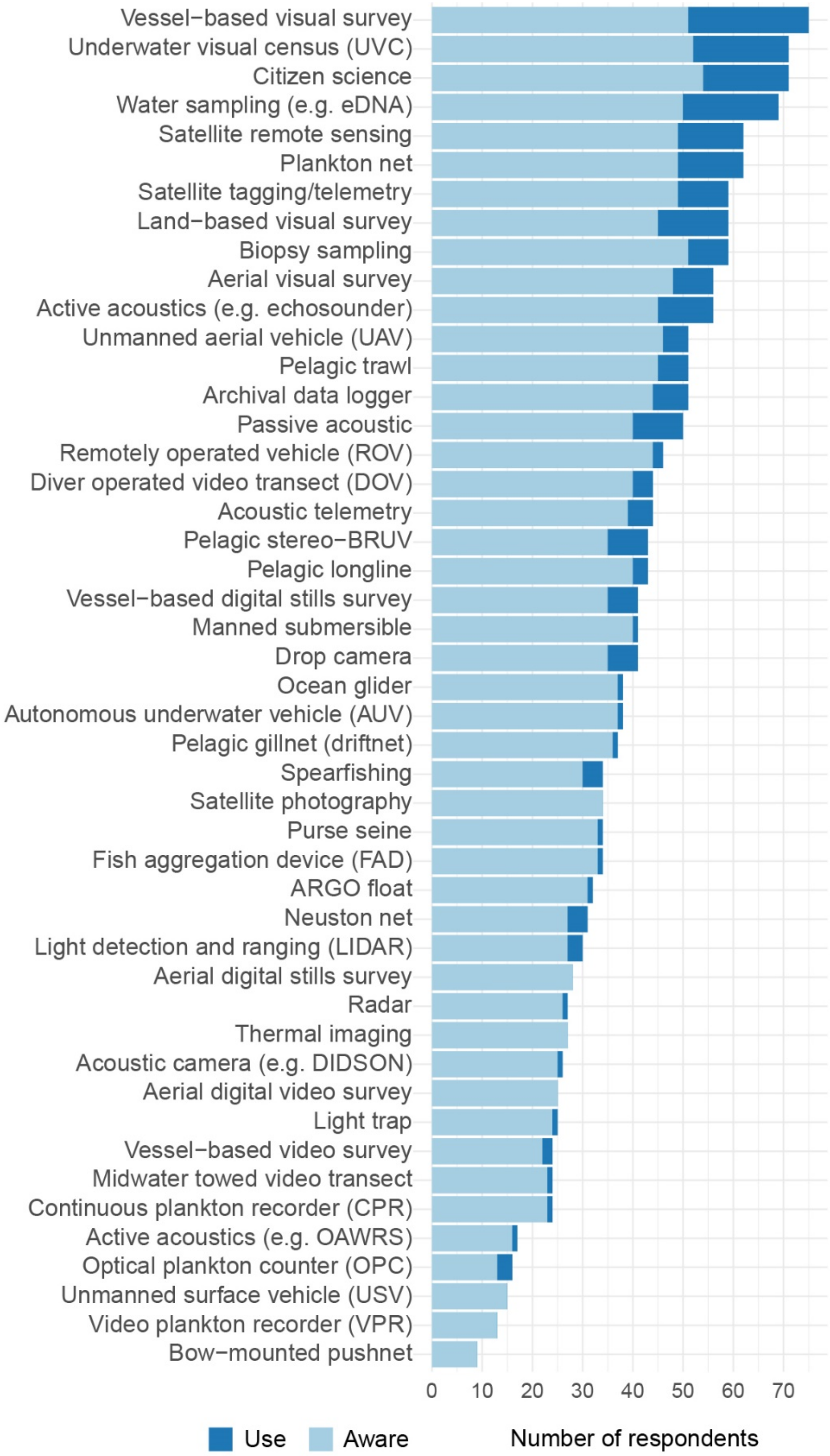

Figure 3.4 Use and awareness patterns of pelagic sampling methods. Respondents were asked to select a maximum of five methods they used most, among those of which they had some knowledge.

Bow-mounted pushnets, acoustic cameras (e.g. DIDSON), aerial photography, VPRs, or USVs were amongst the methods never used by respondents.

A third of respondents preferentially sourced their methods in house (n=19, 32%), although uptake from other research groups or projects (without modification) was also substantial (n=18, 30%). About a quarter (n=15, 24%) of respondents had used the majority of their sampling methods for more than a decade, whilst only 11% (n=7) had mostly been engaged in pelagic sampling for less than a year. Unsurprisingly, emerging methods like OAWRS, aerial videography or infrared technologies had only been in use very recently, whilst pelagic gillnets (driftnets) and light traps were long-term components of respondents' sampling programmes.

Most people who relied on water sampling (e.g. eDNA, n=11, 65%), passive acoustics (n=7, 70%), satellite telemetry (n=7, 70%), remote sensing products (n=7, 58%) or citizen science (n=13, 76%) did so very often to nearly always. By contrast, methods that were rarely employed included: LIDAR (n=2, 100%), manned submersibles (n=1, 100%), CPR (n=1, 100%), acoustic telemetry (n=3, 60%), ROVs (n=1, 50%), pelagic longlines (n=2, 67%), and Neuston nets (n=2, 67%).

As a whole, patterns of use were not expected to change significantly in the future. Some respondents stated they anticipated increasing their use of biopsy sampling, UVCs, active acoustics (e.g. echosounders), and remote tools such as ROVs, USVs and drones, with changes in usage predominantly related to funding availability as well as project-specific requirements. One respondent said they showed mounting interest in stereo-BRUVs as a non-invasive alternative to pelagic longlines for estimating shark abundance and diversity. Another explained that health and safety restrictions and difficulties in maintaining appropriate certifications have made SCUBA surveys challenging and prohibitively expensive, meaning other methods will likely replace them in years to come.

3.2.4 Monitoring

Over three quarters (n=41, 77%) of respondents followed rigorous, probabilistic designs (e.g. random, stratified, adaptive), while the rest (n=22, 23%) collected their data *ad hoc* and spontaneously (e.g. convenience, accidental, ad libitum, and preferential sampling). Most pelagic methods were used in long-term monitoring activities, to estimate patterns in the abundance, distribution and habitat preferences of species, and assess human impacts on communities and habitats (**Figure 3.5**).

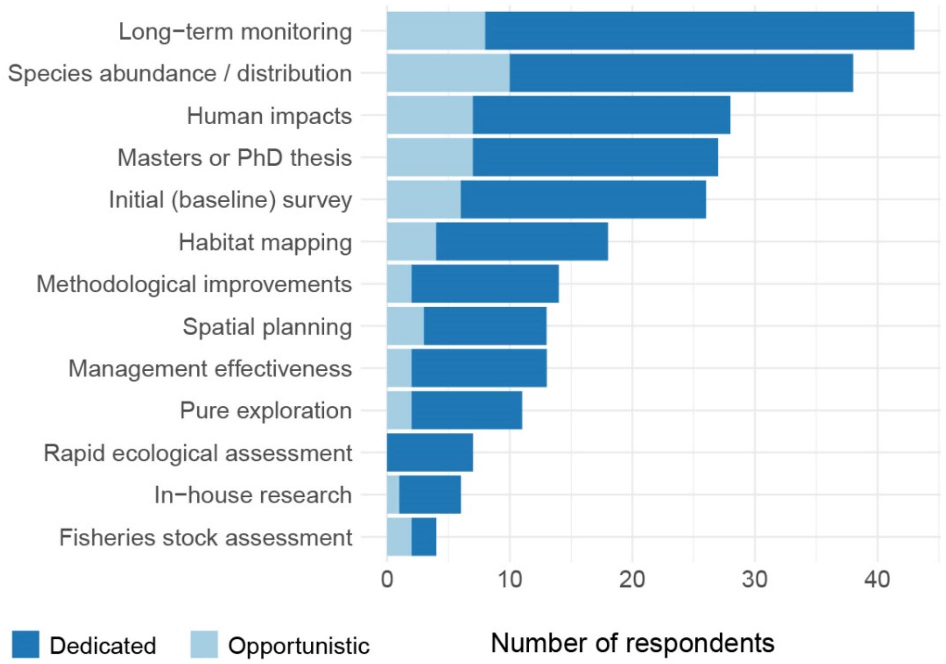

Figure 3.5 Goals of pelagic sampling. Colours indicate whether methods are mostly used as part of dedicated or opportunistic surveys (see Appendix 1 for definitions).

Virtually half (n=23, 44%) of respondents monitored only one biological EOV, irrespective of the number of methods they had access to. An equal proportion preferentially monitored the abundance/distribution of turtles, birds and mammals (n=23, 44%), with the distribution of fish (n=11, 21%) being the second most popular monitoring target. UVCs (n=14, 78%), pelagic trawls (n=4, 80%), spearfishing (n=3, 100%), pelagic longlines (n=3, 100%), light traps (n=1, 100%), active acoustics (e.g. echosounders, n=5, 63%), DOVs (n=2, 67%) and stereo-BRUVs (n=6, 100%) were predominantly or exclusively used to monitor fish abundance and distribution. By contrast, sea turtles, mammals and birds were more frequently monitored with UAVs (n=4, 80%), land-based surveys (e.g. theodolites, n=13, 93%), passive acoustics (n=7, 78%), infrared (n=1, 100%), archival tags (n=5, 71%), as well as boat (visual, n=20, 91%; still photography, n=4, 100%) and aerial (visual, n=6, 86%; videography, n=1, 100%) surveys. Unsurprisingly, CPR (n=1, 100%), OPCs (n=3, 100%), neuston nets (n=3, 100%), and plankton nets (n=7, 64%) were used primarily for characterising zooplankton biomass and diversity, though one respondent stated relying on echosounders also. Argo floats (n=1, 100%), water sampling (e.g. eDNA, n=6, 35%) and remote sensing (n=6, 55%) were mainly of interest in studies of phytoplankton.

No respondent monitored more than two biogeochemical EOVs (1 variable, n=15, 63%; 2 variables, n=9, 37%). Stable carbon isotopes and ocean colour were the preferred targets (n=4, 17%), followed by dissolved oxygen and nutrients (n=3, 13%). ROVs, AUVs, and ocean gliders were rarely used to measure dissolved oxygen concentrations. Neuston nets, spearfishing, and biopsies were used to obtain carbon isotope data. Satellite remote sensing and active acoustics (e.g. OAWRS) were predominantly used in ocean colour applications. Water sampling (e.g. with Niskin bottles) was seen as well-suited to capture most biogeochemical EOVs. One respondent indicated using Argo float for monitoring oxygen, nutrients, inorganic carbon, suspended particulates, dissolved organic carbon and ocean colour.

Exactly 80% of respondents (n=36) monitored no more than two physical EOVs. The variables most frequently considered were sea state (n=12, 27%) and sea surface temperature (n=8, 18%). One respondent stated recording sea ice using UAVs. LIDAR was used exclusively for sea surface height measurements (n=1, 100%), whereas sea surface salinity data were more readily monitored by respondents using ocean gliders (n=1, 100%), ROVs (n=1, 100%), AUVs (n=1, 100%), or water sampling (n=5, 50%). Sea surface temperature was often recorded during activities involving UVCs (n=5, 63%), spearfishing (n=1, 100%), remote sensing (n=8, 80%), plankton nets (n=2, 100%), OPCs (n=1, 100%), infrared cameras (n=1, 100%), FADs (n=1, 100%), drop cameras (n=1, 100%), vessel-based videography (n=1, 100%), archival tagging (n=1, 100%), and citizen science (n=2, 100%).

Where data were collected through time, respondents often reported using passive acoustics techniques (n=6, 75%), remote sensing (n=5, 45%) and pelagic longlines (n=1, 50%) every day, whereas FADs (n=1, 100%), Argo floats (n=1, 100%), and vessel-based visual and video surveys (n=2, 50% and n=1, 50% respectively) were more typically used weekly. Satellite telemetry (n=2, 29%) and drop cameras were more predominant (n=1, 25%) in monitoring programmes run quarterly, and those respondents who used light traps (n=1, 100%), pelagic trawls (n=4, 100%) and DOVs (n=2, 100%) only did so on an annual basis.

The majority of methods were used during daylight hours (UVCs, n=15, 94%; UAVs, n=5, 100%; land-based surveys (e.g. theodolites), n=13, 100%; manned submersibles, n=1, 100%; LIDAR, n=1, 100%; FADs, n=1, 100%; water sampling (e.g. eDNA), n=12, 75%; DOVs, n=2, 100%; BRUVs, n=5, 100%; vessel-based visual surveys, n=20, 95%; vessel-based digital stills surveys, n=4, 100%; vessel-based videography, n=2, 100%; aerial visual surveys, n=7, 100%; aerial videography, n=1, 100%). One respondent stated using infrared technology during the day. Light traps were only used at night (n=1, 100%), whilst several other methods functioned around the clock (pelagic trawls, n=2, 67%; ROVs, n=2, 100%; remote sensing, n=7, 100%; passive acoustics, n=8, 89%; OPCs, n=2, 67%; active acoustics (e.g. OAWRS), n=1, 100%; Neuston nets, n=2, 67%; pelagic longlines, n=2, 100%; ocean gliders, n=1, 100%; pelagic gillnets, n=1, 100%; active acoustics (e.g. echosounders), n=5, 63%; drop cameras, n=3, 75; CPR, n=1, 100%; AUVs, n=1, 100%; Argo floats, n=1, 100%).

3.2.5 Advantages and drawbacks

Questionnaire respondents were asked to list what they perceived as being the pros and cons of each sampling method. Their feedback is summarised below:

Acoustic telemetry was deemed easy-to-use and generally cost-effective, providing valuable data.

Aerial videography benefits from standardised methodologies that can be replicated, but is time-consuming and very costly.

Aerial visual surveys provide opportunities to detect wildlife in real time and refine species identifications using a circle-back approach. Absolute estimates of abundance can also be obtained, for instance, using distance sampling methodology. However, surveys often prove expensive, particularly in remote regions away from airports, and are weather-dependent. The need for skilled observers is limiting.

Argo floats allow a wide range of variables to be sampled, their autonomous nature making them very flexible. Nonetheless, they remain constrained by limited battery life and fouling.

AUVs and ocean gliders are valuable for generating long-term datasets in remote locations but can be challenging to deploy and recover.

Biopsy sampling tends to be relatively affordable and can be easily paired with additional methods to maximise data collection opportunities. Technical requirements are also generally low, such that it can be implemented in developing countries under low budgets. Local scientists/students can be trained quickly.

Vessel-based surveys (visual, photography and videography) can capture relevant behavioural data and are a reliable and tested means of documenting marine mammal distribution/abundance. However, they are expensive, labour-intensive, hindered by local weather and daylight conditions, and give little spatial coverage.

Pelagic BRUVs were seen as a non-invasive alternative to fisheries-dependent methods, often suitable for environments that are too difficult to access via scuba (e.g. deep, wave-exposed habitats). The digital records that BRUVs produce also have value for supporting outreach activities. The main drawbacks identified by respondents included camera movements due to swell and wave refraction off headlands, the labour-intensive nature of data post-processing, a decreased reliability in species identification with increasing distance from the platform, and the need for bait. BRUVs also likely suffer from imperfect and heterogeneous detectability and may demand high levels of effort over long time periods to produce sensible results.

Whilst **citizen scientists** generate large amounts of data and facilitate public engagement, they may not be as accurate as trained professionals if appropriate training isn't provided. Disparities in the protocols used by various citizen science (and by extension, pelagic sampling) programmes may also preclude meaningful comparisons.

The **CPR** boasts a long-term dataset but is selective and does not sample marine predators.

Pelagic trawls (and FADs) yield samples with generally high taxonomic resolution that can support genetic and stable isotope analyses. They are effective at collecting abundance data on marine fishes, though another emphasised that they can only provide a snapshot in time and cannot detect natural fine-scale spatial relationships between species. A major disadvantage lies in that trawls are extractive and require appropriate facilities to store biological samples in ethanol/formalin.

Unravelling the movements and behaviour of large pelagic species is virtually impossible without the use of **satellite and archival tags**. That said, these tags are often very costly and funding insufficient to achieve large sample sizes and/or high spatial or temporal resolution. The quality and frequency of satellite tag uplinks can also be highly variable, with no way of determining what happened when a signal stops.

DOVs were favoured for their ease of use and storage but were flagged as requiring long processing times.

Drop cameras generate quality video footage of deep environments but lack manoeuvrability.

Active hydroacoustics (e.g. echosounders, OAWRS) have the advantage of being non-extractive, and relatively cheap once the gear is in place, though their taxonomic resolution tends to be low.

Water sampling (e.g. eDNA) is easy to use, low-cost, and does not typically require special treatment in the field. More sophisticated extraction methods like Niskin bottles can target specific depths but are more expensive and slow. Standard solutions may need to be obtained for calibration purposes.

On the one hand, it is straightforward to record numerous variables from the same vessel when using **plankton nets**. On the other hand, the samples collected by nets can take a long time to process and their processing and analysis is contingent on the availability of skilled taxonomists who are able to identify specimens reliably. Information on commercial fishing catch rates, bycatch rates, or fleet dynamics is impossible without access to logbook data derived from **pelagic longlines.** However, they are highly invasive and processing their samples proves costly, and time- and labour-intensive.

Passive hydroacoustics is ideal in long-term monitoring programmes and can run on continuous 24-hour cycles, independently of weather conditions. Whilst it provides valuable information without disturbance to wildlife or their habitats, the volume of data typically generated by passive acoustic methods is enormous and requires significant investment in post-processing. Sampling bias towards more vocal or acoustically active species/population cohorts is also an issue. Additionally, the marine environment eats electronics, so there may be a steep curve to go past the experimental version 1 prototypes (that usually fall over at some point) to frequently used, reliable tools.

UVCs and scuba surveys yield insights into the ecology of species that cannot be surveyed from the surface, but can only be conducted in a narrow range of conditions deemed safe for divers. They are also difficult to standardise, as the total time spent underwater may vary as a function of divers' different air consumption rates, and as some species exhibit differing behavioural responses to diver presence.

Satellite remote sensing allows a holistic understanding of oceanography at various time scales, and provides a platform for retrospective analyses of historical data. Its usefulness is frequently undermined by obstructions from cloud cover.

Spearfishing is destructive, but selective, such that its impacts are constrained to the target species. It obviates the need to collect specimens by scuba, and allows the selection of only the exact size and species of fish/shellfish under study.

Land-based surveys (e.g. theodolite) are cheap once the equipment has been procured, and can offer a window into natural animal behaviour and the influence of anthropogenic impacts with no observer bias. Protocols are well established and can be easily replicated, yet areal coverage is limited as the method can only capture data on a small number of animals transiting within viewing distance of the station.

3.2.6 Ownership and purchase/maintenance costs

About a quarter (n=12, 26%) of respondents indicated owning their pelagic sampling equipment, though a large proportion both rented and purchased equipment (n=30, 65%) (**Figure 3.6A**). Vessel-based visual surveys (>$50,000, n=5, 50%), active acoustics >$50,000, n=5, 63%), satellite remote sensing (>$50,000, n=1, 33%), aerial surveys (visual and videography, >$50,000; n=2, 100% and n=1, 100% respectively), ocean gliders (>$50,000, n=1, 100%), AUVs (>$50,000, n=1, 100%), pelagic trawls (>$50,000, n=1, 50%) and Argo floats (>$50,000, n=1, 100%) were amongst the most expensive gear to acquire (**Figure 3.6B**). Conversely, citizen science programmes (<$5,000, n=6, 75%), UVCs (<$5,000, n=8, 89%), stereo-BRUVs (<$5,000, n=3, 75%), spearfishing (<$5,000, n=3, 100%), DOVs (<$5,000, n=2, 100%), light traps (<$5,000, n=1, 100%), pelagic gillnets (<$5,000, n=1, 100%) and acoustic tags (<$5,000, n=1, 100%), were reported as affordable. Maintenance daily operation costs were usually far lower, with the exception of aerial and vessel-based visual surveys, remote sensing, and to a lesser degree LIDAR, active acoustics (e.g. echosounders) and pelagic longlines.

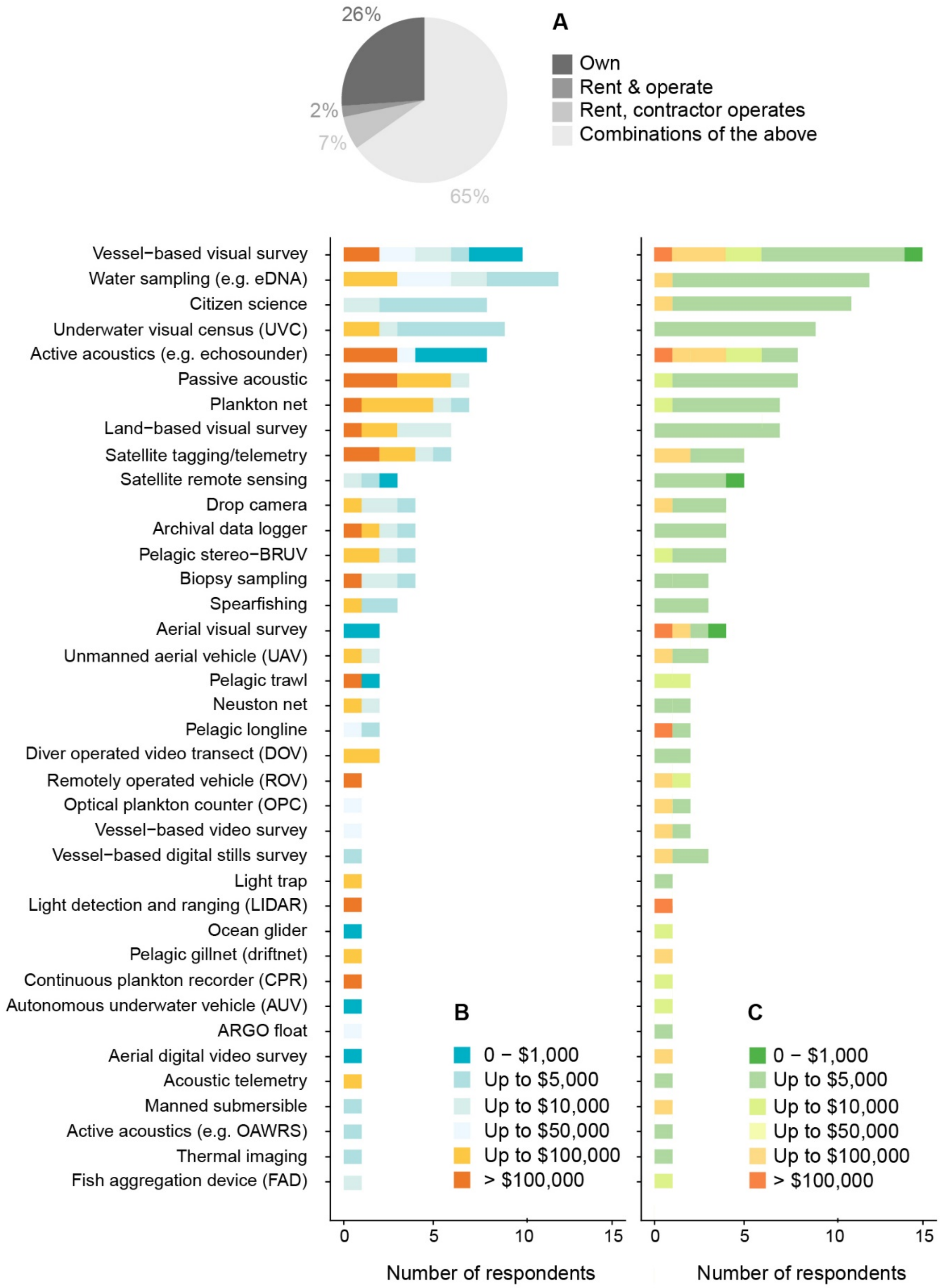

Figure 3.6 Trends in **(A)** equipment ownership among respondents, **(B)** purchase costs and **(C)** maintenance/daily costs to operate.

3.2.7 User perceptions

Citizen science was almost universally seen as a low-cost, easy-to-use and safe method (**Figure 3.7**). Similarly, DOVs, drop cameras, active acoustics, Argo floats, CPR, light traps, and pelagic gillnets were deemed mostly affordable with low technical requirements, though some presented moderate levels of risks and operational complexity. FADs and manned submersibles were unanimously reported at moderate and high levels in all classes respectively. Aerial digital video surveys, aerial visual surveys, vessel-based visual surveys, AUVs, and ocean gliders were among the methods perceived as being the costliest. Those that required most technical expertise to operate included active acoustics (e.g. echosounders), archival tags, passive acoustics, remote sensing, and satellite telemetry. One respondent believed LIDAR to be prohibitively expensive, and UVCs and AUVs to be prohibitively risky.

3.2.8 Improvements and aspirations

When asked what priorities should be addressed to abate costs and support monitoring efficiency and/or data quality, many respondents cited gear modifications as a significant potential area for improvement. These included the development of optimal attractants to entice pelagic species into the field of view of stereo-BRUVs, increased battery life/data storage/memory in Argo floats and acoustic loggers, longer cables for drop cameras, more powerful thrusters fitted in ROVs, more reliable lighting and optics in towed video systems, more reliable surface-fish positioning, more fuel-efficient outboard engines, or higher-resolution digital cameras for underwater imagery. Greater information sharing, collaboration and coordination of planning across agencies would also go a long way towards making pelagic sampling more cost-effective and filling current gaps in capability, as would further transparency around the analytical requirements for robust data collection and synthesis and increased opportunities for relevant training. Interestingly, several respondents emphasised the need to produce standardised software and operating procedures for the consistent monitoring of pelagic systems.

3.2.9 Minimum needs for method selection

Affordability and reliability were essential minimum requirements for a pelagic method, according to respondents (n=12, 55%). Simplicity and ease of maintenance/operation (n=4, 22%) were also mentioned, as was the capacity to produce ‘useful’ data by allowing standardised designs and cross-calibration against other methods (e.g. pelagic trawls capable of sampling depths targeted during acoustic surveys).

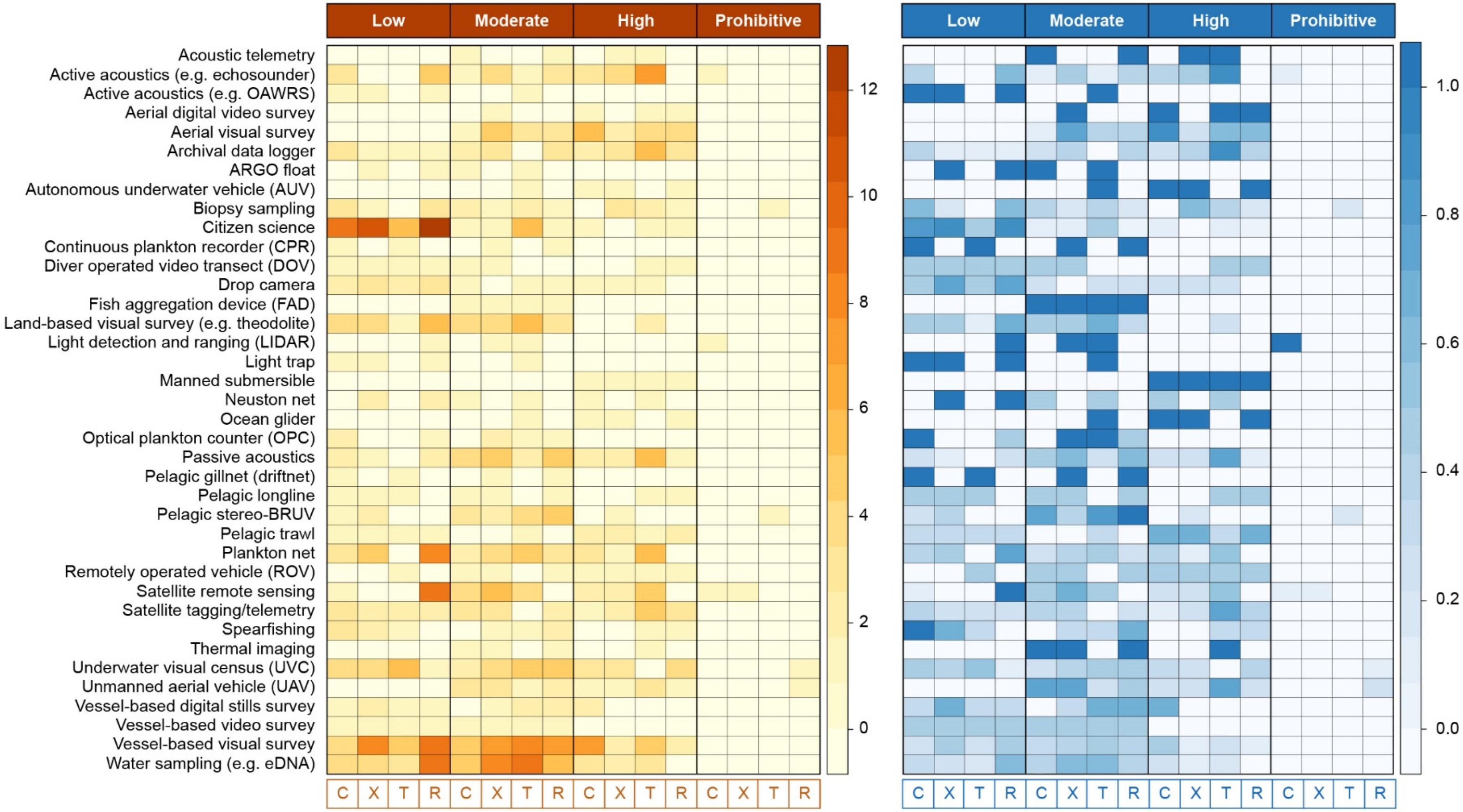

Figure 3.7 User perceptions of pelagic sampling methods. Left: Number of responses. Right: Same data, shown as a percentage of class-specific totals for each method. Classes are as follows: C = Total cost (including post-processing of data), X = Operational complexity, T = Post-processing or technical requirements, and R = risks (health and safety, expense and potential loss, etc.).

These reports closely reflected respondents' primary motivations for choosing a given method (**Figure 3.8**), which placed an emphasis on costs (n=30, 75%), taxonomic specificity and relevance to the target organisms under scrutiny (n=29, 73%), as well as data quality (n=24, 60%). Popularity (n=2, 5%) and relevance to management needs (n=4, 10%) were amongst the least prominent reasons for using pelagic methods.

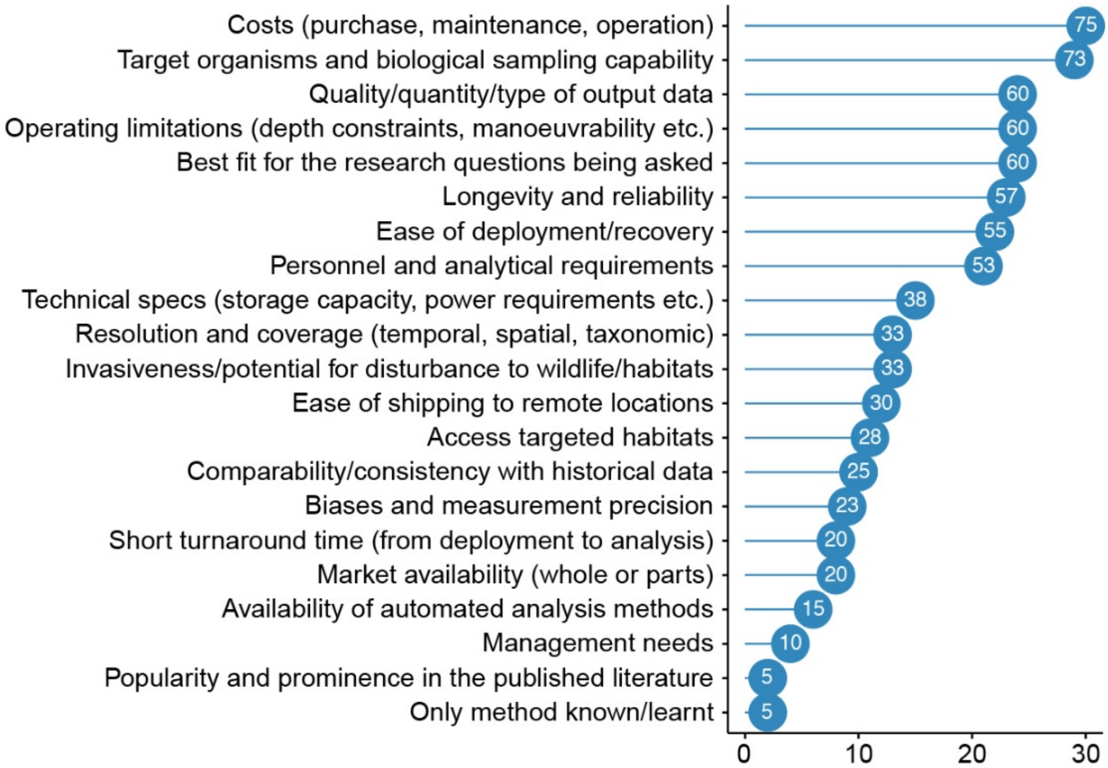

Figure 3.8 Summary of respondents' motivations for selecting a pelagic sampling method. Segments denote response counts, and filled circles represent the corresponding percentages (from a total of n=40 respondents).

3.2.10 Standard operating procedures

Nearly half of respondents (n=21, 49%) were in favour of an Australian national standard for pelagic monitoring, whilst 7% (n=3) opposed the idea and 44% (n=19) could not decide. Those who disagreed provided the following reasons:

- There is no need for datasets to be comparable.
- Data collection methods should be chosen and designed to answer specific questions, not to make every study comparable. Besides, the range of variables/questions at hand is often too great to support standardisation efforts without undermining one's ability to yield answers in the most cost- and time-effective manner possible. In addition, the equipment available to different research groups is likely to vary. This makes comparisons difficult, but it should not limit the value each study can add if it is a scientifically justifiable one.

- Instruments are deployed (in most instances) to address particular questions/issues that may vary depending on the species involved, the area or mode of deployment, which makes standardisation a challenge. There is, however, a case for ensuring that data is collected in such a way that comparative assessments can be undertaken. Definite improvements could be made to ensure that researchers collect, manage and analyse data in an appropriate and educated manner.

On average, respondents were no more inclined than disinclined to modify their current methodology to adopt a standardized pelagic monitoring protocol, should one be proposed (**Figure 3.9A**). However, respondents indicated that uptake would be more likely should the protocol provide flexibility (n=32, 80%), incur minimal costs (n=28, 70%) and offer opportunities to incorporate legacy data (n=26, 65%) (**Figure 3.9B**). Agreement among experts (n=24, 60%), succinctness and clarity (n=23, 58%), comparability to other datasets (n=21, 53%) and the capacity to continue answering questions from existing projects (n=21, 53%) were also important. Defensibility (n=12, 30%) was the factor that least increased respondents' interest in standard protocols.

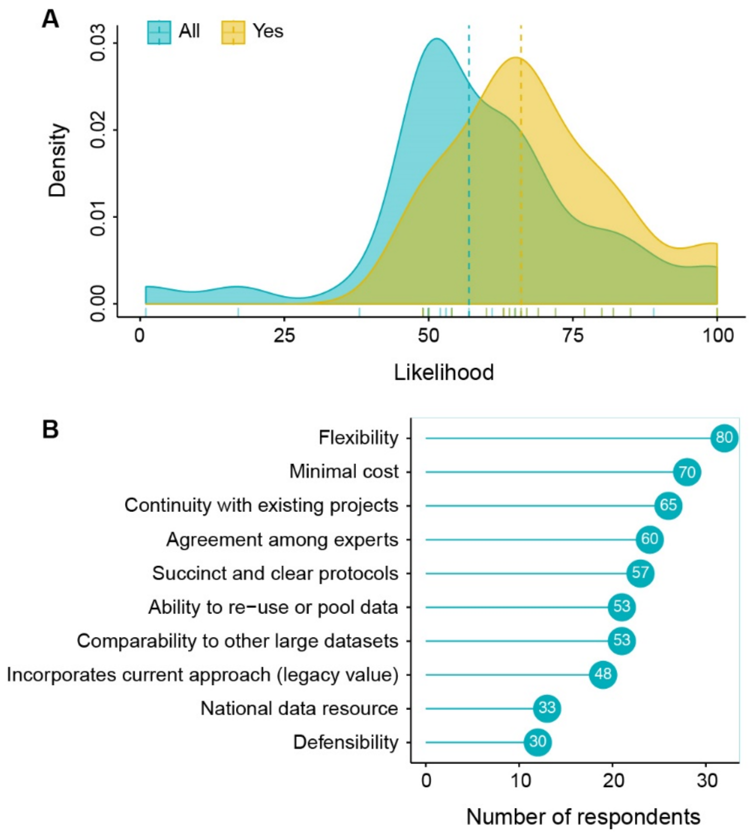

Figure 3.9 Summary of interest in adopting a standardized pelagic monitoring protocol. **(A)** Density plots showing the level of intent (on a scale of 1 to 100) to modify current methodology, between all respondents ('All') and only those favourable to standardisation ('Yes'). **(B)** Factors likely to increase interest in, and/or uptake of, a standardised protocol. Segments denote response counts, and filled circles the corresponding percentages (from a total of n=40 respondents).

4. Comparison of pelagic sampling methods

A growing number of ecological studies aim to appraise similarities and differences in the distributions of marine species across seascapes (e.g. Langlois *et al.* 2012), quantify the responses of ecological communities to management schemes (e.g. zoning within MPAs) and human threats (e.g. Queiroz *et al.* 2016), and describe trends in biodiversity through time (e.g. Paleczny *et al.* 2015). Such endeavours require datasets that can be reliably compared, yet instrumentation-related variability can introduce biases that may impair direct comparability (Caldwell *et al.* 2016). All methods have inherent strengths and weaknesses, and the effectiveness and suitability of one over another ultimately depends on survey objectives, available resources, and the environment being sampled (depth, currents, etc.), all of which underpin data quality and scalability (Przeslawski *et al.* 2018). Direct field comparisons of individual methods therefore remain challenging, as they require synchronised observations that can control for a multitude of factors (weather, season, habitat etc.). Although there are numerous benefits to combining methods (Day 2008), pairwise deployments also risk standardising samples only by location and species, without giving a full representation of all facets of complementarity (Mallet & Pelletier 2014).

4.1 Methods

In this section, we conduct a qualitative review of the literature to contrast common pelagic sampling methods. Insights are drawn from the Web of Science (filtering by "Marine Freshwater Biology") and Google Scholar databases, which we searched using combinations of the terms: "pelagic", "compar*", and "method*", along with individual gear types (as the sheer number of gears identified herein precluded every pairing from being tested). Only peer-reviewed journal articles published in the English language and focusing on pelagic or bentho-pelagic taxa/methods were retained. We did not consider comparisons of single gear types with different configurations or minor modifications (e.g. two trawls vs. one) (Heino *et al.* 2010).

We note that a number of detailed reviews already exist in the literature for a subset of methods (e.g. Hofmann & Gaines 2008, Katsanevakis *et al.* 2012, Duffy *et al.* 2013, Mallet & Pelletier 2014, Maxwell *et al.* 2014, Caldwell *et al.* 2016, Danovaro *et al.* 2016, Bean *et al.* 2017, Letessier *et al.* 2017, Paris *et al.* 2018).

All else being equal, the most straightforward way to compare multiple sampling methods is to identify whether there are any significant differences between their outputs (Przeslawski *et al.* 2018). Ideally, this is done by including methodology as a factor term in a statistical test or in a model with a common dependent variable. Alternatively, correlations between output values can be computed, or as a last resort, qualitative inference is possible (e.g. visual comparisons of spatial distribution maps) (**Table 4.1**).

4.2 Results

4.2.1 Statistical significance

The most prevalent comparative benchmarking studies involved aerial vs. shipboard visual surveys, and capture sampling (e.g. pelagic trawls) vs. hydroacoustics, perhaps because these methods are among those that have been established the longest (**Table 4.1**). A number of novel approaches (e.g. eDNA) are beginning to receive attention, but remain in great need of direct validation under a range of field conditions. With the exception of Santana-Garcon *et al.* (2014a) and Boldt *et al.* (2018) we could not find any records of pelagic BRUVS being compared to other techniques. Pelagic species are often seen on demersal BRUVS, however, and numerous authors have compared these to other forms of sampling (Willis *et al.* 2000, Cappo *et al.* 2004, Watson *et al.* 2005, Colton & Swearer 2010, Watson *et al.* 2010, Lowry *et al.* 2011, Lowry *et al.* 2012, McCauley *et al.* 2012, Holmes *et al.* 2013, McLean *et al.* 2015, Bosch *et al.* 2017).

Overall, few studies compared more than two methods at a time, and no consistent pattern in relative efficiency could be detected, as results varied widely between areas, taxonomic groups, and survey parameters (**Table 4.1**). We caution that publication bias should also be considered, as studies that found no differences between sampling methods are less likely to be published (Jennions *et al.* 2013).

4.2.2 Congruence of ecological relationships

Many studies do not lend themselves to direct methodological comparisons, owing to the use of different metrics, detection range etc. Examining the level of congruence between inferred ecological relationships (Flannery & Przeslawski 2015) provides another window into patterns of complementarity between sampling methods. Only a small proportion of studies assessed relationships with habitat variables (**Table 4.1**), but the majority identified congruent ecological relationships, despite methodological discrepancies.

Table 4.1 Summary of selected studies comparing pelagic sampling methods in both marine and freshwater ecosystems. Studies that tested for statistical significance are indicated by **[*]**. Agreement between methods is reported as ✓ (no difference), ✓/× (variable, e.g. by metric, conditions, taxon) and × (difference). Ecological relationships are colour-coded where appropriate (green: congruent, orange: incongruent). Method categories are as per **Section 2.3**.

Reference	Taxon	Variable	Method 1	Method 2	Method 3	Results	Details
Andaloro *et al.* (2013)	Fishes	Diversity Abundance Structure	**Robotics** (ROV)	**Visual** (UVC)	n/a	✓/×	**[*]** Strong differences in assemblage composition at all depths. No significant difference in abundance of pelagic fishes.
Bach *et al.* (2003)	Fishes	Distribution	**Capture** (Longline)	**Satellite** (Biotelemetry)	n/a	✓	**[*]** Depth distributions similar.
Bagge *et al.* (1996)	Crustaceans / larvae	Density	**Capture** (Trawl, Plankton net)	**Capture** (Light trap)	n/a	×	**[*]** Catching efficiency species- and gear-dependent.
Båmstedt *et al.* (2003)	Jellyfish	Abundance Distribution	**Hydroacoustics** (Echosounder)	**Capture** (Plankton net)	**Robotics** (ROV)	✓	Range of abundance estimates obtained with acoustic and video methods overlapped with results from net samples.
Baran *et al.* (2017)	Fishes	Abundance Size	**Capture** (Trawl)	**Hydroacoustics**	n/a	✓/×	**[*]** Differences in size distributions between methods vary with depth. No significant differences in abundance estimates.
Boldt *et al.* (2018)	Fishes	Abundance Biomass	**Capture** (Trawl)	**Visual** (Pelagic BRUV)	n/a	✓/×	Higher trawl catches associated with higher mean numbers of fish observed per BRUV frame, but fish sizes significantly larger on BRUVS.
Briggs *et al.* (1985)	Seabirds	Density	**Visual** (Aerial visual)	**Visual** (Shipboard visual)	n/a	✓/×	**[*]** Differences under controlled survey experiments. None under variable field conditions.

Reference	Taxon	Variable	Method 1	Method 2	Method 3	Results	Details
Buckland *et al.* (2012)	Seabirds	Abundance	**Visual** (Aerial visual)	**Visual** (Aerial photo)	**Visual** (Aerial video)	✓/x	No difference between digital methods, but abundance appreciably lower from aerial visual data.
Bulleri & Benedetti-Cecchi (2014)	Fishes	Diversity Density Structure	**Capture** (Spearfishing)	**Visual** (UVC)	n/a	✓/x	**[*]** Significant differences in community structure, density, but estimates of species richness are comparable for any sample size.
Carrera *et al.* (2006)	Fishes	Occurrence	**Visual** (LiDAR)	**Hydroacoustics** (Echosounder)	n/a	✓/x	**[*]** Target detection performance differed amongst areas. Spatial patterns dissimilar.
Choat (1993)	Fish larvae	Density Composition	**Capture** (Plankton net, Neuston net)	**Capture** (Light trap)	n/a	✓/x	**[*]** Differences between nets for all but the most abundant families. Light traps more selective.
Churnside & Thorne (2005)	Zooplankton	Occurrence Density	**Visual** (LiDAR)	**Hydroacoustics** (Echosounder)	n/a	✓	**[*]** Significant positive correlation at optimal data threshold.
Churnside *et al.* (2003)	Fishes	Occurrence Density	**Visual** (LiDAR)	**Hydroacoustics** (Echosounder)	n/a	✓	High correlation between LIDAR and acoustic reflectivity of fish schools in surface layers.
Churnside *et al.* (2009)	Fishes	Occurrence Density	**Visual** (LiDAR)	**Hydroacoustics** (Echosounder)	**Capture** (Trawl)	✓/x	**[*]** Generally positive correlations, though strength varies by time of day and data treatment protocol.

Reference	Taxon	Variable	Method 1	Method 2	Method 3	Results	Details
Clark *et al.* (2010)	Cetaceans	Abundance Density	**Visual** (Aerial visual)	**Hydroacoustics** (Passive logger)	n/a	×	**[*]** Lower detection rate during aerial surveys. No strong relationship between visual numbers and acoustic detections.
Cragg *et al.* (2016)	Seabirds	Occurrence Abundance Activity	**Land-based** (Visual surveys)	**Land-based** (Radar)	**Hydroacoustics** (Passive logger)	✓	**[*]** No difference in mean counts or behaviours across methods.
Emmrich *et al.* (2010)	Fishes	Biomass Density Size	**Capture** (Trawl)	**Hydroacoustics** (Echosounder)	n/a	✓/×	**[*]** No difference between biomass estimates, except for deepest layers. Size-frequency distributions considerably different, but with similar mean size.
Ferguson *et al.* (2018)	Cetaceans	Distribution Density Abundance	**Visual** (Aerial visual)	**Visual** (Aerial Photo)	**Visual** (UAV)	✓/×	Density estimates differ among methods for each species, but some distributions patterns consistent. Comparisons depend on survey sector and analytical method.
Gargan *et al.* (2017)	Elasmobranchs	Occurrence Abundance	**Visual** (Shipboard visual)	**Genomics** (eDNA)	n/a	✓	**[*]** Statistically significant association between the visual and genetic detection.
Godwin *et al.* (2016)	Cetaceans	Behaviour	**Land-based** (Theodolite)	**Visual** (Shipboard visual)	**Satellite** (Biotelemetry)	✓/×	**[*]** Significant differences found only for some behaviours.
Hansson & Rudstam (1995)	Fishes	Abundance	**Capture** (Gillnet)	**Hydroacoustics** (Echosounder)	n/a	✓	**[*]** Statistically significant positive relationship between vertical gillnet catches and hydroacoustic fish abundance.

Reference	Taxon	Variable	Method 1	Method 2	Method 3	Results	Details
Hara (1990)	Fishes	Occurrence Density	**Visual** (Aerial visual)	**Hydroacoustics** (Echosounder)	n/a	×	In field tests, considerably higher counts on sonar than from the air.
Heinänen *et al.* (2017)	Seabirds	Density Abundance	**Visual** (Aerial visual)	**Visual** (Shipboard visual)	n/a	✓/×	**[*]** Positive but species-dependent correlations between methods. Spatial patterns comparable, but influential predictors vary between species.
Henkel *et al.* (2007)	Seabirds	Density Diversity	**Visual** (Aerial visual)	**Visual** (Shipboard visual)	n/a	✓/×	**[*]** Significant differences for some species and population sizes.
Hernandez & Shaw (2003)	Plankton	Density Diversity Size	**Capture** (Plankton net)	**Capture** (Light trap)	n/a	✓/×	**[*]** Density variable among locations, sampling depths, and gear configurations. Size distributions mostly overlap. Variable taxonomic similarity between gears.
Hosia *et al.* (2017)	Zooplankton	Diversity Distribution Abundance	**Capture** (Trawl)	**Robotics** (ROV)	**Visual** (VPR)	×	Differences in the catch composition obtained with each of the three methodologies. Diversity and abundance underestimated in nets.
Johnston *et al.* (2017)	Pinnipeds	Abundance	**Visual** (UAV)	**Visual** (Aerial stills)	n/a	✓	Only minor differences in counts of both adults (<1%) and pups (3.7%).
Koslow *et al.* (1997)	Fishes	Biomass Structure	**Capture** (Trawl)	**Hydroacoustics** (Echosounder)	n/a	✓/×	**[*]** Overall similarity in biological community structure but biomass estimate from trawl 7-fold lower.

Reference	Taxon	Variable	Method 1	Method 2	Method 3	Results	Details
Louzao *et al.* (2009)	Seabirds	Behaviour	**Visual** (Shipboard visual)	**Satellite** (Biotelemetry)	n/a	✓	Foraging probability from vessel-based data concordant with feeding probability from tracking data. Common habitat predictors identified from both sets of data.
Mikkelsen *et al.* (2016)	Cetaceans	Distribution Density	**Hydroacoustics** (Passive logger)	**Satellite** (Biotelemetry)	n/a	✓	**[*]** Significant linear relationship between methods, with similar patterns of presence/detections across study region.
Olin & Malinen (2003)	Fishes	Abundance Diversity Size	**Capture** (Trawl)	**Capture** (Gillnet)	n/a	×	**[*]** Most abundant species in trawl almost totally missing from gillnet. Large diurnal differences in catch composition and size distribution between gears.
Pelletier *et al.* (2011)	Fishes	Abundance Diversity	**Visual** (UVC)	**Visual** (DOV)	n/a	×	**[*]** Method type and site effects significant, though not their interaction.
Perez *et al.* (2017)	Fishes	Abundance Biomass	**Capture** (Gillnet)	**Genomics** (eDNA)	n/a	×	**[*]** No significant relationship between methods for either of the two species surveyed.
Pita *et al.* (2014)	Fishes	Abundance Composition	**Visual** (UVC)	**Robotics** (ROV)	**Visual** (RUV)	✓/×	**[*]** No differences for most abundant taxa. Differences for rare taxa, with precision UVC > ROV > RUV.
Porter *et al.* (2008)	Larvae	Abundance Composition	**Capture** (Plankton net)	**Capture** (Light trap)	n/a	×	The two methods collected different taxa at different developmental stages.

Reference	Taxon	Variable	Method 1	Method 2	Method 3	Results	Details
Rayment *et al.* (2018)	Cetaceans	Occurrence Abundance	**Land-based** (Photo)	**Hydroacoustics** (Passive logger)	n/a	✓	Similar seasonal distribution, but much higher % detections on acoustic recorder at peak occurrence.
Santana-Garcon *et al.* (2014a)	Sharks	Abundance Composition Size	**Capture** (Longline)	**Visual** (Pelagic BRUV)	n/a	✓	**[*]** Relative abundance comparable, no differences in the shape of length distributions.
Schaub *et al.* (2018)	Jellyfish	Density % cover	**Capture** (Plankton net)	**Visual** (UAV)	n/a	✓	Estimates of % cover from UAV and density estimate from net comparable.
Schofield *et al.* (2017)	Sea turtles	Density	**Visual** (Aerial visual)	**Visual** (UAV)	n/a	✓	**[*]** No difference in density between methods. Similar seasonal site visitation patterns.
Shaw *et al.* (2016)	Fishes	Diversity Composition	**Capture** (Fyke net)	**Genomics** (eDNA)	n/a	✓/✗	**[*]** No significant differences in species richness but significant differences in community composition.
Soldevilla *et al.* (2014)	Cetaceans	Abundance Density	**Visual** (Aerial visual)	**Hydroacoustics** (Passive logger)	n/a	✓/✗	Acoustic methods enabled greater temporal effort, yielding a 2- to 10-fold increase in days with right whale detections over visual methods. % of days with combined acoustic/visual detections varies among sites/seasons.

Reference	Taxon	Variable	Method 1	Method 2	Method 3	Results	Details
Starr *et al.* (1996)	Fishes	Distribution Density	**Robotics** (Submersible)	**Hydroacoustics** (Echosounder)	n/a	✓/✗	Densities significantly positively correlated but more than 6 times higher form submersible.
Sveegaard *et al.* (2011)	Cetaceans	Distribution Density	**Hydroacoustics** (Passive logger)	**Satellite** (Biotelemetry)	n/a	✓	**[*]** Strong spatial accord between the number of acoustic detections and their density distribution from satellite tracking data.
Tátrai *et al.* (2008)	Fishes	Density Size	**Capture** (Gillnet)	**Hydroacoustics** (Echosounder)	n/a	✓	**[*]** Positive and highly significant correlation between gillnet catches and acoustic abundance. Significant amount of variation in acoustic abundance explained by catches.
Taylor *et al.* (2014)	Fishes, sharks, turtles, seals, cetaceans	Density	**Visual** (Aerial visual)	**Visual** (Aerial photo)	n/a	✓/✗	**[*]** Significantly higher estimates from photographic method for some species. No significant difference for other species.
Thomsen *et al.* (2012a)	Fishes	Occurrence Diversity	**Genomics** (eDNA)	**Visual** (UVC)	**Capture** (Seine, Gillnet, Fyke, Pushnet)	✓/✗	eDNA recorded higher or equal diversity than traditional methods.
Williamson *et al.* (2016)	Cetaceans	Density	**Visual** (Aerial visual)	**Visual** (Aerial video)	**Hydroacoustics** (Passive logger)	✓	Estimates of density correlate highly between visual methods. Strong correlations between visual and acoustic detections. Similar patterns of spatial variation across study area.
Winiarski *et al.* (2014)	Seabirds	Density	**Visual** (Aerial visual)	**Visual** (Shipboard visual)	n/a	✓	**[*]** Densities comparable at level of survey effort segments.

5. Applications to monitoring

Based on findings from the literature review, the general advantages of key pelagic sampling methods were identified (**Table 5.1**). Each method is also designed to measure one or more target variables, and it is useful to link these to global monitoring indicators, including:

- Essential Ocean Variables (EOVs) (see **Section 2.2**).
- Essential Biodiversity Variables (EBVs) (see **Section 2.2**).
- Essential Environmental Measures (EEMs), currently being developed under a pilot programme in Australia (https://measures.environment.gov.au/) to identify the metrics necessary for tracking change in the state of environment.

Table 5.2 (see below) summarises the capacity of pelagic sampling to quantify EOVs and EBVs. A similar synthesis will be warranted once all EEMs are released.

Table 5.1 General advantages of key groups of pelagic sampling methods. 🔴 = little applicable, 🟡 = somewhat applicable, 🟢 = mostly applicable. Refer to **Table 1.1** for a description of each method. A = Active, P = Passive, M = Manned, U = Unmanned, Rem Sens = Satellite remote sensing (including photography).

	Capture		Acoustics		Visual, optical & thermal				Robotics		Satellite		Genomics	Participatory
	A	*P*	*A*	*P*	*Underwater*	*Airborne*	*Shipboard*	*Land*	*M*	*U*	*Rem Sens*	*Tags*	*eDNA/biopsy*	*Citizens*
Large spatial coverage possible	🟡	🟡	🟡	🔴	🟡	🟢	🟢	🟡	🟢	🟡	🟢	🟢	🔴	🟡
Fine spatial resolution possible	🟡	🟡	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟡	🟡
Long deployment/sampling times	🟡	🟡	🔴	🟢	🔴	🟡	🟡	🟡	🔴	🟡	🟢	🟢	🔴	🟡
Ability to revisit exact sites	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟡	🟢	🟢
Can identify unknown/cryptic species	🟢	🟢	🔴	🔴	🟡	🟡	🔴	🔴	🟢	🟡	🔴	🔴	🟢	🟡
Genetic/morphological data	🟢	🟢	🔴	🔴	🟡	🔴	🔴	🔴	🟢	🟡	🔴	🟡	🟢	🔴
Behavioural observations/data	🔴	🔴	🟢	🟢	🟢	🟡	🟢	🟢	🟢	🔴	🔴	🟢	🔴	🟡
Animal abundance estimates	🟢	🟢	🟢	🟢	🟢	🟢	🟡	🟢	🔴	🔴	🟡	🟡	🟡	🟡
Minimal technical expertise needed	🟡	🟡	🔴	🔴	🟡	🟡	🟡	🟡	🔴	🔴	🟡	🔴	🟢	🟢
Deployment flexibility (range of platforms)	🔴	🔴	🔴	🟡	🟢	🟡	🟢	🟢	🔴	🟡	🔴	🟢	🟢	🟡
Suitable in deep waters (>30 m)	🟢	🟢	🟢	🟢	🟡	🟢	🟢	🔴	🟢	🟢	🟢	🟢	🟢	🔴
Suitable in high turbidity/sea states	🟢	🟢	🟢	🟢	🔴	🔴	🔴	🟡	🔴	🟡	🟡	🟢	🟢	🔴
Non-selective	🔴	🔴	🟢	🔴	🟢	🟢	🟡	🟡	🟡	🟢	🟢	🔴	🟢	🟡
Non-invasive/non-lethal	🔴	🔴	🟢	🟡	🟢	🟢	🟢	🟢	🔴	🟢	🟢	🟡	🟢	🟢
Autonomous	🔴	🔴	🔴	🟢	🟡	🟡	🔴	🟡	🔴	🟢	🟢	🟢	🔴	🔴
Low cost per sample unit	🟡	🟡	🔴	🟢	🟢	🟡	🔴	🟢	🔴	🔴	🔴	🔴	🟢	🟢

Table 5.2 General capacity of pelagic sampling methods to measure Essential Ocean Variables (EOVs) and Essential Biological Variables (EBVs) most relevant to pelagic monitoring. 🔴 = little capable, 🟡 = somewhat capable, 🟢 = mostly capable. TBM: Marine turtles, birds and mammals. Refer to **Table 1.1** for a description of each method. A = Active, P = Passive, M = Manned, U = Unmanned, Rem Sens = Satellite remote sensing (including photography).

	Capture		**Acoustics**		**Visual, optical & thermal**				**Robotics**		**Satellite**		**Genomics**	**Participatory**
	A	*P*	*A*	*P*	*Underwater*	*Airborne*	*Shipboard*	*Land*	*M*	*U*	*Rem. Sens*	*Tags*	*eDNA/biopsy*	*Citizens*
Essential Ocean Variables (EOVs)														
Phytoplankton diversity & abundance	🟡	🔴	🔴	🔴	🟡	🟡	🔴	🔴	🔴	🟡	🟡	🔴	🟡	🔴
Zooplankton diversity & abundance	🟡	🟡	🔴	🔴	🟡	🟡	🟡	🔴	🟡	🟡	🟡	🔴	🟡	🔴
Fish abundance & distribution	🟢	🟢	🟢	🟢	🟢	🟡	🟡	🔴	🟡	🟡	🔴	🔴	🟡	🟢
TBM abundance & distribution	🟡	🟡	🔴	🟢	🟡	🟢	🟢	🟡	🔴	🔴	🟡	🟢	🟡	🟢
Microbial activity, biomass & diversity	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🟡	🔴
Essential Biological Variables (EBVs)														
Genetic composition *	🟢	🟢	🔴	🔴	🔴	🔴	🔴	🔴	🟢	🟡	🔴	🔴	🟢	🔴
Species populations **	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟡	🔴	🟡	🟡	🟢	🟡	🟢
Species traits ***	🟢	🟢	🔴	🟡	🟡	🟡	🟡	🟡	🟡	🟡	🔴	🟢	🔴	🟡
Community composition †	🟢	🟢	🟡	🟡	🟢	🟡	🟡	🔴	🟡	🟡	🔴	🟡	🟢	🟢
Ecosystem function ¥	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🟡	🟢	🔴	🔴	🟡
Ecosystem structure ‡	🔴	🔴	🟡	🔴	🟡	🟡	🔴	🔴	🟡	🟡	🟢	🔴	🔴	🟡

***** Includes candidate EBVs co-ancestry, allelic diversity, population genetic differentiation, breed and variety diversity.
** Includes candidate EBVs species distribution, population abundance, population structure by age/size/class.
*** Includes candidate EBVs phenology, body mass, natal dispersion distance, migratory behaviour, demographic traits, physiological traits.
† Includes candidate EBVs taxonomic diversity, species interactions.
¥ Includes candidate EBVs net primary productivity, secondary productivity, nutrient retention, disturbance regime.
‡ Includes candidate EBVs habitat structure, ecosystem extent and fragmentation.

5.1 Monitoring and survey objectives

Sampling methods should be chosen to meet survey and monitoring objectives with efficiency, with a recommended emphasis on longevity and repeatability.

All method types (**Table 1.1**) are generally relevant for pelagic surveying, but their suitability is a function of the stage of monitoring and the purpose of data collection. A comprehensive marine monitoring programme will include aspects of all of the below goals:

- Baseline data are required to assess condition and provide a reference point for subsequent observations (Bouchet & Meeuwig 2015). This may also include multibeam mapping, which can inform the design of pelagic surveys and data interpretation (Bouchet *et al.* In review).
- The identification of critical habitats or taxa can guide management priorities and refine regions and metrics for monitoring activities (Maxwell *et al.* 2011). This objective can occur concurrently with the acquisition of baseline data (first point above).
- Trend detection is made possible by fulfilling the previous objectives and requires repeat sampling. Knowledge of biological trends can help inform marine spatial planning, MPA enforcement, and other management strategies, but needs to be considered in light of inherent uncertainty as trends are typically only derived from limited subsets of possible observations. Trend detection will thus be more readily achievable where reliable baseline information is available, from a range of survey approaches.

For instance, remote sensing methods provide a baseline map of oceanographic conditions, from which appropriate sampling designs can then be implemented, although repeating remote sensing surveys may not be needed on subsequent surveys to detect change. Capture sampling yields valuable biological specimens, particularly in remote, unexplored areas, from which a species inventory can be derived to inform subsequent change detection. Non-extractive methods such as visual surveys, underwater imagery and unmanned robots are currently the most appropriate methods to detect change and quantify impacts due to their capacity to collect true repeat observations at generally limited costs, which increases efficiency when estimating trend. Imagery also provides a permanent record of a snapshot in time with minimal interference, compilations of which can then be used to detect trends.

5.2 Environment

Pelagic habitats are highly dynamic and three-dimensional, with currents and operational depth playing enormous roles in defining the range of sampling methods appropriate for a given location/habitat. Some methods are inevitably constrained to the ocean surface or the upper layers of the water column (e.g. aerial/shipboard visual surveys, LiDAR, satellite remote sensing) or to shallow depths (e.g. UVC, DOV), whereas others can function across multiple bathomes (Drop cameras, gliders, floats, passive acoustic loggers). All imagery systems, in particular, must be fitted in underwater housings with appropriate pressure ratings, and equipped with light systems as appropriate.

5.3 Equipment availability

Marine sampling gears vary in their level of complexity and market availability in Australia and internationally. For instance, while UVC usually requires no specialised equipment other than standard dive apparel (and a transect reel, underwater slate and camera), AUVs and the larger ROVs are far more complex and expensive, requiring time and expertise to build. Other methods like pelagic BRUVS entail specialised but simple equipment that is relatively easy to source from established research groups or even build if raw materials are available.

5.4 Expert knowledge

Several sampling methods are reliant on experts for successful data acquisition and treatment. For instance, UVC/aerial/shipboard surveys require certified divers/observers trained to identify species *in situ*, although the rise of recreational diver/observer-based citizen science programmes suggests that minimal training can be sufficient to conduct successful monitoring. ROVs, LiDAR, UAVs, or active acoustic systems often require technicians for installation, calibration, and deployment, and the availability of these experts can become a constraint. By contrast, pelagic BRUVS, drop cameras, theodolites, or eDNA do not require complex calibration and can be operated by the crew of a vessel following standard operating procedures.

5.5 Costs and resources

Monitoring is costly in a resource-constrained world, and managers are thus often faced with making difficult decisions about how to best allocate limited funds between data acquisition and on-the-ground action (Jones *et al.* 2013). Strategic monitoring must therefore include a necessary trade-off between the costs (measured in time and/or money) and the value of additional information gain relative to existing knowledge (Gerber *et al.* 2005). Understanding the nature of these costs and benefits is vital to evaluating them in a rational context that accounts for the urgency of conservation problems (McDonald-Madden *et al.* 2010). For instance, both optimal sampling methods and optimal levels of monitoring effort may vary through space and time (e.g. depending on the state of the system) (Hauser *et al.* 2006), and in rare cases, the best use of resources may actually be to forgo monitoring altogether. This is more likely to be the case if the cost of monitoring and/or the risk of incorrect inference are high, and alternative, more affordable management options exist that can be put into effect immediately (Di Fonzo *et al.*). As a result, it cannot be assumed that established or popular methods are always the most cost-effective and suitable for surveying pelagic biodiversity (Costello *et al.* 2017).

The true expenditure incurred by each sampling method is challenging to estimate with accuracy, consistency, and generality, as it depends on a range of method-specific factors such as equipment purchase or hire, supplies and consumables, calibration, maintenance and repair, vessel time, staff time, fuel consumption, health and safety risks, sample or data processing, capital depreciation, and personnel training. For example, basic UAV models can sell for under USD $2,000, whereas military-grade, multi-sensor drones can easily be valued at six figures or more (Pimm *et al.* 2015). Similarly, the purchase price of thermal imaging systems can go from USD $20,000 to over $200,000 USD (Verfuss *et al.* 2018). Cost-benefit analyses for pelagic sampling are rare, and should be further encouraged. Additional information on the published costs of various methods can be found below.

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6. Conclusions and recommendations

It has become abundantly clear that successful marine management cannot occur without effective monitoring (Day 2008). Until recently, pelagic sampling at broad spatial or taxonomic scales was a prohibitively expensive and nearly impossible logistical task (Williams 1995), made all the more difficult by the wide range of temporal (seconds to years) and spatial (cm to thousands of km) scales over which open ocean processes routinely take place. Rapid developments in marine instrumentation, however, have transformed the way in which data are collected at sea, earmarking significant advances in our capacity to target previously inaccessible pelagic habitats in coming years (Bean *et al.* 2017).

No single device is capable of sampling all organisms and oceanographic processes (Woodall et al. 2018). A universal, one-size-fits-all approach to pelagic monitoring is thus not feasible, nor necessarily desirable. Rather, effective whole-of-ecosystem monitoring (i.e. across habitats, body sizes and trophic levels) will demand holistic approaches to sampling that capitalise on the combined strengths of multiple instruments (Costello et al. 2017), deployed within coordinated networks (Suberg et al. 2014). Examples of the successful implementation of such networks already exist in the California Current Ecosystem, for instance, where satellite remote sensing imagery, shipboard surveys, gliders, and oceanographic floats are used in tandem to provide compatible insights into ecosystem dynamics at various spatio-temporal resolutions (Ohman *et al.* 2013).

Australia is well placed to follow suit by embracing a wide array of novel pelagic monitoring technologies as they become further available and practicable (Hedge 2016). A pervasive hurdle, however, will be to ensure that these can be appropriately benchmarked and validated (Danovaro *et al.* 2016), such that information gathered by a variety of equipment types and a multitude of actors can be successfully integrated to answer policy-relevant questions (Jones *et al.* 2013). As such, multi-gear sampling strategies represent both an opportunity and a challenge. On the one hand, combining methods can help ground-truth observations, maximise efficiency, and increase returns on investment when typical funding horizons limit most ecological research to short time frames (Giron-Nava *et al.* 2017). On the other hand, concurrent deployments require a higher level of effort and resources that may exceed what is possible, depending on specific goals.

Although the simultaneous use of several methods is increasingly common, their best combination remains unknown, in great part because optimal pairings depend on the target organisms, the habitats being explored, and the underlying survey objectives (Jech *et al.* 2009). In practice, method choice is often be driven by geographical origin and institutional attributes (e.g. academic tradition or region-specific training) (Caldwell *et al.* 2016). Undeniably, progress in marine sampling will therefore be fostered by international, cross-institutional collaborations (Bouchet *et al.* 2018a), and can be further encouraged by the adoption and dissemination of standard field/analysis protocols that can accommodate historical (legacy) datasets/surveys (Foster *et al.* 2017, Przeslawski & Foster 2018).

Our hope with this comparative assessment was to deliver a broad synthesis of the advantages and caveats of commonly used pelagic methods that can be used to guide pelagic sampling activities. This is crucial to guarantee cost-effectiveness, repeatability, objectivity, and transparency in monitoring programmes. Irrespective of the sampling methods chosen, a premium should always be placed on robust survey designs to enable meaningful comparisons over time and space.

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Appendix 1. Online questionnaire on pelagic sampling methods.

NESP D2 - Pelagic sampling methods questionnaire

Introduction and background

The Commonwealth Scientific and Industrial Research Organisation (CSIRO, https://www.csiro.au/) and the Australian Government's National Environmental Science Programme (NESP) Marine Biodiversity Hub (https://www.nespmarine.edu.au/) invite you to take part in a questionnaire about pelagic sampling methods. We are seeking input from scientists, technicians and managers to gauge their use and perceptions of various marine sampling equipment and methods. This information will support the development of appropriate field manuals and standard operating procedures for marine monitoring as part of a national NESP project (Project D2 - Analysis methods and software to support Standard Operating Procedures for survey design, condition assessment and trend detection; https://www.nespmarine.edu.au/project/project-d2-analysis-methods-and-software-support-standard-operating-procedures-survey-design).

Background

In an era of unprecedented concern about global biodiversity loss, marine researchers, managers and policy advisors constrained by both diminishing budgets and rising pressures to build accountability must now more than ever design monitoring programmes that are not only robust but also cost-effective. A vast number of modern tools are available for surveying ocean habitats and wildlife, however choosing among them can be difficult as most differ widely in costs, accessibility, capabilities, mobilisation constraints, resolution or sensitivity, and are evolving rapidly without always being critically evaluated or compared.

In response to this, scientists from the Australian Government's National Environmental Science Programme Marine Biodiversity Hub - Project D2 are **undertaking a detailed comparative assessment of sampling methods used in marine monitoring applications.** The work aims to guide the development of standard operating procedures that can support the collection of consistent, comparable, interpretable and fit-for-purpose empirical evidence for assessing status and trends in ocean ecosystems.

Examples of pelagic methods considered in this survey

Key to achieving this objective is a **fundamental understanding of the current patterns of use, perceptions, and awareness of various sampling gears.** In capturing these, the following questionnaire aims to determine common motivations for, or obstacles to, selecting given methods. We also aim to highlight potential synergies and variation between institutions, programmes, regions or times that can inform future standardization strategies.

What will I be asked to do?

Completing the survey will take **approximately 15 minutes.**

The questions relate to respondents' marine survey experience, equipment use, and perceptions. **Note that our focus here is on pelagic sampling/monitoring**, meaning that benthic methods (e.g. sleds, bottom trawls, towed video etc.) will not be considered. A list of relevant pelagic methods has been established from a review of the published literature and consultation with experts. However, we encourage you to provide responses in relation to any other method that may not be listed. Your participation is voluntary, and you are free to withdraw by stopping at any time. If you decide to withdraw from the survey, any responses you have provided up to that point will be deleted. You may also skip any question you do not want to answer.

How will the results of the study be used?

All information collected through the survey will be anonymous, unless you choose to identify yourself, in which case your identity will only be available to the project leaders. Data will be aggregated to ensure participants are not identifiable. Results from the questionnaire will inform the completion of NESP Project D2 and provide focus for the production of standard operating procedures and field manuals for selected sampling methods. Results may be released in NESP publications, and will be available to participants upon request.

Whom do I contact about this questionnaire?

Questions can be directed to the Project Leaders Dr. Rachel Przeslawski (Geoscience Australia, rachel.przeslawski@ga.gov.au) and Dr. Scott Foster (CSIRO, scott.foster@csiro.au) and/or survey coordinator Dr. Phil Bouchet (University of Western Australia, phil.bouchet@uwa.edu.au).

This study has been approved by CSIRO's Social Science Human Research Ethics Committee, in accordance with the National Statement on Ethical Conduct in Human Research (2007). Any concerns or complaints about the conduct of this study can be raised with the Manager of Social Responsibility and Ethics on (07) 3833 5693 or by email at csshrec@csiro.au.

NESP D2 - Pelagic sampling methods questionnaire

About you

Q1. What is your current role?

- Senior/project scientist
- Postdoctoral researcher
- PhD student
- Masters student
- Undergraduate student
- Engineer
- Technician
- Manager
- Consultant
- Other (please specify)

Q2. What type of institution do you currently work for?

- University
- Government (research)
- Government (non-research)
- Consultancy/private company
- Non-profit
- Unemployed
- Other (please specify)

Q3. In what country are you based?

Q4. In what city are you based?

Q5. How many marine expeditions have you taken part in?

NESP D2 - Pelagic sampling methods questionnaire

Sampling locations

Q6. Where do you undertake most of your fieldwork?

- Australia
- Overseas
- Both

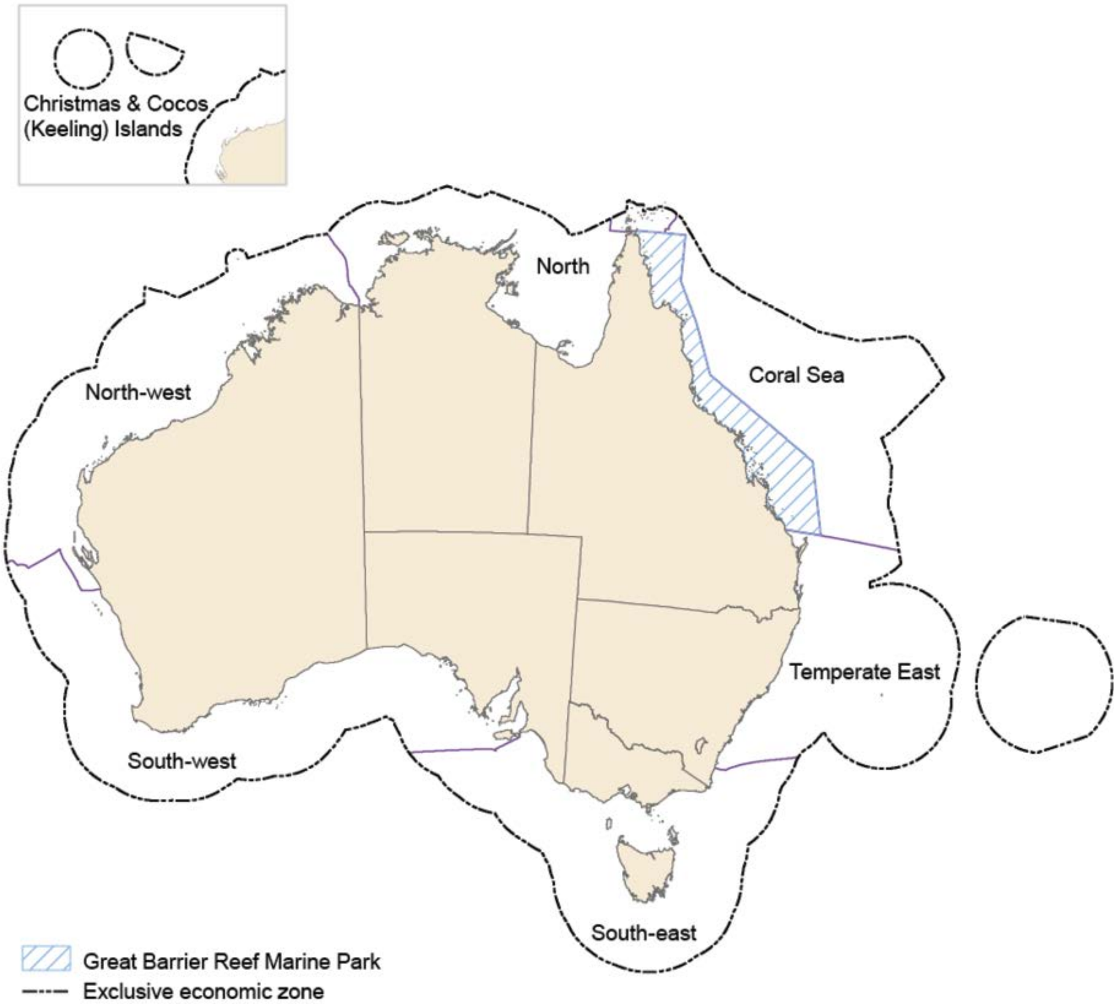
NESP D2 - Pelagic sampling methods questionnaire

Sampling locations (Australian marine regions)

Q7. In what Australian marine region(s) does most of your fieldwork occur? (select all that apply)

A map of Australian marine regions is shown below for reference

- Coral Sea/GBR
- North
- North-West
- South-West
- South-East
- Temperate East
- Polar
- Other (please specify)



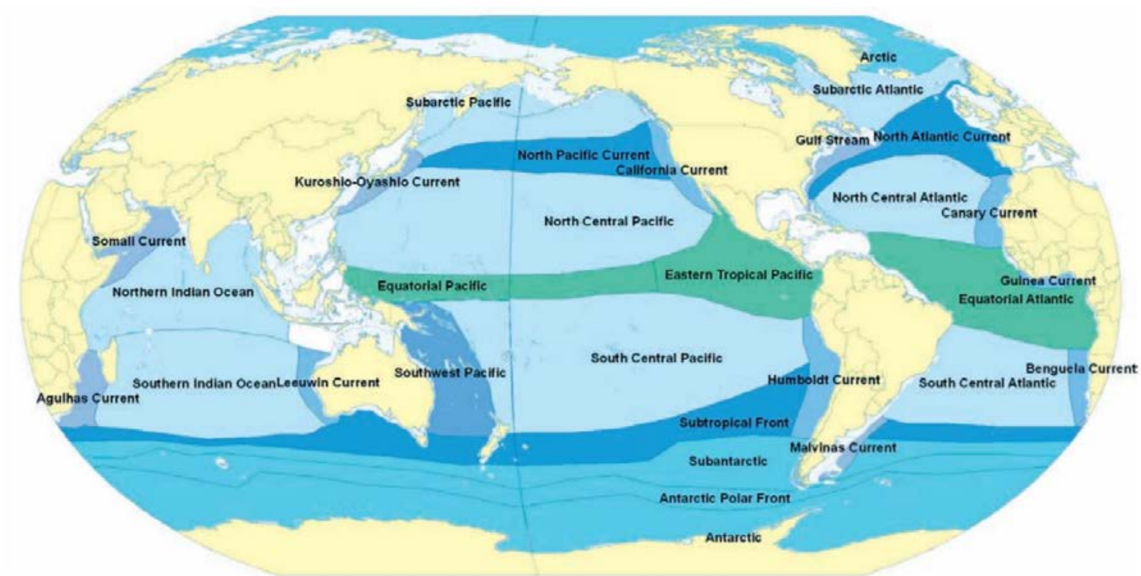
NESP D2 - Pelagic sampling methods questionnaire

Sampling locations (Pelagic Provinces)

Q8. In which Pelagic Province(s) does your fieldwork normally occur? (select all that apply)

Pelagic Provinces from the Global Open Ocean and Deep Seabed (GOODS) biogeographic classification are shown below for reference

- Arctic
- Subarctic Pacific
- Subarctic Atlantic
- Gulf Stream North Atlantic Current
- North Central Atlantic
- South Central Atlantic
- Canary Current
- Guinea Current
- Equatorial Atlantic
- Benguela Current
- North Pacific Current
- California Current
- North Central Pacific
- Humboldt Current
- Subtropical Front
- Malvinas Current
- Equatorial Pacific
- Kuroshio-Oyashio Current
- Southwest Pacific
- Somall Current
- Northern Indian Ocean
- Southern Indian Ocean
- Leeuwin Current
- Agulhas Current
- Subantarctic
- Antarctic Polar Front
- Antarctic



NESP D2 - Pelagic sampling methods questionnaire

Sampling locations (Pelagic Provinces)

Q9. In what environments do you undertake most of your fieldwork? (select all that apply)

- Coastal (intertidal/subtidal)
- Shelf shallow (<50m)
- Shelf (50-200m)
- Slope (200-1000m)
- Deep sea (>1000m)
- Other (please specify)

NESP D2 - Pelagic sampling methods questionnaire

Pelagic methods (awareness)

Q10. What pelagic sampling method(s) are you aware of? (select all that apply)

NESP D2 - Pelagic sampling methods questionnaire

Pelagic methods (usage)

Q11. Of these, which do you actually use or have use(d)? (select a maximum of 5)

NESP D2 - Pelagic sampling methods questionnaire

Method development

Q12. How were these methods developed?

- Designed by yourself or within your home institution
- Adopted from another research institution or project
- Adopted *and modified* from another research institution or project

NESP D2 - Pelagic sampling methods questionnaire

History of use

Q13. How long have you been using these methods?

- < 1 year
- 1-2 years
- 2-5 years
- 5-10 years
- > 10 years

NESP D2 - Pelagic sampling methods questionnaire

Past, present and future use

Q14. How often have you used the following methods in the past?

- Rarely
- Sometimes
- Often
- Very often/always

Q15. How often do you currently use the following methods?

- Rarely
- Sometimes
- Often
- Very often/always

Q16. How often will you use the following methods in the future?

- Rarely
- Sometimes
- Often
- Very often/always

Q17. If your use of pelagic methods has changed over time or is expected to change, please indicate when this did/will occur and any reasons why.

Example: I used to conduct regular boat-based visual surveys before 2003 but have been using pelagic BRUVS since then as they are more effective at monitoring species X and Y.

NESP D2 - Pelagic sampling methods questionnaire

Monitoring objectives

Q18. What is the purpose of the data you are collecting? (select all that apply)

- Long-term monitoring
- In-house research
- Masters or PhD thesis
- Rapid ecological assessment
- Initial (baseline) survey
- Assessment of management effectiveness (e.g. MPAs)
- Fisheries stock assessment
- Assessment of human impacts on habitats/populations
- Estimates of species abundance/distribution/relationships with habitat
- Methodological/sampling design improvements
- Spatial planning (e.g. guiding the design/location of MPAs)
- Pure exploration (e.g. geological, ecological, biological, oceanographic)
- Habitat mapping

Q19. What kind of sampling do you mostly conduct?

We define dedicated samples as those derived from following a rigorous, probabilistic design (examples include random, stratified, or adaptive sampling). Opportunistic sampling, on the other hand, is a non-probabilistic form of data collection occurring *ad hoc* and spontaneously (including convenience, accidental, ad libitum, and preferential sampling).

- Dedicated
- Opportunistic

NESP D2 - Pelagic sampling methods questionnaire

Monitoring variables – biology and ecosystem

Q20. Expert panels from the Global Ocean Observing System (GOOS) have developed a list of Essential Ocean Variables (EOVs) deemed critical for ocean observation and monitoring in a global context.

Please indicate which biological and ecosystem EOVs you monitor with pelagic methods (select all that apply).

If your variable isn't listed, please tick 'Other' and provide more information in the comment box at the bottom of the page

- Fish abundance and distribution
- Marine turtles, birds, mammals abundance and distribution
- Zooplankton biomass and diversity
- Phytoplankton biomass and diversity
- Other

NESP D2 - Pelagic sampling methods questionnaire

Monitoring variables – biogeochemistry

Q21. Expert panels from the Global Ocean Observing System (GOOS) have developed a list of Essential Ocean Variables (EOVs) deemed critical for ocean observation and monitoring in a global context.

Please indicate which biogeochemical and ecosystem EOVs you monitor with pelagic methods (select all that apply).

If your variable isn't listed, please tick 'Other' and provide more information in the comment box at the bottom of the page

- Oxygen
- Nutrients
- Inorganic carbon
- Transient tracers
- Suspended particulates
- Nitrous oxide
- Stable carbon isotopes
- Dissolved organic carbon
- Ocean colour
- Other

NESP D2 - Pelagic sampling methods questionnaire

Monitoring variables – physical oceanography

Q22. Expert panels from the Global Ocean Observing System (GOOS) have developed a list of Essential Ocean Variables (EOVs) deemed critical for ocean observation and monitoring in a global context.

Please indicate which physical oceanography and ecosystem EOVs you monitor with pelagic methods (select all that apply).

If your variable isn't listed, please tick 'Other' and provide more information in the comment box at the bottom of the page

- Sea state
- Ocean surface stress
- Sea surface height
- Sea surface/subsurface salinity
- Sea surface temperature
- Surface/subsurface currents
- Other

NESP D2 - Pelagic sampling methods questionnaire

Frequency of use

Q23. If your data are collected through time, how often do you conduct surveys using these methods, and why?

- Daily
- Weekly
- Biweekly (every fortnight)
- Monthly
- Bimonthly (every 2 months)
- Quarterly
- Semi-annually
- Annually
- Other (please specify)

NESP D2 - Pelagic sampling methods questionnaire

Time of day

Q24. What time of day do your surveys typically occur?

- Daytime
- Night-time
- Both

NESP D2 - Pelagic sampling methods questionnaire

Pros and cons

Q25. Please indicate what you see as the main advantages of the methods you use.

Q26. Please indicate what you see as the main drawbacks of the methods you use.

NESP D2 - Pelagic sampling methods questionnaire

Ownership

Q27. Do you own the survey methods that you use?

- Yes
- No, I rent them but operate them myself
- No, I lease them out and the contractor operates them
- Combinations of the above

NESP D2 - Pelagic sampling methods questionnaire

Purchase and operation costs

Q28. If you own any sampling methods, what was their (approximate) initial purchase cost?

- 0-$1,000
- Up to $5,000
- Up to $10,000
- Up to $50,000
- Up to $100,000
- >$100,000

Q29. What would you say are their typical maintenance costs/daily costs to operate?

- 0-$1,000
- Up to $5,000
- Up to $10,000
- Up to $50,000
- Up to $100,000
- >$100,000

NESP D2 - Pelagic sampling methods questionnaire

User perceptions

Q30. Please rate methods based on your perceptions of total cost (including post-processing of data), particularly relative to other sampling methods.

- Low
- Moderate
- High
- Prohibitive

Q31. Please rate methods based on your perceptions of operational complexity, particularly relative to other sampling methods.

- Low
- Moderate
- High
- Prohibitive

Q32. Please rate methods based on your perceptions of post-processing or technical requirements, particularly relative to other sampling methods.

- Low
- Moderate
- High
- Prohibitive

Q33. Please rate methods based on your perceptions of risks (health and safety, expense, and potential loss), particularly relative to other sampling methods.

- Low
- Moderate
- High
- Prohibitive

NESP D2 - Pelagic sampling methods questionnaire
Opportunities

Q34. What would you improve in each method to reduce costs and optimise monitoring efficiency and/or data quality? What gaps in capability and availability can you identify?

Please indicate which (if any) additional method(s) you anticipate using in the future.

NESP D2 - Pelagic sampling methods questionnaire
Drivers of method choice

Q35. What are the minimum needs a method should meet to be useful for your applications?

Q36. What are your primary considerations/motivations when selecting a method? (select all that apply) Costs (purchase, maintenance, operation)

- Personnel and analytical requirements
- Operating limitations (depth constraints, weather capabilities, terrain abilities, support ship requirements, thruster power, ability to work in currents, navigation abilities, manoeuvrability)
- Target organisms and biological sampling capability
- Market availability (whole or parts)
- Popularity and prominence in the published literature
- Ease of deployment/recovery
- Quality/quantity/type of output data
- Biases and measurement precision
- Longevity and reliability
- Technical specs (data storage capacity, power requirements, interchangeability/upgradability of hardware/software)
- Ease of shipping to remote locations
- Management needs
- Availability of automated analysis methods
- Invasiveness/potential for disturbance to wildlife/habitats
- Resolution and coverage (temporal, spatial, taxonomic)
- Short turnaround time (from deployment to analysis)
- Access targeted habitats
- Comparability/consistency with historical data or data from other research institutions or projects
- Best fit for the research questions being asked
- Only method known/learnt
- Other (please specify)

NESP D2 - Pelagic sampling methods questionnaire

Standard operating procedures

Q37. In your opinion, would an Australian national standard for pelagic monitoring be valuable?

- Yes
- No
- Unsure

Q38. If not, why not?

- There is no need for datasets to be comparable
- The data generated by different research programmes are not incomparable, even though the same gear was used
- Changing current methods would compromise long-term/historical datasets
- The effort required would be too great (e.g. personnel training, upgrade of equipment)
- Each method is unique, and the questions they help answer are not comparable
- Other (please specify)

Q39. To what extent would you be willing to modify your current methodology to adopt a standardized pelagic monitoring protocol, should one be proposed?

1 (not willing to change) 2 3 4 5 6 7 8 9 10 (adopt new method)

Q40. What factors would increase your interest in/likelihood of adopting it?

- Minimal cost
- Succinct and clear protocols
- Flexible (e.g. for different environments, taxa etc.)
- Incorporates my current approach (legacy value)
- Agreement among experts (i.e. a majority of colleagues adopted the method as well)
- Ability to re-use or pool data (i.e. ensure that existing data would not become irrelevant)
- National data resource (e.g. spatial and temporal extension)
- Defensibility
- Made data more comparable to other large datasets
- If standardized method continued to answer the existing research questions of the project
- Other (please specify)

NESP D2 - Pelagic sampling methods questionnaire

Comparative assessments

Q41. Are you aware of any entities, groups, institutions etc. that provide methods, protocols, SOPs or technical guidance for any of the methods mentioned here? If this includes yourself, would you be willing to make these available for use within the context of NESP D2?

Q42. Do you know of any reports, peer-reviewed publications, websites, that compare any two or more of the pelagic methods covered in this questionnaire? Please provide details.

NESP D2 - Pelagic sampling methods questionnaire

Last but not least

Q43. Please enter your full name and email address (optional)

Q44. Do you agree to being acknowledge in any report(s) produced by NESP as a result of this survey?

Q45. Would you like to receive a summary of the results from the survey?

- Yes
- No

Q46. Do you have any outstanding comments or feedback?

Contact

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