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Comparative assessment of seafloor sampling platforms

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Project D2:
Standard Operating Procedures (SOP) for survey design, condition assessment and trend detection

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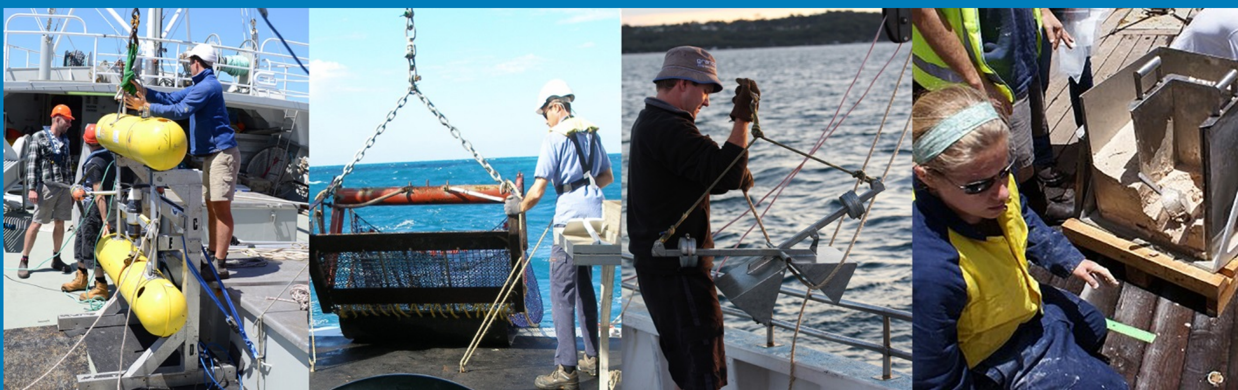

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Important Disclaimer

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EXECUTIVE SUMMARY

The Australian Marine Parks are the largest network of marine protected areas in the world, and their establishment means that Australia is now tasked with managing an area almost 3.3 million km^2. In addition, Australia has the third largest exclusive economic zone in the world, with an extensive geographic area on which to report for State of Environment. The vastness of Australia's marine estate means that appropriate, efficient, and comparable sampling methods are crucial to meet management and reporting obligations.

The overarching objectives of environmental monitoring are to assess condition and detect trends, and numerous marine sampling platforms exist to acquire data to meet these needs. It is daunting to consider all marine sampling platforms in the context of a single monitoring program and to ensure that the most appropriate methods are used for a given purpose. There is thus a need to synthesise and compare these platforms as they relate to the design and implementation of monitoring programs.

The purpose of the current study is to describe and comparatively assess common seafloor sampling platforms. We do this by conducting a qualitative assessment and comprehensively reviewing the available literature to identify their potential limitations and advantages. For the purposes of this report, marine sampling platforms include those that acquire seafloor data using underwater equipment or methods. We focus on sampling platforms near (i.e. demersal) or at (i.e. benthic) the seafloor because the habitat and associated biota targeted by these platforms are usually fixed and can be revisited, making them well-suited to monitoring activities.

This report is divided into four sections, as well as an introduction (Section 1):

- Section 2 describes each major benthic and demersal biological sampling platform, including their advantages, disadvantages, and innovations. These include acoustics platforms (e.g. multibeam echosounder (MBS), sidescan, single-beam), visual methods (e.g. autonomous underwater vehicle (AUV), baited remote underwater vehicle (BRUV), towed imagery, underwater visual census (UVC)), and direct sampling (e.g. ROV, sleds, dredges, corer, grabs).
- Section 3 describes the use and perceptions of six benthic and demersal sampling platforms (AUV, BRUV, MBS, towed imagery, sleds/trawls, grabs/box corers) via results from an online questionnaire released on 15 Dev 2016 to gauge use and perceptions of common marine sampling platforms in Australia. A total of 49 people completed the questionnaire, and three platforms were frequently used by a large proportion of respondents: MBS (42.5%), grabs/boxcores (41%), and towed imagery (40%). Highest perceptions of cost and deployment effort were associated with the AUV and MBS.
- Section 4 presents results from a literature review in which we searched for studies that used two or more marine benthic or demersal biological sampling platforms, excluding acoustics methods. We then refined this search to include studies that either i) directly compared methods (50 studies) or ii) tested for similar ecological relationships among two or more gear types (42 studies). Based on direct comparisons, the platforms with the least similarity between them may be operator-based direct sampling and sled/trawl, operator-based imagery acquisition and UVC, and UVC and BRUVs. Based on ecological

congruence, data from sleds/trawls and grabs/corers showed similar ecological patterns, while UVC and BRUV and UVC and grabs/corers may be the least ecologically congruent.
- Section 5 relates our results to marine monitoring by linking each sampling platform to its capability to measure global indicators (Essential Ocean Variables, Essential Biodiversity Variables). We also provide further advice on choosing an appropriate sampling platform as related to monitoring program objectives, target environment, and available resources including cost.

Our study confirms that marine surveys are undertaken to acquire baseline environmental data, identify important habitats or taxa, or detect change (including quantifying impacts), each of which is associated with optimal survey designs and sampling platforms. A comprehensive marine monitoring program can include aspects of all of these goals. For example, seafloor acoustic methods provide a baseline map of the seabed from which a powerful and appropriate survey design can then be implemented. On subsequent surveys to detect change, however, such methods may not be needed unless an assessment of seabed stability and geohazards is required. Direct sampling yields valuable biological specimens, particularly in unexplored areas, from which a species inventory can be derived to inform subsequent change detection. Non-extractive methods such as underwater imagery and visual censuses are currently the most appropriate methods to detect change and quantify benthic impacts due to their capacity to collect true repeat observations, which increases efficiency when estimating the 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.

There is no universal method appropriate for all marine sampling; a one-size-fits-all approach is neither feasible nor desirable in monitoring programs. For surveys collecting baseline or descriptive information, a diversity of gear may be more appropriate, while for monitoring surveys, fewer platforms capable of repeatable sampling would be more appropriate. This comparative assessment provides information that can be used to guide marine sampling activities as they relate to monitoring objectives. Such information is crucial to ensure cost-effectiveness and efficacy of marine monitoring activities, specifically that the best methods are being used with appropriate knowledge of limitations and challenges. In addition to the marine sampling platforms that are chosen, robust survey designs and standard operating procedures are crucial to ensure consistency of data and comparability over time and space.

1. INTRODUCTION

Environmental monitoring requires sampling of biological and physical factors over space and time in order to assess status and detect trends, including how well management practices are protecting ecosystems. For marine protected areas, there are many methods used to sample benthic biota and environments, some of which are frequently used and well-established (Hopkins 1964), and others that are new or contentious (Rhoads et al. 2001). For biological sampling at or near the seafloor, sampling methods include destructive epifaunal samplers (sleds, trawls, dredges), destructive infaunal samplers (grabs, corers, some sleds), and non-destructive samplers (imagery systems, visual census) (Bowden et al. 2015). There is also a range of acoustic methods that can be used to map the seafloor (multibeam sonar, sidescan sonar, single-beam sonar) (Rees et al. 2014) and occasionally even provide biological data (Foote et al. 2006, Cook et al. 2008). These seafloor acoustic methods are the foundation for monitoring activities in large regions, as they facilitate extensive and precise descriptions of physical habitats (Bax et al. 1999). A number of established protocols for benthic and demersal marine sampling exist for various regions, habitats, and objectives (reviewed in Coggan et al. 2005) and have been nationally standardised and implemented for shallow Australian waters and reefs within their respective programs (e.g. Survey Manual for Tropical Marine Resources in English et al. 1997, Reef Life Survey in Stuart-Smith et al. 2017). It is daunting to consider all marine sampling platforms in the context of a single monitoring program and to ensure that the most appropriate methods are used for a given purpose. There is thus a need for a synthesis and comparison of benthic marine sampling platforms as they relate to the design and implementation of monitoring programs.

The overarching objectives of environmental monitoring are to assess condition and detect trends, hopefully with attribution. These objectives are sometimes expressed as hypothesis-driven research, including relationships to changing environmental conditions (Lantz et al. 2014, Vethaak et al. 2017), impact of human activities (Bowden et al. 2015) or efficacy of management strategies (Kelaher et al. 2014). Results from such research then inform evidence-based decision making and reporting (Hayes et al. 2015). Australia's management plans for Australian marine parks specify a commitment to adaptive management, including consideration of 'the effectiveness of monitoring and evaluation, and the appropriateness of key indicators and performance measures'. There is a similar focus at a global scale on the identification of standard and suitable indicators through the GEO-BON (e.g. Essential Biodiversity Variables in Turak et al. 2017)) and GOOS (Essential Ocean Variables in Miloslavich et al. 2018) programs.

Marine management plans and associated monitoring programs often target biological variables, specifically those related to biodiversity (Stuart-Smith et al. 2017), habitat provision (Bell et al. 2015), or invasive (Darling and Frederick 2017), commercial or vulnerable (Burtenshaw et al. 2004) species. Abiotic (i.e. physical) environmental data are also extremely valuable to monitoring programs and can inform causal relationships between pressures and biological values. Whereas biological data can reveal changes over space and time, abiotic environmental co-variates can help to provide explanations for such patterns and guide future scientific research and management strategies (Edgar et al. 2014). For this reason, an important attribute of many of the marine sampling platforms included in this report is that they also collect abiotic data. Multibeam sonar (MBS) is a common benthic mapping sensor but does not provide direct quantitative assessment of benthic biological species data (outside of potential biomass estimates) (but see Colbo et al. 2014 for MBS capacity to provide pelagic biological data). It is, however, valuable to monitoring programs because of the continuous seafloor maps derived from it which provide the detailed fine-scale topography and broader topographic context necessary to interpret and monitor the distribution of substrate-attached fauna. Other platforms, such as AUVs and grabs, can be purposely deployed for the concurrent collection of both biotic and abiotic data, thereby reducing spatial variability and increasing statistical power to detect spatial and temporal patterns.

Marine sampling platforms each target a particular data type and/or faunal group, and it becomes challenging to separate differences due to platform-specific functions (e.g. point vs transect) and taxa-specific responses. The only way to separate such effects is to maintain consistency within the data type and target taxa across all sampling platforms, and since this is counter-productive to the rationale often applied to using multiple platforms, it is only rarely done (Section 4.2.2). Other studies have addressed this challenge by standardising data among discrete deployments of different gear (e.g. proportional data from BRUVs and UVCs in Colton and Swearer 2010, Lowry et al. 2012). In the current study, we assume that sampling gear is chosen due to its ability to assess a target taxon. Researchers accept that they need to select a gear type prior to measuring marine fauna, but there is little information to assist in this selection. However, the choice of platform will greatly influence the observed spatial and ecological patterns, so platforms need to be chosen to match the individual research question or study objective (Bowden and Hewitt 2012).

1.1 Objectives & Scope

As part of the National Environmental Science Program (NESP), Marine Biodiversity Hub Project D2 (‘Standard Operating Procedures (SOPs) for survey design, condition assessment and trend detection’), this report will compare benthic and demersal marine sampling platforms, including their suitability for use with different monitoring objectives. A complementary comparative assessment report is under development for pelagic platforms (Bouchet et al. 2017). For the purposes of this report, marine benthic and demersal sampling platforms include those that acquire seafloor data using underwater equipment or methods. We do not include satellite imagery in this report due to its vastly differing spatial extent and limited applicability outside shallow clear waters. Neither do we include acoustic telemetry, as this is usually employed for the spatial mapping of pelagic organisms.

There have been numerous reviews that assess discrete marine sampling platforms, including BRUVs (Whitmarsh et al. 2017), AUVs (Nicholson and Healey 2008, Wynn et al. 2014), and grabs and corers (Blomqvist 1991b). Others address a topic that spans several platforms (e.g. habitat mapping in Bax and Williams 2001, Brown et al. 2011, marine imagery in Durden et al. 2016b) or multiple discrete assessments (Eleftheriou and Mcintyre 2005, Danovaro 2010, Clark et al. 2016). The purpose of the current study is to describe and comparatively assess common benthic and demersal marine sampling platforms. We do this by conducting a qualitative assessment and comprehensively reviewing the available literature to identify potential limitations and advantages of platforms. Results will indicate if broad scale biodiversity patterns are likely to be consistent among datasets derived from different sampling equipment and which combination of sampling gear provides the most reliable results for biodiversity assessments. This assessment is intended to help inform the decision to employ particular marine sampling platforms in monitoring programs, thus leading to appropriate SOPs (e.g. NESP Field Manual package) to ensure data are comparable over space and time.

1.2 Format

This report is divided into four further sections:

- Section 2 describes each major benthic sampling platform, including major advantages, disadvantages, and innovations.
- Section 3 describes the use and perceptions of marine benthic sampling platforms via results from an online questionnaire.
- Section 4 details the literature review and the associated qualitative comparative assessment.
- Section 5 relates our results to marine monitoring.

2. DESCRIPTION OF BENTHIC SAMPLING PLATFORMS

In this section, we describe the advantages and limitations of common benthic sampling platforms. We have grouped these platforms in three broad categories based on the data type collected (Acoustics – geophysics data, Direct sampling – specimen data, Visual method – imagery or observation data). In addition, we summarise the variation within a single broad category. The platforms considered and their general characteristics are listed in Table 1.

2.1 Acoustics (multibeam, single-beam, sidescan)

Marine acoustic technologies use methods that create data based on the interaction between underwater sound waves and physical obstacles. Targets can be revealed either within the water column or on the seafloor. Marine acoustic sampling is used to create measurements of water depth and substrate distribution. Data acquisition involves a transmitter which sends out a sound pulse and a receiver which listens and receives this sound pulse. The difference in time between the send and receive signal will result in a depth measurement and the strength of the return as a ration of the emitted pulse will result in an impedance (or backscatter value) (reviewed by Foote 2009, Colbo et al. 2014). Bathymetric data provides information on depth, while backscatter data measures seabed reflectance and accordingly provides information on substrate hardness or objects in the water column. Using post-processing methods in specialised software, spatial surfaces can be produced on the depth distribution and seafloor composition within a survey area. Common marine acoustic systems that target the seafloor include single-beam sonar, sidescan sonar, and multibeam sonar (MBS). All of these systems can be vessel mounted or deep-towed, the latter for higher resolution. Sidescan sonar produces backscatter and associated pictures of the seafloor, but does not generate direct bathymetric data. Single- and multi- beam sonar systems collect both bathymetric and backscatter data

The main benefit of marine acoustic platforms to benthic monitoring is their capacity to generate high resolution maps of the physical features of the seafloor over a broad spatial area. These maps can then be used to identify key seabed features (e.g. paleoshorelines in Brooke et al. 2017) or choose sampling locations (Bax and Williams 2000, Brown et al. 2002), and they can also be combined with biological data to produce habitat maps (Brown et al. 2011). Acoustic techniques can also be used to detect spatial extent of broad ecological communities such as kelp and sessile invertebrates (Rattray et al. 2013). All of these contribute to determination of marine reserve boundaries and zoning (e.g. IUCN II sanctuary zone in Oceanic Shoals Marine Park in Heap et al. 2010, Director of National Parks 2017). In addition, acoustic technologies are often the only way to image the seafloor in extremely turbid environments where optical imagery is not appropriate (Matsumoto et al. 2015), and in some cases they can provide similar information on habitat patchiness as underwater video systems (e.g. seagrass beds in Lefebvre et al. 2009).Occasionally marine acoustic systems are used to detect change based on data acquired from repeated surveys (Przeslawski et al. 2011, Rattray et al. 2013).

The main restriction of many acoustic platforms is the cost associated with various aspects of equipment acquisition (e.g. purchase or hire price), data acquisition (e.g. vessel hire) and data processing (e.g. specialised staff and software). These are detailed more fully in Section 5.3.4.

Table 1 Summary of major types of benthic sampling platforms and their acquisition targets. With the exception of ROVs, hybrid platforms are not included (e.g. grab with video); these combine characteristics of the component platforms and are briefly described in Section 0. The qualifier 'accumulated' indicates transects that combine all data collected into a single datum (e.g. sled hauls)'. 'Continuous' refers to a grid. Unless otherwise specified, 'imagery' refers to optical imagery.

		Data Type	Data Target	Spatial coverage	Environment
Acoustics	MBS	Bathymetry, backscatter (water column & seabed)	Seafloor, water column	Continuous[1]	All
Acoustics	Sidescan	Backscatter (seabed), acoustic imagery	Seafloor	Continuous or transect	All
Acoustics	Single-beam	Bathymetry, backscatter (water column & seabed)	Seafloor, water column	Continuous or transect	All
Visual Methods	AUV	Imagery, bathymetry	Epifauna, substrate	Continuous or transect	All
Visual Methods	BRUV	Imagery	Demersal fish, substrate, epifauna	Point (qualitative)	All
Visual Methods	Towed Vid	Imagery	Epifauna, substrate, demersal fish	Transect	All
Visual Methods	Sediment Profile Imagery	Imagery	Infauna, substrate and sub-surface	Point (vertical transect)	Unconsolidated substrate
Visual Methods	Drop camera	Imagery	Epifauna, substrate	Point	All
Visual Methods	UVC	Observations	Demersal fish, Epifauna, substrate	Transect	Shallow depths
Visual Methods / Direct Sampling	Manned submersible	Imagery, biological/geological samples	Epifauna, demersal fish, sediment, rocks	Transect	All
Visual Methods / Direct Sampling	ROV	Imagery, biological/geological samples	Macrofauna, demersal fish, sediment, rocks	Transect	All
Visual Methods / Direct Sampling	Diving, snorkelling	Biological samples	Epifauna	Point (quadrat), transect	All
Direct Sampling	Grab/box core	Biological and sediment samples	Macrofauna, infauna, sediment, porewater	Point	Unconsolidated substrate
Direct Sampling	Sled/Trawl	Biological samples	Macrofauna, epifauna	Transect (accumulated)	All
Direct Sampling	Rock dredge	Geological samples[2]	Rock, sediment	Transect (accumulated)	Consolidated substrate
Direct Sampling	Piston core, vibrocore, rock drill	Geological samples[2]	Rock, sediment	Point (or vertical transect)	Various
Direct Sampling	Suction sampler	Biological samples	Epifauna	Point	All

[1] Even when only a single pass is made with MBS, this is still classed as 'continuous' data due to the wide swath (10s metres) compared to transect-based platforms (metres).
[2] Biological samples can be opportunistically collected

2.1.1 Innovative acoustic techniques

Acoustic methods are continuously evolving with advancing technology, including expanding data storage and processing capabilities. Several emerging technologies, such as multi-frequency multibeam sonar, autonomous surface vehicles (ASVs), and acoustic cameras are likely to be used more frequently as equipment is refined and costs are reduced. Multi-frequency multibeam sonar uses two or more discrete centre frequencies so that researchers can identify features using an optimal frequency relevant to the composition of the target. These systems have the ability to esonify targets that may not have been detectable by single-frequency systems (Brown et al. 2017).These improved data can then be applied to refine seabed classifications (Hughes Clarke 2015) or track fish (Williamson et al. 2016).

Autonomous survey vehicles (ASVs) are robotic platforms deployed at the surface of the water and programmed to follow a path along which various seafloor data can be collected, including multibeam and sidescan. ASVs have been successfully used to map the seafloor, including coral reefs (Ackleson et al. 2017). They are generally less technically complex than AUVs due to their surface location which means there is no need to account for pressure or geo-referencing issues due to underwater deployment (Iscar et al. 2015). However, their high cost and transportation challenges have so far precluded their widespread use.

Acoustic cameras are sonar systems that operate at approximately 720kHz – 3.5MHz, and at close range they can reveal the size and shape of sharks, finfish, and other fauna (Kohji et al. 2006, Becker et al. 2011, Parsons et al. 2017). They are particularly useful in turbid environments where traditional photography or video is unsuitable. Other options in such environments may include using a laser line scanner; this innovative method has been used to successfully characterise polychaete tubes at very fine spatial scales (mm) (Schönke et al. 2017).

2.2 Direct Sampling

Direct sampling methods return specimens from the seafloor and comprise a range of platforms, including sleds, dredge, trawls, grabs, and corers (Table 1), each of which is designed for sampling a particular environment or community. The sampling equipment is generally deployed from the vessel via hand (if lightweight) or winch (> 10 kg), although equipment can also be operated by divers in shallow waters (Somerfield and Clarke 1997). Samples are then collected upon equipment retrieval and processed using methods that vary according to equipment, deployment success, and target biological or environmental data. Despite its extractive and sometimes destructive nature, direct sampling is a necessary first step in baseline data acquisition in all but the most well-studied areas. There are also potential ongoing needs for direct sampling for certain taxa or habitats not well-suited to imagery-based monitoring (e.g. infaunal scallops in Przeslawski et al. 2018).

The main benefit to direct sampling is the acquisition of specimens that can be taxonomically identified for species-level data. This information can be used for species inventories and predictive models, both of which can be applied to marine management (Woolley et al. 2013, Przeslawski et al. 2015). Specimens also allow a range of genetic, biochemical and biological analyses to be undertaken (O'Hara et al. 2014). Some direct sampling methods (e.g. sled, trawl) are advantageous in environments with low densities of organisms (e.g. deep-sea Williams et al. 2010a, Rowden et al. 2016) because they collect information over a much larger spatial area than point-based grabs and corers. However, they may also integrate information from multiple habitats/communities which can obscure data patterns.

The main disadvantages to direct samplers are their destructiveness and increased uncertainty due to variable and sometimes unobservable deployment conditions. Many direct sampling methods disrupt the surrounding habitat and associated organisms, and fragile organisms may be damaged by anything other than a box core or operator-based collections. Data derived from some platforms, particularly sleds and trawls, should be considered qualitative if their performance during deployment is not consistently monitored (e.g. to detect skipping over the seafloor) leading to uncertainty in effective sampling area. In addition, the effectiveness of a particular sled, trawl, grab or corer strongly depends on the environment (substrate, slope and depth) meaning that the relationships within the data may be biased. A specially adapted sled must be used for deployment over rugged terrain (Clark and Stewart 2016), and grabs and box corers are confined to use over unconsolidated sediments. Recent additions of cameras to direct samplers permit observation of the sampling and retrieval process and can reduce ineffective deployments, especially important in deep water (e.g. Williams et al. 2015).

2.2.1 Epifaunal samplers (sled, trawl, dredge)

Benthic sleds (also called sledges) and bottom trawls both use nets to collect organisms while they are towed across the seafloor. While trawls typically use free nets with doors to spread the net, sleds use frames and runners to protect and anchor the net (Eleftheriou and Mcintyre 2005). Benthic sleds target sessile or sedentary macrofauna and megafauna, with some designs able to be deployed over rugged terrain. In contrast, bottom trawls are typically more successful in collecting demersal or mobile fauna and are deployed over smooth terrain or soft sediments. Beam trawls are a well-tested sampling device with a heavy fixed mouth opening that improved sampling of sedentary macrofauna and associated communities. Most sleds target epifauna but also collect infauna, although these are not usually sampled as comprehensively (Rice et al. 1979).

A dredge is sturdier than a sled or a trawl and has a heavy metal frame. Samples of broken rock are collected in dredges, and biota are able to be scraped off the hard substrate (Eleftheriou and Moore 2005). Specialised dredges may target coarse sediments and associated large infauna (e.g. anchor dredge in Kaiser et al. 2000) or both epifauna and infauna (Brenke 2005). There are also a range of sleds, nets, and pumps used to sample planktobenthic and hyperbenthic animals (immediately above the seafloor) (Dahms and Qian 2004, Przeslawski and McArthur 2009), but these are not further included in this study due to their low frequency of use in monitoring programs.

2.2.2 Infaunal samplers (grab, corer)

Grabs and box corers both use receptacles to collect sediment after they are dropped to the seafloor. While the scooping motion of grabs disrupts sediment to various degrees, box corers return intact samples of the sediment strata (Eleftheriou and Mcintyre 2005), thus facilitating geochemical analyses that require in situ sediment layers. Grabs and box corers target surface sediment and associated porewater and fauna. They are typically deployed over sandy or muddy substrates, although some grabs accommodate gravel or cobbles. There are numerous historical studies that compare different infaunal samplers (Gage 1975, Baker et al. 1977 and references therein, Tyler and Shackley 1978, Shirayama and Fukushima 1995).

Although grabs are simpler and usually easier to deploy than box corers, they do have more limitations, in addition to the sediment disruption mentioned above: Most grabs are not ideal for use on coarse-grained sediments as the grains can prevent closure (Jørgensen et al. 2011) which results in sample loss and underestimation of density or richness (Lozach et al. 2011). Certain types of grabs are also difficult to successfully deploy in consolidated muds as the grab jaws cannot penetrate the cohesive materials to obtain a sample. Furthermore, larger organisms that are able to burrow deeply within the sediment are prone to abundance underestimation (Kendall and Widdicombe 1999), and widely dispersed or rare fauna are susceptible to being overlooked (McIntyre 1956). Both grabs and corers can be used for quantitative analysis, but only if certain issues are resolved, including consideration of sample volume variation, loss of surficial sediments, and redistribution of enclosed sediment (Blomqvist 1991a).

2.2.3 Operator-based collection (diver, suction, ROV, submersible)

If water depth, environmental conditions, and logistics allow, specimens can be collected directly by walkers, swimmers or divers. Direct sampling is particularly useful in areas of high biodiversity and shallow or intertidal waters. For shore surveys, the Riley push-net can be used to collect fast, active biota. For both shore and shallow water surveys, square frames (quadrats) placed upon the substrate can be used as boundaries in which organisms can be counted and surveying can also be completed by the use of a transect (Eleftheriou and Moore 2005). Divers can undertake written, audio, photographic or video recordings of benthic biota, as well as collecting specimens (Munro 2005).

Suction samplers are tubes that use suction to either penetrate the substrate or extract sediment into an overlying tube (Hopkins 1964). These systems can either be diver operated or remotely operated, but most suction samplers are only suitable for use in shallow and relatively calm waters (Eleftheriou and Moore 2005). They are valuable for sampling in coarse sediments and for obtaining deep burrowing biota, but their use may artificially increase abundance data where surrounding biota are sucked into the sampling area (Munro 2005). Furthermore, sedimentary layering is not preserved.

Remotely operated vehicles (ROVs) and submersibles are used to collect specimens in environments unsuitable for other direct sampling methods (e.g. deep-sea, canyons). Unlike most grabs and corers, these methods allow an operator to control the sampling equipment to choose an appropriate area for sampling, thereby providing a level of precision not achieved by other sampling platforms (Kelley et al. 2016). However, extra care must be given to the design and analysis of these ‘adaptive’ surveys (Thompson 2012). These platforms can be fitted with various other equipment and sensors to collect animal, rock, water, and sediment samples, as well as imagery and continuous data about the environmental conditions (e.g. temperature, salinity). They can be used for species inventories and taxonomic identification, and with appropriate storage containers, they can also collect and maintain live specimens from deep-sea habitats (Takemura et al. 2010). Historically, submersibles were considered superior to ROVs regarding their precision of sampling in challenging environments, but advances in ROV technology have made the choice between ROV and submersible more a matter of equipment availability than any true advantage (Kelley et al. 2016). The increased sampling precision of ROVs is traded off against a smaller sampling area and higher deployment costs.

2.2.4 Innovative direct sampling techniques

Innovations regarding direct sampling technique involve i) adaptations to existing platforms, ii) development of new platforms, and iii) implementation of citizen science programs. Researchers are continually adapting traditional sampling platforms mentioned above to improve sampling efficiency, including the capability to collect undisturbed sediment cores across a range of environments (Xu et al. 2011, Blomqvist et al. 2015).

Fragments of species-specific environmental DNA (eDNA) can be analysed from seawater and sediment to develop species inventories, identify rare or invasive species (Rees et al. 2014) and even detect different populations (Sigsgaard et al. 2016). Metabarcoding of eDNA is often quicker and more sensitive than traditional sampling (Boussarie et al. 2018), although its application to environments in which the fauna are poorly known is limited (e.g. deep-sea platyhelminthes and nematodes in Sinniger et al. 2016). Although there have been numerous efforts to apply eDNA to abundance and biomass estimation (Pilliod et al. 2014, Rees Helen et al. 2014, Klymus et al. 2015), these parameters have yet to be proven as a reliable measurement from eDNA. Current technology requires different eDNA methods among different environments, but there is a move to develop standardised methods so that data can be compared among different surveys (Djurhuus et al. 2017).

2.3 Visual Methods

Visual methods are observations of the target environment as it generally appears to the human eye. These include both imagery-based platforms (e.g. towed video, AUVs, BRUVs) and underwater visual censuses (UVC) from snorkelling, SCUBA diving, or submersibles (Table 1). Marine imagery systems present a range of choices to be made according to the target habitat, taxa, and metric to be analysed. Many of these are photographic specifications of the camera and lighting equipment (e.g. lens type, image resolution, aperture, flash type, (reviewed by Smith and Rumohr 2013, Bowden and Jones 2016, Durden et al. 2016b). Other options are more directly linked to the type of data to be extracted from imagery, including choices between monocular or stereo cameras (and associated ability to quantify size), inclusion of lasers for scaling, oblique or downward-facing cameras, and colour or monochrome imagery (see Boutros et al. 2015, Durden et al. 2016b and references therein). Imagery systems can include video, still cameras, or a combination. Ideally, the system is based on the metric to be analysed, although practical considerations such as available equipment mean that although still cameras provide higher resolution imagery, data is frequently extracted from screenshots of video footage for habitat and sessile invertebrate characterisation (Sheehan et al. 2016).

In addition to all the camera specifications, the housing body itself can affect data acquisition. Sheehan et al. (2016) compared three towed bodies, each with a forward-facing video camera, and found significant differences between them in species richness, density, cover, and assemblages of sessile and sedentary benthic taxa. The camera housing body can be towed, autonomous, or diver-operated.

The main advantage to visual methods are their non-destructive nature, thus making such methods preferred in benthic research that requires repeat sampling of the same area (e.g. Stuart-Smith et al. 2017) or in areas where destructive methods are prohibited. Visual methods also allow the quantification of organisms that may otherwise disintegrate upon collection or processing (e.g. xenophyohores in Rice et al. 1979, acorn worms in Anderson et al. 2011). Most visual methods also allow imagery to be readily shared among data users, thus providing a permanent and accessible record of the sampling. In addition to providing quantitative data, imagery can also be used to inform sampling locations, including using video-mounted sleds, trawls, and grabs (Thouzeau et al. 1991).

The main disadvantages of visual methods is that they are generally limited to larger organisms and may involve uncertainty regarding species-level identification, particularly in areas with poorly known flora and fauna that require specimens to confirm identification (Buhl-Mortensen et al. 2015, Lee et al. 2015). In such circumstances, individual taxa are often grouped into classes based on colour and morphological traits (Schönke et al. 2017), although this is less common in data acquired from BRUVs or UVC, likely due to the propensity for these platforms to be deployed in shallower waters in which species inventories are well-documented. Despite this inherent limitation, using broader morphological classes to calculate richness and community metrics has been shown to correlate well with actual species richness and diversity in sessile invertebrates elsewhere (e.g. sponges in Bell and Barnes 2001). Another challenge when using imagery systems is the variable data quality due to environmental conditions (i.e. turbidity, ice cover) and the associated difficulty in classifying biota at even high taxonomic levels. Mueller et al. (2006) found that acoustic cameras may be more suitable than optical imagery for quantifying fish size and density in ice-covered shallow freshwater environments, although there was a limited ability to differentiate species. In addition, observer bias during image annotation and UVC may also affect data quality (Durden et al. 2016a). As mentioned above, some visual methods (e.g. towed video, AUVs) generate thousands of images or hours of video. Although the sheer amount of data able to be extracted is an advantage, this can also be a disadvantage due to the long amount of time needed to manually annotate images. Automatic classification and image recognition in the marine environment may help alleviate these challenges, but these techniques are still in their infancy. There has, however, been recent progress with categorisation of broad habitat types (e.g. presence or absence of kelp in (Bewley et al. 2012) and development of marine image annotation software (Gomes-Pereira et al. 2016). For further challenges on using cameras in marine monitoring, see Panel 1 in Bicknell et al. (2016).

2.3.1 UVC

Underwater visual census (UVC) is when a swimmer, snorkeller, or diver records observations within a standardised area or time. This technique is particularly useful in shallow waters < 30 m, noting that specific diving safety regulations may limit UVCs to shallower depths. Due to the comparative ease of revisiting a particular site, especially with the high accuracy of modern day GPS, UVCs are a popular tool for monitoring coral and rocky reefs (Stuart-Smith et al. 2017) and are the main tool in the global citizen science monitoring program [Reef Life Survey](). Importantly, UVCs often collect data on multiple groups of taxa, including fishes, mobile invertebrates and sessile flora and fauna. Although UVC data is acquired primarily through *in situ* observations, many UVCs including Reef Life Survey now use diver-operated imagery to identify unknown or uncertain species or to provide photo-quadrats at regular intervals for sessile fauna. As with other visual platforms, these images can be double-checked or used for other purposes (such as annotation using different classification systems; e.g. (Cresswell et al. 2017). Divers or snorkelers are sometimes towed on manta boards behind vessels to increase the area covered and the likelihood of encountering faster moving pelagic species.

In addition to the limitation of diving depths, the main disadvantages of UVCs relate to i) under-representation of diver-shy fish species, although bias in representation of species is still typically lower than other methods (Lowry et al. 2012) and ii) observer error in estimating distances and sizes underwater. Comprehensive training and regular practise can help reduce the latter (Edgar and Stuart-Smith 2009). For those UVCs that do not use diver-operated imagery or those taxa not targeted by such imagery, the lack of archival records may be a disadvantage in cases where errors need to be checked.

2.3.2 Dropped platforms

Dropped platforms include any unbaited imagery system that is lowered directly from a stationary vessel to the seabed below and remain attached throughout deployment[3]. Unlike towed, autonomous, or remote platforms, they acquire imagery from point locations rather than transects. Dropped platforms include simple cameras (e.g. Go-Pros) designed to take still images or video of the seafloor directly beneath the vessel. Due to their speed and ease of use, these are often used to provide a preliminary assessment of substrate and habitat. This information can then be applied to habitat classifications, decisions to deploy other equipment, or ground-truthing predictive models. As such, dropped platforms are often used in conjunction with other marine sampling platforms (e.g. Archambault and Bourget 1996, Barberá et al. 2012).

Dropped platforms also include more complex and specialised systems such as sediment profile imaging (SPI) systems in which a camera is inserted into unconsolidated substrate to take a cross-section image of the sediment and sediment-water interface (reviewed by Germano et al. 2011). In most SPI systems, a wiper cleans the lens after image acquisition so that images at multiple points can be acquired in a single drop. Unlike other imagery systems, SPI can target infauna and also allows the quantification of physical and chemical characteristics, such as grain size and redox area, along a vertical sediment profile. This information can be used for environmental monitoring and to detect impacts (e.g. trawling impacts in Nilsson and Rosenberg 2003). SPIs are appropriate to use in benthic monitoring of environments characterised by mud and muddy sands (Keegan et al. 2001).

2.3.3 Towed platforms

Also called video or camera sleds, towed imagery platforms consist of a frame supporting at least one imagery system along with necessary accessories (e.g. lights) that is towed behind or alongside a moving vessel. Compared to other visual methods, towed camera platforms have

[3] Some studies refer to cameras deployed while the vessel is moving as drop cameras (Williams and Leach 1999). For the purposes of this report, we classify these as towed platforms.

high proportions of unsuitable seabed images (Hayes et al. 2015), likely due to difficulty maintaining a consistent altitude during deployment and the associated issues with focus. However, they are popular due to their relative ease of deployment, flexible vessel requirements, and comparatively low cost and technical complexity (compared to AUVs and ROVs). In addition, towed platforms often include both video and still imagery systems, each of which produces a discrete yet complementary dataset. Still images are suitable for quantifying smaller, commonly occurring biota, while video is more suitable for larger, widely dispersed biota (van Rein et al. 2012, Clark and Bowden 2015). Most imaging systems towed along the seabed are confined to epibenthos and demersal animals, but the Burrow-Cutter-Diaz sled acquires continuous profiles of infaunal communities and habitats by using an SPI system to plow through the top 10-20 cm of sediment, (Cutter and Diaz 1998).

2.3.4 Baited platforms

Baited platforms, work by both attracting fish into the field of view of a camera(s) and sampling the ambient fish associated with the habitat within the field of view, thereby recording the diversity, abundance and behaviour of species. There are two main types of baited platforms: landers use downward-looking camera (BUV), and horizontally-facing cameras baited remote underwater video (BRUVs) use either one (mono) or two (stereo). Recent research suggests that stereo BRUVs observe more fish species than downward-looking landers (Langlois et al. 2006, Cundy et al. 2017). Stereo-baited remote underwater video systems (stereo-BRUVs) consist of two video cameras inside a waterproof housing, attached to a frame with some form of baited device placed in front of the cameras. Benthic stereo-BRUVs are lowered to seafloor and are left recoding for a set duration of time. The video footage can then be used to assess the recorded fish assemblage and associated habitat, with photogrammetry used to determine the size of fishes.

Stereo-BRUVs are becoming widely adopted as a non-extractive technique for sampling the relative abundance and length of fish assemblages (Watson et al. 2009, Logan et al. 2017, Whitmarsh et al. 2017, Hill et al. 2018). They have been used to successfully monitor spatial and temporal changes in benthic fish communities and their habitat structure (Cappo et al. 2004, Langlois et al. 2006, Langlois et al. 2010, Harvey et al. 2013). The use of bait increases the relative abundance and diversity of fishes observed, particularly species of interest to fisheries, without precluding the sampling of prey or herbivorous fish species (Lowry et al. 2012, Hardinge et al. 2013, Coghlan et al. 2017). Multiple stereo-video systems can be deployed in the field consecutively, making efficient use of researcher and boat time (Cappo et al. 2007, Langlois et al. 2010, Whitmarsh et al. 2017). This allows for the possibility of large spatial coverage and high replication within short field campaigns.

Some of the limitations of stereo-BRUVs include reliance on good visibility (usually greater than 3 metres) and challenges determining the true area sampled due to bait plume variability biases (Cappo et al. 2007, Whitmarsh et al. 2017). Similarly, the responses of different fish species to the bait and the distances they will travel to get to the bait is largely unknown (Harvey et al. 2007). This also applies for crytobenthic and site-attached species that are often under represented using BRUVs. For these reasons, counts of fish from BRUVs are currently limited to measures of relative abundance rather than density (Cappo et al. 2007). Another limitation is in determining whether fish that are seen in the field of view at one time on a recording are different individuals to those observed at a different time on the recording. To overcome this, counts of the maximum number (MaxN) of individuals of any one species seen over the entire recording have been used. However, even this measure is contentious, with some authors suggesting that it is not well-related to actual abundance and can make populations appear hyper-stable (Schobernd et al. 2013). Compared to other visual methods, BRUVs can alter behaviour, attract certain types of organisms including large predatory fish, and repel others (Watson et al. 2005, Seiler 2013). Nevertheless, while BRUVs are unsuitable for estimating density, they are a powerful and cost-effective method for detecting spatial and temporal changes in the relative abundance and lengths distributions of fish assemblages, as well as providing data on habitat in the immediate vicinity.

2.3.5 Autonomous platforms

Autonomous platforms are able to sample or image the seabed while moving under their own power without being tethered to a vessel. Gliders and floats are relatively small systems that are able to move horizontally (e.g. ocean glider) or vertically (e.g. Argo float) in the water to collect various oceanographic data, including samples close to the seafloor (www.imos.org.au). They are rarely used for biological or benthic data. In contrast, Autonomous Underwater Vehicles (AUVs) are routinely deployed to collect seafloor images and data on the seafloor benthos and physical structure. AUVs are untethered robotic submarines that operate independently to complete a pre-programmed survey, with endurance and depth range varying among AUVs. Similar to ROVs, depending on their depth rating and size, AUVs can be equipped with a range of sensors (CTDs, ADCPs, chemical sensors, photo cameras, sonars, magnetometers, gravimeters etc. (e.g. Williams et al. 2010b, Connelly et al. 2012, Sumner et al. 2013).

AUVs are typically categorised as either "cruising" or "hovering" vehicles. Cruising AUVs are generally torpedo-shaped and driven by a single propeller. They move at speeds up to 2 ms^{-1}, and are optimised to cover large distances along pre-designed tracks (Wynn et al. 2014). They form the main type of AUVs used in the commercial world. By contrast, hovering AUVs have several propellers/thrusters, which allow them to move in any direction and provide them with a high manoeuvrability, much like a pre-programmed ROV. They are designed for precision operations, slow motion surveys (e.g. seabed photography) and work in distinctly 3-dimensional terrains, such as around coral reefs (Williams et al. 2016, Langlois et al. 2017). When equipped with navigational sensors such as GPS, Ultra Short Baseline Acoustic Positioning System (USBL), acoustic Doppler profiler, and forward-looking obstacle avoidance sonar, hover class AUVs enable precise tracking along the pre-programmed transects. These characteristics make them particularly suited to collecting highly detailed sonar and optical images to be geo-referenced at high precision, facilitating photomosaics that focus on large features or specific details of the seafloor.

The application of AUVs to monitor marine ecosystems has experienced a rapid increase over the past two decades. Researchers have used a hover class AUVs to acquire benthic images used in monitoring the impacts of invasive species (Ling et al. 2016b), ecosystem-based fisheries management (Department of Biodiversity Conservation and Attractions 2017), population trends in demersal fishes (Clarke et al. 2009, Seiler et al. 2012), benthic habitat mapping (Golden et al. 2017), examining diversity in reef communities (Bridge et al. 2011, Monk et al. 2016, James et al. 2017), changes in structural complexity of coral reefs (Coghlan et al. 2017), and mapping the spatial and depth extent of kelp forests (Alory et al. 2007).

2.3.6 Remote platforms

Remotely operated platforms such as ROVs can be used to acquire marine imagery, often in conjunction with direct sampling (Section 2.2.3) and other sensors (e.g. temperature, pressure). ROVs are particularly useful in environments in which other imagery platforms are unsuited, such as high relief areas including vertical canyon walls (Thresher et al. 2014). Their ability to be controlled by an operator in real-time is particularly suited for imagery of targeted taxa and habitats, as well as repeat images at a particular location. However, this non-randomisation means that images from ROVs are usually unsuitable for quantitative analyses related to monitoring unless strict protocols are followed, which require specialist analyses (Thompson 2012). ROV imagery is often used to complement other visual (Rathburn et al. 2009) or direct sampling (Vetter and Dayton 1998) methods. The use of ROVs to acquire marine imagery may be associated with challenges including increased cost compared to other drop, towed, and baited platforms; difficulty controlling the ROV in high currents, and observer bias (Azis et al. 2012). As with AUVs, the deployment of ROVs is challenging in high current environments, with increasingly powerful thrusters needed to maintain position.

2.3.7 Innovative visual methods

Imagery systems are rapidly advancing, both in terms of hardware for image acquisition and software for data analysis. In a review of camera technology for marine monitoring, (Bicknell et al. 2016) encourages the use of remote camera systems with multiple methods and sensors to improve knowledge of marine biodiversity and assess impacts.

Due to the increasingly large number of images routinely collected on marine surveys, many of the advancements in marine imagery are related to reducing the time spent on data annotation (i.e. extracting quantitative data from images). Automated classification of imagery can be performed by machine-learning algorithms able to broadly group images (e.g. terrain in Friedman 2013), but wide-spread automation at high taxonomic resolution is still unobtainable. Crowd-sourcing annotation can be done by citizen science programs (reviewed in Durden et al. 2016b) which often incorporate online annotation platforms (e.g. Squidle+). See Section 2.2.4 for further details.

The recent large-scale rollout of several robust citizen science programs has been one of the most exciting developments for marine monitoring. These programs can saved time and money by employing highly trained divers (e.g. Reef Life Survey in Stuart-Smith et al. 2017) or observations from fishermen (e.g. RedMap in Pecl et al. 2014) to record data about species presence and distribution over time, particularly related to threatened species (Edgar et al. 2017). Importantly, citizen science programs must consider uncertainty and error associated with observer bias and misidentification (Bird et al. 2014, Chase and Levine 2016) to ensure quality and independence.

2.4 Hybrid Platforms

Hybrid platforms use two or more of the platforms mentioned above to maximise the number of species, habitats or communities sampled by targeting different ones in a single deployment. For example, the Burrow-Cutter-Diaz sled combines traditional sled sampling with sediment profile imagery to examine both epifauna and *in situ* infauna (Cutter and Diaz 1998). Hybrid platforms can also be used to independently validate commercial data. For example, acoustic systems have been attached to trawl nets to not only corroborate catch but also to investigate behaviour and orientation of fish during trawling operations (Ryan et al. 2009). With increasing technological capability and accessibility, the use of hybrid platforms will likely increase as it incorporates advantages of multiple sampling platforms into a single deployment. For example, the development of the *Nereus* hybrid ROV used innovative buoyancy devices, lithium batteries, and fibre-optics tether to develop a light-weight unit that could operate as either an ROV or an AUV (Kelley et al. 2016).

3. USE AND PERCEPTION OF SAMPLING PLATFORMS

Although user perceptions are subjective and may not reflect a platform's true capability or cost, they are nonetheless important in a comprehensive evaluation to identify the reasons behind a given platform's use, or lack thereof (i.e. an appropriate platform may be underused simply because it is perceived as being too costly or difficult to operate).

3.1 Methods

A questionnaire to gauge use and perceptions of common benthic marine sampling platforms in Australia was released on 15 Dec 2016 to NESP researchers via Survey Monkey: www.surveymonkey.com/r/C2DQCRC. The questionnaire was advertised between 25 Jan – 23 Feb 2017 on the e-news of the Australian Marine Science Association and emailed to individual researchers as appropriate. The questionnaire was approved by CSIRO's Social Science Human Research Ethics Committee in accordance with the National Statement on Ethical Conduct in Human Research (2007). There were 17 questions about respondents' marine survey experience, equipment use, and perceptions. The platforms chosen were based on those most commonly used based on the literature search and through ongoing Australian monitoring programs (e.g. IMOS) to inform the development of a suite of field manuals (Przeslawski and Foster 2018). As such, the platforms do not necessarily match all the platforms included in Sections 2 and 4 (e.g. ROV and UVC were not included in the questionnaire). The platforms included in the questionnaire were MBS, AUV, BRUV, towed imagery, grabs and box corers, and sleds and trawls.

3.2 Results - Demographics & Platform Use

Of the 49 people who completed the questionnaire, the majority were scientists (82%). Most respondents worked at government research institutions (49%) or universities (29%), with all but two (4%) working in Australia. Most respondents (69%) were experienced in marine fieldwork, undertaking over 12 expeditions. Only 10% had undertaken less than four marine expeditions. Marine fieldwork spanned all Australian regions and environments, with the majority of respondents working in the Coral Sea/GBR (42%) or the North-West (44%), and in coastal (69%) or shallow shelf (<50 m) (80%) waters.

Three platforms were frequently used by a large proportion of respondents: MBS (42.5%), grabs/boxcores (41%), and towed video (40%). The other three platforms were never or rarely used by most respondents: AUV (42.5% never used, 21% rarely used), BRUV (40% never used, 19% rarely used), and sleds (30% never used, 26% rarely used).

3.3 Results - Perceptions of Platforms

The majority of respondents stated that an important aspect of a sampling platform to ensure national adoption was the ability to re-use or pool data (76%), followed by flexibility (63%), development of a national data resource (63%), agreement among experts (63%), and succinctness and clarity (59%). Moderately important aspects were low costs (47%) and defensibility (39%). A lower priority was the incorporation of current approaches (16%).

Perceptions of cost, including post-processing of data, are listed in Table 2. The highest were associated with the AUV (84% of respondents marked prohibitive or high cost) and MBS (65% prohibitive/high). The lowest perceptions of cost were with grabs/boxcores (88% low or moderate cost), with the remaining platforms having mostly moderate or high cost (BRUV: 70%; Sled/trawl: 84%; Towed vid: moderate 72%).

Table 2 Percentage of respondents that marked each sampling platform based on their perception of total cost, including post-processing. Grey text indicates numbers of respondents.

	Total No. Respondents	Low	Moderate	High	Prohibitive
Multibeam sonar	49	12.2% 6	22.5% 11	53.1% 26	12.2% 6
AUV	49	0.0% 0	16.3% 8	63.3% 31	20.4% 10
BRUV	48	25.0% 12	39.6% 19	31.3% 15	4.2% 2
Towed video	48	27.1% 13	58.3% 28	14.6% 7	0.0% 0
Grab or boxcore	49	49.0% 24	38.8% 19	12.2% 6	0.0% 0
Sled or trawl	46	10.9% 5	65.2% 30	19.6% 9	4.4% 2

Perceptions of the effort regarding deployment challenges or technical complexities are shown in Table 3. These show high perceived effort associated with AUVs (61% of respondents) and MBS (51%), and a further 6% and 8% of respondents, respectively, indicated the AUV and MBS required prohibitive effort. Grabs/boxcores (39%) and BRUVs (31%) were perceived as being the easiest regarding deployment.

Table 3 Percentage of respondents that marked each sampling platform based on their perception of deployment challenges or technical complexities. Grey text indicates numbers of respondents.

	Total No. respondents	Easy	Moderate effort	High effort	Prohibitive effort
Multibeam sonar	**49**	**6.1%** **3**	**34.7%** **17**	**51.0%** **25**	**8.2%** **4**
AUV	**49**	**0.0%** **0**	**32.7%** **16**	**61.2%** **30**	**6.1%** **3**
BRUV	**49**	**30.6%** **15**	**42.9%** **21**	**26.5%** **13**	**0.0%** **0**
Towed video	**49**	**18.8%** **9**	**70.8%** **34**	**10.4%** **5**	**0.0%** **0**
Grab or boxcore	**49**	**38.8%** **19**	**55.1%** **27**	**6.1%** **3**	**0.0%** **0**
Sled or trawl	**49**	**18.4%** **9**	**55.1%** **27**	**26.5%** **13**	**0.0%** **0**

Perceptions of the <u>effort regarding post-processing challenges or technical complexities</u> were more equitable among sampling platforms (Table 4). Highest post-processing effort was associated with AUVs (65% of respondents) and BRUVs (56%). No platforms were identified by a large proportion of respondents as being easy to post-process.

Table 4 Percentage of respondents that marked each sampling platform based on their perception of post-processing challenges or technical complexities. Grey text indicates numbers of respondents.

	Total No. Respondents	Easy	Moderate effort	High effort	Prohibitive effort–
Multibeam sonar	**45**	**4.4%** 2	**46.7%** 21	**46.7%** 21	**2.7%** 1
AUV	**46**	**4.4%** 2	**30.4%** 14	**58.7%** 27	**6.5%** 3
BRUV	**46**	**4.4%** 2	**40.0%** 18	**55.6%** 25	**0.0%** 0
Towed imagery	**46**	**13.0%** 6	**50.0%** 23	**37.0%** 17	**0.0%** 0
Grab or boxcore	**46**	**15.2%** 7	**58.7%** 27	**26.1%** 12	**0.0%** 0
Sled or trawl	**45**	**13.0%** 6	**50.0%** 23	**37.0%** 17	**0.0%** 0

For imagery and acoustics systems (MBS, AUV, Towed Vid, BRUVs), one of the main advantages many respondents listed was their non-destructive sampling regime and potential for permanent records and the ability to share the data. One of the main drawbacks was constraints on high-level taxonomic identifications due to the lack of biological samples. In contrast the destructive nature of sampling with sleds/trawls and to a lesser extent grabs/boxcores was listed as a drawback, while the ability to identify organisms to a high taxonomic resolution and obtain biological samples (including genetic) was one of the main advantages.

For MBS, the advantage most frequently listed was its capability to efficiently generate large continuous high-resolution maps of the seafloor. Other advantages include profiling the water column, integration with industry surveys, foundation data for further sampling plans, and spatial precision of data. The main drawbacks were listed as high capital or hire cost, highly skilled technical staff needs, extensive set-up, time-consuming post-processing, expensive software, limited biological use (i.e. need for ground-truthing), and larger vessel requirements.

For AUVs, respondents listed the following advantages: inclusion of multiple environmental sensors, autonomy allows concurrent sampling with other platforms, ability to revisit exact areas (temporal monitoring), acquisition of high-quality images, ability to survey complex habitats (e.g. reefs), capacity to create mosaics and fly-throughs, and existence of outreach and online data systems. Many respondents said that the main drawback of the AUV was its high cost and technical support needs. In addition, respondents identified the following other drawbacks: bottlenecks regarding annotation, lack of consistency in mission design, comparatively frequent failed missions, vessel requirements for larger AUVs, unsuitable for high energy environments, and limited number of AUVs available.

For BRUVs, the main advantages listed were its simplicity, cost-effectiveness, ability to observe behaviour, ease of replication, ability to quantify fish size and relative abundance and diversity, few vessel requirements, ability to sample mobile predators, and acquisition of ideal imagery for public interest. The main drawbacks listed were limitations sampling non-predatory fish, water clarity requirements, expense of equipment, and time-consuming post-processing.

For towed imagery platforms, the main advantages listed were its simplicity and low cost compared to AUVs, equipment accessibility and flexibility (e.g. Go-Pro to deep tow system), capability for real-time classifications, flexibility to add multiple still and video cameras, ability to georeference, and no need for specialists. The main drawbacks were listed as difficulty piloting cameras over rugged terrain, inconsistent elevation, variable image quality, time-consuming annotation, water clarity requirements, potential for snagging, inability to maneuver and revisit same transect, limited spatial precision, and inconsistent annotation/analysis methods.

For grabs and boxcores, the main advantages listed were their simplicity and ease of deployment, low cost, ability to collect various co-located samples to support multiple disciplines (e.g. infauna, sediment, porewater), ability to directly link physical and biological samples, and reliability. The drawbacks were limited spatial precision, time consuming post-processing, challenges standardising grabs among different substrates, inability to sample hard ground, need for onboard sample processing and storage, and lack of context for surrounding habitat.

For sleds and trawls, the main advantages listed were their simplicity and ease of deployment, ability to obtain bulk samples, and low deployment costs. Drawbacks were limited spatial precision, inability to maneuver and revisit the same transect, potential for snagging or skipping, inability to pinpoint sample location over entire transect, destruction especially on hard ground, and gear avoidance.

4. COMPARISON OF PLATFORMS

Marine sampling methods have obviously changed over time (Eleftheriou and Mcintyre 2005, Danovaro 2010, Clark et al. 2016), with a marked shift in the mid-200s from direct sampling such as sleds and trawls to more technologically advanced and non-extractive methods based on imagery and remote sensing (Hayes et al. 2015). These methods have surged in popularity, likely due to their reduction in cost and technological improvements (Bicknell et al. 2016). The effectiveness and suitability of marine sampling platforms depends on survey objectives, available resources, and environment (terrain, substrate, depth), all of which underpin data quality. In this section we conduct a literature review to compare common marine benthic sampling platforms.

4.1 Methods

To identify potentially useful data and results incorporating multiple sampling platforms, we searched the Web of Science database using keyword combinations of pairs of gear types (sled*, trawl, grab, *core, video, image*, UVC, BRUV, AUV) and filtering by 'Marine Freshwater Biology'. Previous Hub outputs were also targeted to ensure consistency and legacy value (Flannery and Przeslawski 2015, Hayes et al. 2015).Selected studies (henceforth called 'preliminary selection') were confined to marine benthic or demersal biological sampling platforms. We have excluded acoustic methods from this section due to the marked difference in data type and spatial extent compared to direct sampling and visual observation methods. Similarly, specific instances of platforms without biological data (e.g. SPI in Keegan et al. 2001, video in Brokovich et al. 2008) were not included. Studies comparing a single platform with minor modifications were not included (e.g. McHugh et al. 2015 who compared effectiveness of otter trawls with different board angles), nor were studies employing a single sampling platform with multiple configuration or post-processing methods (e.g. different sieve sizes in Thompson et al. 2003, camera configuration in Boutros et al. 2015, abundance estimates from BRUVs Campbell et al. 2015).

A qualitative comparative assessment was then undertaken based on the number and quality of studies short-listed from the literature review (henceforth called 'refined selection'). To be considered in the assessment, studies had to present separate results from multiple sampling gear (i.e. not pool data among gear types). In addition, studies had to either i) directly compare methods (e.g. sampling method is a factor in statistical analysis) or ii) test for similar ecological relationships among two or more gear types (i.e. environmental relationships to biological assemblages).

4.2 Results

4.2.1 Preliminary selection

A total of 153 studies was included in our preliminary selection. Only 20% (30/153) of studies used information from seafloor acoustics such as multibeam or sidescan sonar prior to sampling. These include studies that opportunistically used available acoustic information (Ringvold et al. 2015), programs that included a designated acoustics survey prior to sampling (Brown et al. 2002), and programs that collected acoustic information on the same survey as sampling (Schoenberg and Fromont 2011). Most studies (64.7%) incorporated only two benthic sampling platforms, with 10.4% of studies using four or more platforms (Figure 1).

The proportion of studies pooling data increased with the number of platforms (Figure 1). A total of 19% (28 of 153) of studies pooled data from all gear types, thereby negating any sort of comparison between methods. Most of these (67.9%,19 of 28) focussed on a single group or species, indicating that the multiple gear types were used solely to maximise the number of individuals for species inventories (Prezant et al. 2002), taxonomic (Mendez and Yanez-Rivera 2015), behavioural (Johansen and Brattegard 1998), or biological (Mikkelsen and Bieler 2001) purposes, rather than analysis of spatiotemporal or biodiversity patterns.

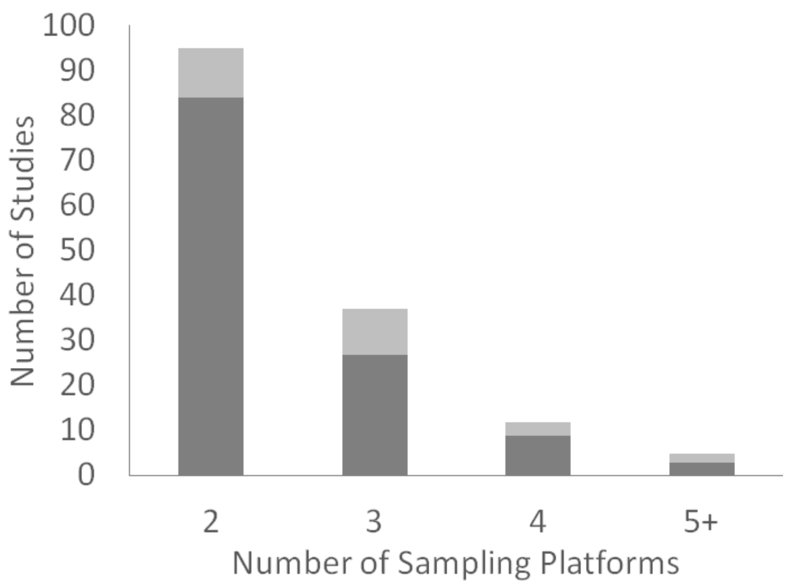

Figure 1 Number of benthic sampling platforms included in studies identified from preliminary selection. Studies that pooled data among platforms are indicated in light grey.

4.2.2 Refined Selection

Direct Comparisons

The most straightforward way to summarise studies directly comparing multiple sampling methods is to identify whether there are statistically significant differences between sampling methods. In this case, sampling method is a factor in a statistical test or model with a common dependent variable. These represent some of the clearest comparisons we have between platforms but was possible in only 34% (51 of 152) of preliminarily selected studies, with a further 15 studies qualitatively comparing methods (i.e. no statistical model) often among different dependent variables. Publication bias should be considered in such studies, as studies that found no difference between sampling methods may be less likely to be published (Jennions et al. 2013).

For most combinations of sampling platforms, there were typically less than three studies directly comparing them, with many pairs of sampling platforms not compared at all (Table 5). Despite their surge in popularity, particularly for marine monitoring, AUVs were only represented by a single comparative study (trawling, towed video and AUV still images in Morris et al. 2014) while comparisons involving more traditional methods (sled/trawl, grab/corer, direct observations, towed imagery) were more common (Table 5). We were unable to assess sampling efficiency (e.g. Tyler and Shackley 1978) since it varies among environments, and the number of studies comparing sampling methods is too low to incorporate environmental variation.

The following key points can be gleaned from Table 5:

- The platforms with the most similarity between them may be sled/trawl and UVC (Spencer et al. 2005), grab/corer and UVC (Aguado-Giménez et al. 2007), and AUV and towed imagery (Morris et al. 2014), although there was only a single study within each of these comparisons so further research is required to test this.

- The platforms with the least similarity between them may be operator-based direct sampling and sled/trawl (Gage 1975, Beisiegel et al. 2017), operator-based imagery acquisition and UVC (Harvey et al. 2002, Pelletier et al. 2011), and UVC and BRUVs (Willis et al. 2000, Colton and Swearer 2010, Lowry et al. 2012, Boussarie et al. 2018).

- Differences between sampling platforms varied among studies (
Table 5), likely due to targeted habitat, taxa, variable, or equipment design. For example four studies showed a significant difference between sleds/trawls and towed imagery (Spencer et al. 2005, Morris et al. 2014, Williams et al. 2015, Beisiegel et al. 2017), while others showed no difference (Vorberg and van Bernem 1998, Smith and Papadopoulou 2003) or indeterminate comparisons (McIntyre et al. 2015).

- Within a given sampling platform, significant differences were often found between equipment designs (
Table 5). For example, assemblages and richness derived from samples collected from different grabs and box corers were significantly different in over half the selected studies (Baker et al. 1977, Bett et al. 1994, Shirayama and Fukushima 1995, Somerfield et al. 1995).

Table 5 Studies identified from refined selection that statistically compare marine sampling methods. For studies including three or more methods, each pair of methods is included in the table below.

		Direct Sampling				Visual Methods						Other
		Grab & corer	Sled & Trawl	Lines & Traps	Operator-based	UVC[4]	Drop	Towed	BRUV	Autono-mous	Operator-based[5]	
Direct Sampling	Grab & corer	[1, 5, 6, 7], [2], [3, 4]	[8], [9, 10] [11, 12, 13]		[2, 3], [13, 14]	[15]		[13]				
	Sled & Trawl		[16, 17], [18, 19] [20, 21]		[12, 13]	[22]		[13, 22, 23, 26], [24, 25] [27]		[23]		
	Lines & Traps					[28], [29, 30, 31, 32, 33]			[29] [34]		[35]	
	Operator-based											
Visual	UVC					[36]		[22]	[37, 38] [29, 39]		[40, 41]	[39][6]
	Drop											
	Towed							[42, 43]	[44]	[23]		[45][7]
	BRUV								[46]		[47], [48, 49]	[39]
	Autonomous											
	Operator-based										[50], [51]	
Other												

Green	No difference
Grey	Undetermined (e.g. depends on metric, low power)
Red	Significant difference
White	No studies found

[4] Includes direct observations (i.e. not digitally recorded) from snorkeler, diver, or submersible passenger
[5] Includes diver, ROV and submersible operations of video or still imagery
[6] e-DNA vs UVC vs BRUVs to measure shark richness
[7] Sidescan sonar vs towed video to measure seagrass

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Congruence of ecological relationships

For many studies, direct comparisons between methods were not possible due to the increased likelihood of different metrics being used. For this reason, we also looked at the congruence of spatial and ecological relationships, adapting the approach of Hewitt et al. (1998). Briefly, we categorised sampling platforms as congruent if similar spatial or ecological patterns were detected in data acquired from each platform (e.g. strong relationship between richness and depth, or distinct assemblages in a given area compared to others). Statistical analysis of ecological or spatial relationships from separate sampling platforms was done in 28% (42 of 152) of preliminarily selected studies, with a further nine studies qualitatively assessing such relationships from at least one gear type (i.e. no statistical model).

For most combinations of sampling platforms, there were typically less than three studies allowing assessment of ecological congruence, with many pairs of sampling platforms not compared at all (Table 6). As with direct comparisons, AUVs were only represented by a single comparative study (trawling and AUV still images in Przeslawski et al. 2017), and baited direct sampling was not represented by any (Table 6). Most studies allowing comparisons of ecological congruence between sampling platforms were from sleds, trawls, grabs, and corers (Table 6).

The following key points can be gleaned from Table 6:

- Ecological congruence was highest between sleds/trawls and grabs/corers, with nine studies showing congruent ecological patterns (Kaiser et al. 2000, Hirst 2004, Serrano et al. 2006, Ward et al. 2006, Duineveld et al. 2007, Ganesh and Raman 2007, Currie et al. 2009, Atkinson et al. 2011, Jørgensen et al. 2011) and only one showing incongruent ecological patterns (Basford et al. 1990). Other platforms with the most ecological congruence between them are drop cameras and grabs/corers (Grizzle and Penniman 1991, Rosenberg et al. 2003, Wilson et al. 2009), operator-based imagery acquisition and operator based direct sampling (Hewitt et al. 1998, Miller et al. 2003).

- Other platforms that may also yield ecologically congruent results are BRUVs and grabs/corers (Juhel et al.), BRUVs and sleds/trawls (Cappo et al. 2004), BRUVs and baited direct sampling (Gardner and Struthers 2013), AUVs and sleds/trawls (Przeslawski et al. 2017), and UVC and operator-based imagery acquisition (Eleftheriou and Robertson 1992), although there was only a single study within each of these comparisons so further comparative research is required.

- Ecologically congruent results were lowest among UVC and grabs/corers (Eleftheriou and Robertson 1992) and UVC and BRUVs (Gardner and Struthers 2013), although again only a single study represents each of these.

Table 6 Studies identified from refined selection that statistically test ecological or spatial relationships. For studies including three or more methods, each pair of methods is included in the table below.

		Direct Sampling				Visual Methods					
		Grab & corer	Sled & Trawl	Lines & Traps	Operator-based	UVC[8]	Drop	Towed	BRUV	Autono-mous	Operator-based[9]
Direct Sampling	Grab & corer	[1] [2]	[3, 7, 12, 14, 15, 18] [4, 5, 6, 8, 9, 10, 13, 17, 19] [20]		[21] [22]	[23]	[24 25, 26][10]	[27] [18]	[28]		[21] [23]
	Sled & Trawl		[18, 22, 29, 30] [17, 31]					[32, 33] [34] [18]	[35]	[36]	
	Lines & Traps					[37] [38]	[39][11]		[38]		
	Operator-based										[21, 40]
Visual	UVC								[38]		[23]
	Drop										
	Towed							[41, 42]	[6]		
	BRUV										
	Autonomous										
	Operator-based										[16]

(green)	Congruent ecological patterns
(grey)	Undetermined or intermediate (e.g. depends on metric, taxa, etc)
(red)	Incongruent ecological patterns
(white)	No studies found

[8] Includes direct observations (i.e. not digitally recorded) from snorkeler, diver, or submersible passenger
[9] Includes diver, ROV and submersible operations of video or still imagery
[10] Sediment profile imagery is considered a dropped visual platform here
[11] Trap-attached video camera is considered a dropped visual platform here

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5. APPLICATION TO MONITORING

Marine monitoring programs are based on investigating a given indicator over time; this requires repeated sampling. Prior to developing such a monitoring program, a discovery voyage may be conducted to catalogue the species and environment; these are often essential in remote or poorly studied areas. When species have been inventoried, a baseline survey may be undertaken, during which targeted taxa or environments are related to ecological or environmental patterns or high-resolution mapping is undertaken. Components of baseline surveys can form the first sampling period from which repeat sampling events can then follow (e.g. monitoring surveys).

Based on the findings from the literature review, the advantages of each key benthic sampling platform were identified as they relate to marine monitoring surveys (including baseline surveys) (Table 7). In general multibeam shared some of the key advantages of visual methods, with the main unique advantage being suitability over a range of environments, non-destructiveness, and repeatability; however multibeam had the least vessel flexibility of all platforms. Visual methods were characterised by their non-destructive nature and ability to acquire data over a variety of environments. In addition, some visual methods such as AUVs and UVCs allowed repeat visits to exact locations. Direct sampling methods were characterised by species-level identifications, genetic or biological analysis, and suitability for turbid environments. Not all characteristics were clearly defined by platform type. For example, the concurrent collection of physical and biological data is possible with visual (AUV, towed imagery, UVC, BRUV) and direct sampling (grab/box core) platforms.

Table 7 Advantages of key benthic sampling platforms, of which visual methods and direct sampling methods are included in Section 4.

	Acoustics	Visual Methods					Direct Sampling		
	MBS	AUV	BRUV	Towed Vid	UVC	Operator-based	Grab /Boxcore	Sled/ Trawl	Operator-based
Continuous broad-scale spatial coverage	X								
Continuous fine-scale spatial coverage	X	X							
Non-destructive	X	X	X	X	X	X			
Able to revisit exact sites (repeatability)	X	X			X	X			X
Able to sample over variety of environments	X	X	X	X	X	X			X
Species-level identifications of unknown or cryptic species							X	X	X
Genetic, morphological etc analysis possible							X	X	X
Behaviour observed			X	X	X				X
Cryptofauna included							X	X	X
Quantitative	X	X		X	X		X		
Concurrent physical and biological data		X		X	X	X	X		X
Minimal technical expertise			X	X	X		X	X	
Vessel flexibility		X[12]	X	X[12]	X	X	X	X[13]	X
Suitable for deeper waters (> 30 m)	X	X	X	X		X	X	X	X
Suitable in high turbidity	X						X	X	X

[12] Smaller platforms (e.g. *Iver* AUV models) can be deployed from a variety of vessels, but larger ones (e.g. AUV *Sirius*) require larger vessels.

Each sampling platform has a particular variable(s) that it measures, and it is useful to link these to global indicators currently being developed:

- The Global Ocean Observing System (GOOS) has proposed 14 essential ocean variables (EOVs) with a further two classified as emerging (Miloslavich et al. 2018), based on their i) relevance in helping to solve science questions and addressing societal needs; ii) contribution to improving management of marine resources; and iii) feasibility for global measurement in terms of cost, available technology, and human capabilities.
- The Group on Earth Observations (GEO) has proposed several essential biodiversity variables (EBVs) which should be i) able to capture critical scales and dimensions of biodiversity, ii) biological, iii) a state variable, iv) sensitive to change, iv) ecosystem agnostic, and v) technical feasible, economically viable, and sustainable in time (Pereira et al. 2013).

Table 8 lists EOVs and marine EBVs and links them with the capability of benthic and demersal sampling platforms to measure them. Australia undertook a pilot program on Essential Environmental Measures (EEMs) to identify variables necessary for tracking change in the state of environment (https://measures.environment.gov.au). If the EEMs are released, it would be useful to include these in a similar table.

Table 8 The capability of marine benthic sampling platforms to measure EOVs and EBVs. Red = not capable, orange = somewhat capable, green = capable.

	Acoustics	Visual Methods					Direct Sampling		
	MBS[13]	AUV	BRUV	Towed Vid	UVC	Operator-based	Grab/Boxcore	Sled/Trawl	Operator-based
Essential Ocean Variables									
Phytoplankton diversity & abundance[14]	Orange	Red	Red	Red	Red	Red	Orange	Orange	Orange
Zooplankton diversity & abundance[13]	Orange	Red	Red	Orange	Red	Orange	Red	Orange	Orange
Fish abundance & distribution[13]	Red	Orange	Green	Orange	Green	Green	Red	Green	Green
Marine turtles, birds, mammals abundance and distribution[13]	Red	Red	Red	Red	Orange	Orange	Red	Red	Orange
Benthic invertebrate abundance & distribution[13]	Orange	Green	Orange	Green	Green	Green	Green	Green	Green
Coral cover	Red	Green	Orange	Green	Green	Green	Red	Orange	Green
Seagrass cover	Red	Green	Orange	Green	Green	Green	Red	Orange	Green
Mangrove cover	Red	Red	Red	Red	Red	Green	Red	Red	Green
Macroalgal cover	Orange	Green	Orange	Green	Green	Green	Red	Orange	Green
Microbial activity, biomass & diversity[15]	Red	Red	Red	Red	Red	Red	Green	Orange	Green
Essential Biological Variables (classes)									
Genetic composition[16]	Red	Red	Red	Red	Red	Red	Green	Green	Green
Species populations[17]	Red	Green	Green	Green	Green	Green	Green	Green	Green
Species traits[18]	Red	Orange	Orange	Orange	Green	Green	Orange	Orange	Orange
Community composition[19]	Red	Green	Green	Green	Green	Green	Green	Green	Green
Ecosystem function[20]	Red	Red	Red	Red	Orange	Orange	Orange	Red	Orange
Ecosystem structure[21]	Red	Green	Orange	Green	Green	Green	Orange	Orange	Orange

[13] MBS does not usually target biological variables such as the EOVs and EBVs, but rather provides a baseline map from which ecological relationships can be investigated.
[14] Combines two EOVs (abundance and diversity/distribution)
[15] Emerging EOVs
[16] Includes candidate EBVs co-ancestry, allelic diversity, populations genetic differentiation, breed and variety diversity
[17] Includes candidate EBVs species distribution, population abundance, population structure by age/size/class
[18] Includes candidate EBVs phenology, body mass, natal dispersion distance, migratory behaviour, demographic traits, physiological traits
[19] Includes candidate EBVs taxonomic diversity, species interactions
[20] Includes candidate EBVs net primary productivity, secondary productivity, nutrient retention, disturbance regime
[21] Includes candidate EBVs habitat structure, ecosystem extent and fragmentation

Although there are many considerations in choosing appropriate sampling gear (see Table 6 in Rogers et al. (2008), the most important based on our literature review are survey and monitoring program objectives (including variable to be measured and methods used to collect previously acquired data), target environment (including depth and substrate), and available resources. We provide further details on these aspects in the sub-sections below.

5.1 Survey and Monitoring Program Objectives

Sampling platforms should be chosen to most efficiently meet survey objectives, including monitoring program objectives where applicable in which repeatability and longevity of platform use into the future (or adding to previously collected data) should be considered. All of the main platform types (acoustic, direct sampling, imagery) are relevant to marine monitoring. The most suitable platform type depends on the stage of monitoring. Marine surveys are undertaken to acquire baseline environmental data, identify important habitats or taxa, or detect change (including quantifying impacts), each of which is associated with optimal survey designs and sampling platforms. A comprehensive marine monitoring program can include aspects of all of these goals:

1 Baseline data is needed to assess condition and provide a time zero for repeat observations and subsequent trend detection (Lawrence et al. 2015).

2 The identification of important habitats or taxa can guide management priorities and refine regions and metrics for monitoring activities (Przeslawski et al. 2015). This objective can occur concurrently with acquisition of baseline data (point 1 above).

3 The detection of change is based on the previous objectives above and requires repeat sampling. It can inform the efficacy of marine zoning (Kelaher et al. 2014), enforcement (Kelaher et al. 2015), and other management strategies.

For example, seafloor acoustic methods provide a baseline map of the seabed from which a powerful and appropriate survey design can then be implemented, but such methods may not be needed on subsequent surveys to detect change. Direct sampling yields valuable biological specimens, particularly in unexplored areas, from which a species inventory can be derived to inform subsequent change detection. Non-extractive methods such as underwater imagery and visual censuses are currently the most appropriate methods to detect change and quantify benthic impacts due to their capacity to collect true repeat observations, which increases efficiency when estimating the 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.

Marine monitoring programs have their own objectives that overarch and inform the specific survey objectives mentioned above. These objectives can be linked to discrete marine sampling platforms to help researchers decide what gear to use (Table 9).

Table 9: Marine monitoring objectives adapted from (Fancy et al. 2009) with the associated suitable marine benthic sampling platform type (acoustic, visual methods, direct sampling).

Objective	Platform type	Rationale
Determine the status of selected indicators of marine park ecosystem conditions	All	All acoustic, visual and direct sampling platforms provide information that can contribute to an assessment of the current condition of an ecosystem.
Enable early detection of trends and changes to selected resource	Visual	Although direct sampling methods can provide some indication of change, the non-extractive, habitat-focussed nature of visual methods is more suited to change detection as it allows for true repeat observations[22].
Provide data to better understand the dynamic nature and condition of marine park ecosystems and to provide reference points for comparisons with other altered environments	Visual, Direct sampling	Both visual and direct sampling methods can provide data on existing species and communities as related to ecosystem health.
Provide data to meet certain legal mandates related to natural resource protection	All	The most suitable platform will be determined by the specific legal mandate.
Inform the evaluation of management effectiveness	Visual	The non-invasive nature of visual methods is suited to research in areas where extractive sampling may not be permitted or may affect results.
Ensure investments are focussed on management actions that will deliver measurable results	None[23]	na
Inform stakeholders whether monitoring program is on track to address key threats	None[24]	na

[22] Among highly mobile pelagic fauna, some direct sampling methods (e.g. catch and release, e-DNA) are more suited than imagery to detect change
[23] Platforms that involve citizen science programs (e.g. Reef Life Survey) may be suitable for this objective
[24] Visual methods or those involving citizen science provide greater communication opportunities with stakeholders

5.2 Environment

The marine environment plays an enormous role in determining the appropriate marine sampling platform, with depth the primary regulating factor. UVC and operator-based sampling methods (excluding ROV) are only suitable for comparably shallow depths (<30 m). Other platforms can generally be deployed in deep waters, but all imagery equipment must be suitably depth-rated with appropriate light sources, and in many cases a USBL will be needed to ensure accurate georeferencing of the sample or image. Even MBS systems are broadly characterised by depth, as lower frequencies penetrate deeper waters but have less horizontal resolution, so higher frequencies give greater precision on the shelf. As such, each MBS system is associated with an effective operating depth range. In addition to depth, substrate is an important factor in choosing a suitable marine sampling platform. Although multibeam and most imagery platforms (excluding sediment profile imaging) can be deployed over both hard and soft substrata, most direct sampling platforms target a particular substrate type. In a broad sense, grabs and corers are most appropriate in soft sediments, while dredges and many epibenthic sleds are more effective over firmer ground. An ROV is one of the few sampling platforms able to collect specimens from extremely rugose or high-relief environments (e.g. canyon walls). An evaluation of indicators specific to Key Ecological Features showed that diver-based observations and BRUVs were preferred in shallow reefs, while trawls and sleds were preferred in deeper waters; imagery platforms (AUV, towed video, ROV) were appropriate for both shallow and deeper shelf waters > 20 m (Hayes et al. 2015).

5.3 Available Resources

Ultimately, the sampling gear most appropriate to survey objectives and environment may not be chosen because there are more pressing constraints on available resources such as equipment availability, expert availability, existing data and cost.

5.3.1 Equipment Availability

Marine sampling platforms vary in their level of complexity and consequently availability in Australia. UVC requires no specialised equipment, usually just standard dive gear, a transect reel, underwater slate and camera. Towed imagery systems, grabs, sleds, box corers, and BRUVs are relatively easy to build as fit-for-purpose gear, and there are numerous such platforms already available in Australia. In contrast, MBS, AUVs, and larger ROVs are much more complex and are very costly or require advanced technical expertise to build. As such, the number of these platforms in Australia is limited and may be a potential bottleneck to their widespread use.

5.3.2 Expert Availability

Several sampling platforms require experts to underpin successful data acquisition. UVC requires certified divers trained to identify local species *in situ,* although citizen science programs such as Reef Life Survey suggest that the availability of trained divers is sufficient to maintain successful global monitoring activities. MBS, AUVs, and ROVs often require a technician or technical team for installation, calibration, and deployment and the availability of such experts is a potential constraint. In contrast, towed imagery systems, BRUVs and direct sampling platforms (e.g. sleds, grabs, corers, small ROVs) do not require complex calibration and can be deployed by the ship's crew using standard operating procedures for a given vessel (e.g. winch operation).

5.3.3 Existing data

If previous data exists that can be used as a baseline or part of a time-series, the sampling platform employed in the original study should be used again, assuming it is suitable for current monitoring objectives (Section 5.1) and environment (Section 5.2). This ensures that data can be comparable across survey periods. For example, a meta-analysis of marine range shifts highlighted the efforts of several studies to consider potential error among surveys by employing the same sampling methods in both historical and recent surveys (Przeslawski et al. 2012).

5.3.4 Cost

Cost can include equipment purchase or hire, calibration, vessel, staff, sample or data processing, and training. Rather than deal with all of these separately, we have attempted to synthesise them below using published accounts.

Acoustics

The purchase cost of a multibeam system is higher than any other platform mentioned here, with the possible exception of large multipurpose AUVs and ROVs. In addition, many systems require a comparatively large vessel with particular specifications for their installation (e.g. moonpool), as well as commercial software and trained technicians for data processing. There are very few peer-reviewed accounts of the cost of marine acoustic mapping, and none of these detail the total costs (from equipment purchase through to data processing) (but see NOAA 2005 for methods behind comprehensive unreleased cost analysis). In addition, costs per unit square for MBS are non-linear due to variation in coverage among depths and environments (NOAA 2005). Therefore, generating an assessment of the cost-effectiveness of multibeam and other acoustic platforms is impossible other than for a survey-specific scale.

Nevertheless, there are some published accounts of time, money, and resources for mapping that can inform the decision to use MBS. In 2012, the average cost to collect multibeam data was estimated to be USD$1000 per ship hour (Price et al. 2012). Lawrence et al. (2015) estimated that it would take between 3.5 and 17.5 years to map the 306, 627km^2 shelf region (40 – 200 m) of the Australian Marine Parks, assuming non-stop vessel operations. More recent unpublished values estimate that mapping of the AMPs (including all depths) would take 23 years non-stop, assuming 8 knot acquisition speed (IXSurvey, pers. comm.). Processing times have been shown to vary between 2-68 minutes per km of swath, depending on beam angle, technical expertise and experience, survey type, and seabed complexity (Abdelrahman et al. 2012). Ultimately, the cost-effectiveness of multibeam data can be maximised by improving data access and re-use, i.e. 'map once, use many times' (Price et al. 2012).

Visual Methods

As with other marine sampling platforms, there is little published information about the overall cost associated with visual methods. However, imagery analysis seems to have even more variability in cost than other platforms due to the different speed, resolution, and accuracy at which observers analyse images (Durden et al. 2016a).

A comprehensive BRUV analysis (from video acquisition to image analysis to data release) may cost approximately $350 per deployment, although this will vary depending on vessel costs and density of fish (E. Harvey, pers. comm.). Unpublished results from a Parks Australia workshop on cost-effectiveness of various marine sampling platforms show a higher estimate of approximately $550 per BRUV deployment in temperate east AMPs (A. Richley, pers. comm.). This same workshop also costed UVC and found that cost substantially varied among suppliers between $200 (Reef Life Survey) and $700 (commercial rates) per transect. For demersal fish, Langlois et al. (2010) found that BRUVs were consistently more cost-effective than diver-operated video in detecting changes in species richness, while Colton and Swearer (2010) found that UVC was more efficient at sampling species richness than BRUVs.

Post-processing of imagery remains one of the bottlenecks for many visual methods, with many systems capable of collecting thousands of images or hours of video on each deployment. An efficiency analysis of still and video processing methods showed that video imagery required more effort to collect, process, and extract (39 minutes per quadrat) while also having the poorest taxonomic benefit (0.15 species per min) (van Rein et al. 2012). Of all visual methods, UVC requires the least amount of time (and cost) for post-processing, although initial training costs may be higher to ensure accurate identifications from divers.

Direct Samplers

Although the purchase and maintenance costs of most direct sampling platforms are comparatively low, costs regarding sample processing can be high. There are several choices to be made regarding direct sampling that affect the time and financial cost of data acquisition and analysis. The most notable of these are sieve size and taxonomic resolution, of which several comparative studies on data acquisition (sampling), processing (sorting) and analysis (identification) are described below:

- The actual deployment times of direct sampling gear vary according to depth from a few minutes in shelf waters to hours in abyssal waters. In addition, deployment times are generally longer for transect-based platforms than point-based platforms. Rogers et al (2008) spent 6-24 min deploying grabs and corers and 24-60 min deploying a variety of demersal trawls. However, once onboard processing times are added (e.g. sieving, elutriation), sampling times can increase, ranging from 0.3-1.9 hours for a grab and hand corer including sample collection and sieving (Lampadariou et al. 2005).
- Sorting the fauna from the rest of the haul may require substantial time investment, particularly for grabs and corers, ranging from 2.5 – 27.1 hours for one sample, depending on sieve size, sample volume, and sampling gear (Lampadariou et al. 2005). Some studies

showed that sample sorting times can increase 22-44% more with a finer sieve (0.5 mm) compared to a coarser sieve size (1 mm) (Thompson et al. 2003, Lampadariou et al. 2005), while another stated that sorting time took four times longer with a 0.3 mm sieve compared to a 0.5 mm sieve (Daniell et al. 2009). To reduce the retained sediment fraction and associated sorting times, elutriation is often used in which sediments are rinsed to suspend fauna in seawater which is then passed through a sieve.

- Taxonomic identification is often the costliest stage of direct sampling, particularly with samples of high species richness or rarity, or unknown species inventories. However, costs of taxonomic identification are difficult to estimate due to variation in taxonomic expertise, richness in each taxonomic group, and revision status of each group (Ferraro and Cole 1995). By using a lower taxonomic resolution such as family, time savings of 33% (Lampadariou et al. 2005), 40% (Thompson et al. 2003), or 55% (Ferraro and Cole 1995) have been estimated. Even sorting to family or class requires training and specialisation that should be factored into a complete costing.

In addition, the need for extra staff for technical or risky operations may increase costs. For example, (Aguado-Giménez et al. 2007) found that the staff and equipment costs of sampling soft sediment communities using SCUBA was 0.6 – 1.2 times higher than those of equivalent sampling using a Van Veen grab. Ultimately the total time needed to produce a dataset from a single sample can be lowest for meiofauna (10 hours) and megafauna (6-12 hours) sampled by corer and trawls, respectively, and highest for macroinfauna sampled by grabs (12-22 hours) (Rogers et al. 2008). Assemblage data may be more suited to monitoring programs, as such multivariate data has been shown to require less replicates than univariate (Rogers et al. 2008) and may be more suited to lower taxonomic resolution (e.g. family-level) (Lampadariou et al. 2005), both of which reduce sampling effort and associated cost.

6. CONCLUSIONS AND RECOMMENDATIONS

To maximise efficiency and to promote concurrent collection of data, combining multiple gear types onto a single platform may be appropriate (e.g. camera-mounted epibenthic sleds in Rice et al. 1979). This approach is particularly effective in deep waters where gear can spend hours in the water column before it reaches the seafloor and it can be difficult to determine whether the gear is correctly deployed without additional information. However, there may be trade-offs associated with a combined gear approach, in that the optimal configuration for one platform may not suit another (e.g. positioning of camera system on sled precludes acquisition of downward-facing imagery). Notably, platforms can be made to be flexible, and combined gear types may require increased complexity for deployment (e.g. towing cable, size of ship, winches and hoists).

Technology is rapidly advancing in acoustics and imagery-based sampling platforms (e.g. data processing methods such as imagery annotation and automated classification), but the incorporation of these advancements in marine sampling protocols is not yet clear-cut. On one hand, innovative methods such as listed in Section 2 may ultimately yield a more cost-efficient and effective tool that eventually becomes standard practice (e.g. AUV). On the other hand, monitoring programs should not rely on the latest and greatest technology, as such platforms mature rapidly and the attributes of data with it. This 'breaks' time-series for detecting trends and means that repeated measurements are not consistent. If these platform's technologies are used then future planning can help maintain a detailed quantitative comparison between legacy technology and innovative technology. Importantly, data collected today needs to be kept in as raw form as possible so that it can be reprocessed as new technologies develop.

By compiling all comparative studies of benthic or demersal biological sampling platforms, we were able to summarise whether gear pairs collected statistically different samples and whether gear pairs collected samples showing similar ecological relationships (i.e. ecological congruence). Statistical difference is relevant for both baseline and monitoring surveys so that the number of communities or habitats being characterised or monitored in a given program is maximised based on available resources (Section 5). Ecological congruence is important for baseline surveys in which general biogeographic patterns may be investigated. Overall, samples from BRUVS showed the most statistical difference of all other methods (Table 5). This places BRUVs as a strong contender for inclusion in baseline and monitoring surveys in which multiple platforms are to be deployed. Notably, many platforms were unable to be similarly assessed due to a low number of comparative studies.

Almost all benthic monitoring programs require high-resolution mapping to be conducted prior to biological sampling. This ensures a spatially balanced sampling design can be developed, provides context for interpreting biological data, and facilitates suitable comparison sites (e.g. similar depth, substrate). After a given area has been mapped in detail, multibeam surveys are not usually required again unless an assessment of seabed stability and geohazards is needed (Przeslawski et al. 2011).

There is no universal method appropriate for all marine sampling; a one-size-fits-all approach is neither feasible nor desirable in monitoring programs and associated baseline surveys (Przeslawski et al. 2016). Even the decision to deploy multiple sampling platforms depends on the survey objectives. For surveys collecting baseline or descriptive information, a diversity of gear may be more appropriate (Uzmann et al. 1977, Daniell et al. 2009, Nichol et al. 2013). These varied platforms will provide a broader species inventory and associated biogeographic patterns from which to then target suitable indicators, habitats, and locations for monitoring. In contrast, fewer platforms capable of repeatable sampling would be more appropriate for monitoring surveys (Smale et al. 2012, Ling et al. 2016a, Stuart-Smith et al. 2017), as they can target similar indicators while providing increased sample size and spatial coverage than would be possible in the same timeframe with many sampling platforms

This comparative assessment provides information that can be used to guide marine sampling activities as they relate to monitoring objectives. Such information is crucial to ensure cost-effectiveness and efficacy of marine monitoring activities, specifically that an appropriate method is being used with appropriate knowledge of its limitations and challenges. Regardless of the marine sampling platforms that are chosen, robust survey designs and standard operating procedures are necessary to ensure consistency of data and comparability over time and space (e.g. *Field Manuals for Marine Sampling to Monitor Australian Waters,* Przeslawski and Foster 2018).

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