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The Role of Rapid Environmental Assessment in Monitoring the Environmental Impacts of Deep-Sea Mining

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The Role of Rapid Environmental Assessment in Monitoring the Environmental Impacts of Deep-Sea Mining

Abstract
We here consider the extent to which Rapid Environmental (or Ecological) Assessment (REA) can be applied to the deep-sea benthic environment and the associated water column as a tool for surveying very large areas to assess the risk of environmental impact from deep-sea mining. In particular, we propose that REA could be conducted more widely through video and still imagery collected by the various platforms employed in benthic and pelagic studies. REA protocols, as developed for the assessment of large areas of shallow-water marine habitat, and used on land, have in common that they accept semi-quantitative estimates of the abundances of biological taxa and of the strength of environmental factors, these often being assessed on a predefined 5-, 6-, or 10-point scale. REAs also accept taxonomic identification limited to higher taxonomic levels or even morphospecies level and may restrict identification to preselected orders or families, focusing on indicator and sentinel species, preferential habitats, and trait-based indicators.

While estimates of abundance may be made with lower accuracy than achievable given unlimited resources, REA allows much greater areas to be monitored and can result in greater precision and statistical sensitivity if it results in much greater numbers of replicate samples being processed. Metrics of biodiversity such as taxonomic distinctiveness can also be applied to the data, as can indices of environmental sensitivity which, when combined with monitoring of indicator taxa, can be used to provide a traffic-light system indicative of ecosystem health. REA, relying on imaging, is proposed to be used in conjunction with operational oceanographic systems covering the entire water column and other emerging technologies, such as environmental DNA, acoustics, bioluminescence monitoring, multi-parametric satellite tagging, and hydrodynamic modelling, that may prove effective for monitoring extensive ocean areas in the face of environmental change. With the progress of machine learning to manage large datasets of taxa, habitat identification and oceanographic concepts developed by experts, annotation efforts can be facilitated, datasets formatted and standardised, and models developed. We also highlight the potential role that citizen scientists/volunteers and indigenous peoples can play in REA if adequate training is provided and well-constructed protocols are available. The necessity of in situ long-term multi-parametric observation platforms is highlighted as is the standardisation of REA worldwide, enabling comparisons and joint management decisions as oceanographic processes occur on regional scales.

Keywords Rapid Environmental Assessment · Pelagic ecosystems · Video image analysis · Machine learning · Operational oceanography · Deep-sea mining

1 Introduction

1.1 Deep-Sea Mining

The World Bank estimates that over three billion tons of new metals will be needed to implement the necessary wind, solar, and energy storage technologies required to limit climate change to below +2 °C (World Bank, 2020; Hund et al., 2023). To meet these demands, exploration for new and sustainable sources of metals, particularly nickel, cobalt, manganese, copper, zinc, and rare earth elements, is essential. However, the current supply of minerals from onshore sources struggles to keep pace due to several challenges (Rötzer & Schmidt, 2020; Xeni, 2021) such as: • Limited Onshore Reserves: Many land-based mineral reserves are heavily exploited or located in regions with geopolitical or environmental challenges.

• Environmental and Social Concerns: Expanding onshore mining often faces opposition due to the pollution generated, habitat destruction, corruption, and social impacts on local communities.
• Lengthy Development Timelines: Onshore mining projects can take 20–25 years to develop, being constrained by complex regulations and substantial infrastructure requirements.

On the deep seafloor, these metals are contained within slow-forming polymetallic nodules (black spherical oxide deposits that occur on abyssal seafloor), as well as in polymetallic sulphides (large deposits made up of sulphur compounds and other metals that form around hydrothermal vents) and metal-rich crusts (found as oxide layering on rocks exposed on seamounts). One promising solution is the exploitation of these deep-sea minerals, with the global ocean floor being estimated to contain more than five times the amount of cobalt found on land. The seabed of the Clarion Clipperton Zone (CCZ) of the East Pacific Ocean alone contains over 21 billion metric tons of nodules (US Geological Survey, 2024; Wang et al., 2024). If managed responsibly throughout the entire value chain, deep-sea minerals can diversify the global supply of these critical resources and play a key role in supporting the energy transition. As the industry evolves, fostering a shared understanding of Environmental, Social, and Governance (ESG) responsibilities is crucial.

Transparent ESG disclosures will help document performance, promote accountability, and build trust among stakeholders, ultimately supporting the social license to operate. As international and national regulations, such as the European Union's 2023 Corporate Sustainability Reporting Directive (CSRD)1[1] mature, marine mineral developers must align their practices with evolving compliance requirements.

Promoting responsible practices and ensuring transparency will allow the deep-sea minerals industry to address global demands for critical metals while safeguarding ecosystems and communities.

Currently, the deep-sea mineral industry remains in the exploration phase, with no commercial extraction underway. The International Seabed Authority (ISA) which regulates the activities associated with the development of seabed resources in the international waters has issued licences only for exploratory work. However, individual countries have the power to approve licences within their own Exclusive Economic Zone (EEZ), as some countries have done, e.g., the Cook Islands and Papua New Guinea. International regulations for mineral exploitation beyond national jurisdictions are still under development by ISA. Meanwhile the ISA has established exploration guidelines for polymetallic nodules, polymetallic sulphides, and cobalt-rich ferromanganese crusts. These guidelines include recommendations for assessing environmental impacts and ensuring responsible practices (ISBA/25/LTC/6 Rev.3 Legal and Technical Commission).

It is now widely appreciated that deep-sea mining has the potential to cause significant widespread marine environmental impact (Tilot, 1988, 1989; Ahnert & Borowski, 2000; Sharma, 2005; Tilot, 2010; Tilot et al., 2018). Nodule fields harbour surprisingly diverse benthic communities, with many species new to science (Tilot, 2006a; Glover et al., 2002; Wang et al., 2010; Ramirez-Llodra et al., 2011; Janssen et al., 2015; Paterson et al., 2015). Key faunal groups within the CCZ are the cnidarians, echinoderms, and sponges among the megafauna (as defined as organisms (> 1–4 cm) that are visible in photographs of the ocean floor (Grassle et al., 1975; Gage & Tyler, 1991) and polychaete worms, nematode worms, and protozoan foraminifera among the macrofauna (including 80–100 species per square meter) (Amon et al., 2016) and meiofauna. These taxa represent over 50% of faunal abundance and species richness in abyssal sediments and display a broad range of ecological and life history types. Not only deep-sea mining directly impacts epi-benthic communities, but given that the particles in sediment plumes and mine tailings are relatively fine, they are likely to remain in suspension in the water column for extended periods of time and disperse over very large areas of ocean. Disturbances at the point of collection and return water outlet include sediment plumes that cause oxygen depletion, increased turbidity, possible dispersion of pathogenic material, and increased proportions of mineral particles, leading to nutritional deficiencies and repercussions at all trophic levels of the food web (Christiansen et al., 2020; Drazen et al., 2020). Additionally, deep-sea mining will impact the water column as ore is pumped upward as concentrated slurry which could be washed at different depths. Moreover, the release of return water, composed of seawater and low-concentration seabed sediments, either at midwater or near the seabed before the ore is shipped to a terrestrial processing plant (Miller et al., 2018) could also cause negative effects on the marine ecosystem.

1 https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32022L2464

In consequence there have been widespread calls for a moratorium on deep-sea mining until the potential effects of disturbance to benthic habitats, sediment plume dispersal, and tailings disposition have been more fully investigated on the benthos and in the water column. Yet, even though about 25% of the world's global seafloor has been adequately mapped, only 5% of the abyss (3000–6500 m), which constitutes 60% of the global ocean, has been explored[2]; the ISA apparently expects to finalise its exploitation regulations in 2025.

Therefore, there is an increasingly urgent need to standardise environmental monitoring protocols at a global level to ensure that adjacent regions and habitats, that may underpin ocean productivity and buffer against climate change, are not unnecessarily impacted (Tilot et al., 2010). In this chapter, we start the discussion with examples from shallow water monitoring methods (e.g. Corals, fish) and go on to describe the Regional Environmental Assessment process (REA) and environmental surveys in deep-ocean targeted by deep-sea mining, followed by image and video analysis techniques, REA in water column, as well as integration with next-generation monitoring methods.

1.2 Environmental Monitoring of Marine Ecosystems

Effective monitoring of marine ecosystems, on the scale required for deep-sea mining or to assess global change, will necessitate a multi-disciplinary approach and the deployment of a comprehensive suite of tools. In-depth investigation is required in such varied fields as carbon chemistry, physical oceanographic modelling, biogeochemistry, species biodiversity, benthic ecology, and fisheries. Background data is desirable on parameters including water temperature, salinity, acidity, oxygen concentration, currents, sediment loads, and carbon content, all across a range of depths extending from the surface to 5000 m or more. In addition, information from oceanographic buoys and satellite-based sensors should prove invaluable. These deep ocean regions, characterised by great hydrostatic pressure, low temperatures, and complete darkness, present unique challenges for habitat assessment, biodiversity, and adaptation studies as well as the overlying water column in the high seas.

As a consequence, the more direct task of assessing and monitoring, at the required scale, the effect of mining activities on benthic and pelagic habitats and biota has yet to be fully grasped, given methodological issues that are not necessarily evident to all parties. Specifically, while biological studies of the deep benthos have revealed a diverse micro-, macro-, and megafauna, often in associated communities, and large numbers of species new to science, the subject has not been adequately addressed regarding how to survey and monitor deep-sea habitats and biota on the scale required. Until recently, most studies have been undertaken by taxonomists with limited experience in landscape scale surveys and ecology applied to management and conservation, and a majority of studies have been limited to comparatively very small areas of seabed; recording images along relatively few transects or taking cores from a limited number of sites can inevitably sample only a tiny proportion of the benthic habitat concerned and lead to misinterpreting the structure and functioning of deep-sea faunal communities.

In this chapter, we will consider as to how the experience gained from surveying and monitoring shallow water habitats, such as coral reefs and associated communities, can be used to improve the cost-effectiveness of deep-sea benthic surveys, in particular, the effectiveness of monitoring using still or video camera carrying platforms such as remote

2 https://eos.org/articles/new-seafloor-map-only-25-done-with-6-years-to-go

observation vehicles (ROVs), sledges, and seabed landers combined with new technologies for surveying the benthos and the water column to the surface and above.

Specifically, we shall consider the extent to which Rapid Environmental (or Ecological) Assessment (REA) can be applied to the deep-sea benthic environment as a tool for monitoring very large areas of seabed, as previously proposed (Tilot, 2010, 2014; Tilot et al., 2018). This will be particularly through its application to video and still imagery collected by various platforms used to investigate deep-sea benthos. REA protocols, originally developed for surveys of terrestrial vegetation (Braun-Blanquet, 1932, 1964; Tait and Dipper, 1998; Rodwell, 1998, 2006), then large areas of shallow-water marine habitats (Price, 1999; Tilot, 2007), have in common that they accept semi-quantitative abundance estimates of biological taxa and of environmental factors, often assessed on a predefined 5- to 10-point scale (Crisp & Southwood, 1958; Tait and Dipper, 1998; Strong & Johnson, 2020).

To assess biodiversity or monitor the presence of keystone trophic groups, REAs typically restrict species identification to preselected orders or families, such as fish or seabirds, which are more readily detected and to categorised habitats (Peres & Picard, 1958; Hourigan et al., 1988; Roberts et al., 1988; Montevecchi, 1993; Whitfield & Elliot, 2002; Price, 2004). While individual estimates may lack complete accuracy, statistical methods can be applied to the data, enabling quantitative changes or differences to be detected with some confidence. If photographic and video methods are used, provided all the imagery is archived, imagery for particular sites or taxa can be re-analysed in greater detail at a later date. Similarly, if water samples are properly stored, additional analysis, e.g., of e-DNA, remains possible.

REAs can be used to generate reference levels for management decisions, with clear protocols developed for evaluating the trade-offs needed for the spatial planning and ecosystem-based management of marine environments (Link & Browman, 2017), such as for managing the nodule field regions of the CCZ.

In contrast, Alert Systems, that warn of the need for remedial action, remain novel in their application to marine areas, although the identification of criteria based on scientific knowledge, to assess endangered and critical habitat, thresholds of tolerance, was used for managing marine areas and resources since the 1960s–1970s (Humphreys & Clark, 2020). Already, they were part of the management tools boosted by the fourth World Congress on National Parks and Protected Areas, organised in Caracas (Venezuela) by IUCN, UNEP, and UNESCO in February 1992, where was expressed the need for monitoring end evaluation of the status of the sites, for a better evaluation of the impacts of the protection on the environmental, social, and cultural environment, principles that underlie the concept of integrative marine resource management (Levine et al., 2015).

Alert Systems were originally designed for decision-makers rather than technical specialists. They identify multiple thresholds and generate easy-to-interpret outputs using a traffic lights system (Halliday et al., 2001; Ceriola et al., 2007; Barange et al., 2010; Price et al., 2014b). Alert systems have been proposed, in particular, in relation to coastal areas and islands in the Mediterranean Sea, Red Sea, Arabian Gulf, and the Indo-Pacific Ocean (Jeudy de Grissac, 2002, 2003, 2007; Price et al., 2014a; Tilot, 1996, 2002a, 2016; IUCN/ROWA, 2016). Levels of key biological (e.g., live coral cover), physical (e.g., sediment load), or chemical (e.g., oil hydrocarbon) parameters can be set not only to trigger management action to halt damaging activities, such as seabed dredging or an oil leak, but also to monitor the effectiveness of restitution activities (e.g., oil spill clear up) until the previously determined values are restored.

2 Shallow Water Surveys and Monitoring Methods

Already in the 1950s, marine surveys performed by snorkelling, then by SCUBA, were based on a bionomic manual referencing benthic habitats associated to faunal communities of the Mediterranean Sea, North Sea, and Atlantic Ocean (Peres & Picard, 1955, 1958, 1964). Progressively scientists and managers have adapted their sampling strategies to formalise the use of rapid environmental assessment protocols incorporating semi-quantitative methods, categorised habitats for conservation surveys (Hayne, 1949), and thresholds for impact studies, as tools for spatial management planning.

We shall review some of the methods used in coral reef and colder-water marine studies to inventory or monitor large habitat areas, before discussing how these approaches might be applied to the deep ocean.

2.1 Coral Reef Survey and Monitoring Methods

While quantitative ecological surveying and monitoring of terrestrial and intertidal habitats have a history spanning a bit more than 100 years (e.g. Pearsall, 1924; Ashby, 1935), the first marine surveys were conducted earlier through snorkelling, such as the transect line monitoring held in Pago Pago Harbour (Mayor, 1924) and the pioneering surveys of Fischer-Piette (1936) in the English Channel. With the advent of SCUBA diving, monitoring strategies started to develop, for example, the assessment of urban impacts on coral reefs in Samoa (Dahl, 1977; Dahl & Lamberts, 1977). In New Caledonia, a monitoring programme was established to identify and evaluate the principal components of the coral reef communities, basic water quality and pollution indicators (Dahl, 1981a). Similarly, in the Maldives, semi-quantitative methods derived from socio-phytology were developed, relying on visual assessment of quadrats ranging from 1 to 10 m in size (Scheer, 1978).

Quantitative surveys became standard after Loya (1972) described a line transect methodology used to survey subtidal reefs along the Israel coast of the Gulf of Aqaba. This Line Intercept Transect (LIT) technique (Fig. 9.1a, b) records, centimetre by centimetre, the substrate or biota present beneath a weighted transect line run horizontally along the reef across successive depth contours. In the 1980s, long-term monitoring programmes were established to detect ecological interactions and any environmental change, such as those initiated, in New Caledonia (Dahl, 1981b), on the Great Barrier Reef, Australia (Done, 1982), in the Caribbean at sites in Jamaica (Hughes, 1994), and throughout the Western Indian Ocean countries with the EU programme "Regional Environmental Programme of the Indian Ocean" (PRE-COI/UE, 1998). These programmes also included the UNEP/FAO/EAF-5 project for the "Protection and management of the coastal and marine areas of the East African region (FIR Comoros, Mauritius, Rodrigues, Seychelles, Madagascar, La Reunion) (1995–2002) (Tilot, 1994), the Gulf of Aqaba and the Egyptian offshore islands with EU/Government of Egypt/EEAA for the National Parks of Egypt Protectorates Development Programmes project (1988–2002), particularly the video-monitoring programme (Jeudy de Grissac, 1999, 2002; Tilot-de Grissac et al., 2000; Tilot, 2002b, 2003b; Tilot et al., 2008), in Yemen with the Ministry of Planning and Development of Yemen/UNDP GEF (Tilot, 2002a), in Oman in Al Damyanat islands (Tilot, 2017), in the Philippines with the Museum National d' Histoire Naturelle de Paris" (1999–2000) (Tilot, 2003a) and Eritrea with the project UNDP/ECMIB/GEF to assess the marine biodiversity of the Dahlak archipelago in a sustainable manner (Jeudy de Grissac, 2006, 2007; Price & Tilot, 2009; Tilot, 2006c; Tilot et al., 2008, 2012).

Fig. 9.1 (a, b) Left, Diagram illustrating how coral cover is recorded using the Line Intercept Transect method. The substrate is recorded along the transect represented by the horizontal dashed line and only the distances a to d recorded, where the line is immediately overlying the life forms (represented by stippled areas), i.e., it is not the full or maximum widths of the life forms that are recorded (From Loya, 1978). Right, one of the authors video monitoring along a transect line in the red sea (Tilot, 2003) filming a video transect by slowly moving along the transect while pointing the camera at the reef perpendicular to the substrate. Photo by Rupert Ormond

In 1992, the Caribbean Coastal Marine Productivity Program (CARICOMP) (www.uwimona.edu.jm/centres/CMS/caricomp/) introduced long-term monitoring at 25 sites of coral reefs and other marine ecosystems, including mangrove and seagrass habitats (Cortés et al., 2019). Following this, a Global Task Team on the Implications of Global Climate Change on Coral Reefs (UNEP, IOC-UNESCO, ASPEI, IUCN) developed monitoring guidelines for a global programme (Wilkinson, 1993; Wilkinson & Buddemeier, 1994). A Survey Manual for Tropical Marine Resources was subsequently published by the Australian Institute of Marine Science (AIMS) (English et al., 1994). In 1995, The International Coral Reef Initiative promoted the establishment of the Global Coral Reef Monitoring Network (GCRMN), which led to the publication of an improved survey manual (English et al., 1997).

However, due to unavoidable logistical constraints, and limitations imposed by SCUBA dive time, all these research programmes were restricted to a limited number of sites and depths. Hence although invaluable in characterising the reef communities at a range of sites, and contributing to the understanding of impacts on the reefs, such as sea urchin (Diadema spp.) die-offs, outbreaks of coral-eating crown-of-thorns starfish (Acanthaster planci), and climate change linked coral bleaching, they are not suitable for assessing or monitoring reefs over hundreds or thousands of kilometres. As a result, some scientists developed a number of rapid assessment techniques.

The need to survey and monitor on a much larger scale (to identify sites of species significance or areas subject to localised impacts) than is possible using fully quantitative techniques led to several developments. First reef managers developed simplified methods that were quicker to apply, and second, they increasingly utilised teams of volunteers or citizen scientists to assist in surveying and in image analysis. The most well-known of these initiatives are the Reef Check programme (www.ReefCheck.org) established in 1997 and the Atlantic and Gulf Rapid Reef Assessment (AGRRA) programme widely used in the Caribbean (www.coral.noaa.gov/agra/). These are science-based programmes designed to provide managers with rapid assessments of reef health and data indicative of potential causal relationships. However, the Reef Check coral survey method has been criticised as data inefficient, as points are sampled at only one- or half-metre intervals along a transect line, compared to continuous sampling in the standard Line Intercept Transect (LIT) (Leujak & Ormond, 2007). AGGRA identifies the organisms at 10 cm intervals along transect lines (Lang et al., 2010), which is likely more data-efficient than either Reef Check or the continuous LIT method.

2.2 Use of Photography and Video in Coral Reef Monitoring

Researchers have also been influenced in their choice of methods by developments in underwater camera technology. When Loya (1972) introduced the LIT method, underwater cameras could only take a small number of images (12 or 24) per dive, often with the aid of a flash gun or strobe, while underwater video cameras were bulky and expensive and produced indifferent results recorded on film. Now, modern point-and-press digital cameras can be used in affordable underwater housings to take hundreds, if not thousands, of photos along line or belt transects. Since normal photographic images are more suited to recording the contents of quadrats (rectangular substrate samples), rather than the narrow strip underneath a transect line, many researchers have turned to capturing photo-quadrats placed sequentially or at 2 or 5 m intervals along a baseline. Other researchers have trialled the use of underwater video cameras to capture close-up recordings of the reef along the full length of each transect (Fig. 9.1b).

However, experience has shown that the resolution of most video cameras is inferior to that of still images, which, combined with the movement of the camera, makes it difficult or impossible to accurately identify corals and other species. By contrast, close-up still images of 0.5 m square quadrats obtained by a competent SCUBA diver allow most corals to be identified at the species level.

The development of photogrammetry applied to coral reefs has accelerated rapidly in the last 15 years, with as many as 55 metrics in 10 categories being extracted from the imagery to inform studies of habitat structural complexity, ecosystem condition, and trajectory (Remmers et al., 2023). Notably, several teams have adopted the use of high-resolution still cameras held by a free-swimming diver to take high-frequency still photos, often at two magnifications, from which detailed photo-mosaics of reef areas can be constructed using appropriate software (Lirman et al., 2007). In relatively shallow water, these photo-mosaics can be linked to aerial imagery obtained by a plane or drone. Furthermore, Suan et al. (2025) tested successfully on four distinct reefs an approach that integrates drones, various colour space information, and deep learning neural networks (AI) to design a 3D image of a coral reef system across large (thousands of meters) spatial scales, thus providing a more cost-effective understanding of the complexity of reef habitats for conservation and management purposes.

2.3 Identification of Taxa and Substrate in a Citizen Science and Participatory Approach

Species of large invertebrates and fish that have been identified from surveys in shallow water, or that have often been photographed in situ, e.g., most reef fish and corals, may be classified with some confidence. For example, during manta board surveys, or scuba diving surveys of the Great Barrier Reef, observers (scientists/volunteers/citizen scientists) are being towed rapidly over the substrate to identify lifeforms and benthic organisms, e.g., hard corals being distinguished as table-form, branching, encrusting, or massive.[3] A similar approach has been taken in citizen science projects such as Reef Check. Less experienced volunteers or local staff could use growth or lifeforms to record quadrat or transect data, more experienced observers identify corals mostly to genus, and experts could identify most corals to species. Flexibility was incorporated in the design of the data sheets used for analysis; relevant families were listed in successive lines below the appropriate lifeform and the most common relevant species in successive lines under the appropriate genus. Thus an observer can enter the presence of a coral, either by its lifeform or its genus or its species, according to their confidence in identification. Results are then revised by experts.

3 https://www.aims.gov.au/research-topics/monitoring-and-discovery/monitoring-great-barrier-reef/long-term-monitoring-program

Horton et al. (2021) addressed the same problem in relation to the difficulty of identifying organisms in low-resolution imagery from the deep ocean. They propose the use of an Open Nomenclature (ON) which allows the annotator to acknowledge taxonomic uncertainties by using standardised qualifiers when an organism cannot be identified with certainty. For example, they propose that terms such as “cf.” (confer) or “aff.” (affinis) to be used in a standardised way when definitive identification is not possible. This approach allows the inclusion of valuable biological data in analyses even when exact species identification is not possible. By standardising the reporting of uncertain identifications, ON mitigates the limitations imposed by AUV imaging systems and ensures that the data collected remains meaningful for habitat and biodiversity assessments despite the challenges of achieving fine-scale species resolution.

The survey manual published by the Australian Institute of Marine Science (AIMS) includes a classification of lifeform categories of tropical marine resources (English et al., 1994, 1997, with updated versions), one page of which is displayed in Fig. 9.2b. As for sediments, the Wentworth grain size chart is often used to classify sediments (Fig. 9.3). This AIMS manual served as a model for other regions of the world, such as for the Western Indian Ocean (Tilot, 1997). It has been compiled during monitoring surveys assessing the health of coral reefs in the Western Indian Ocean performed from 1994 and 2000 as part of the PRE/COI/ EU and FAO/UNEP/RAC SPA/IUCN programmes. It also served as a tool for capacity training in the region with site surveys in FIR Comoros, Madagascar, Mauritius, Rodrigue, and Seychelles (Tilot, 1998). A more recent coral reef monitoring manual has been assembled by Obura (2014) for the SW Indian Ocean islands Global Coral Reef Monitoring Network (GCRMN) node through the ISLANDS project Coral Reef Facility.

Fig. 9.2 (a) (left), A visual tool used to assess the % of coral cover, adapted from Dahl (1981b) for algal cover. (b) (right), a page from the Australian Institute of Marine Sciences manual for Manta Board surveys of the Great Barrier Reef, illustrating some of the lifeforms used to categorise benthic cover [available under Creative Commons]

2.4 Statistical Effectiveness of Reef Surveys

As emphasised by Underwood and Chapman (2013, p. 18), to be of value to science or conservation, “surveys must always be designed to take into account the fact that benthic animals and plants are extremely patchy in distribution and abundance,” a patchiness understood to be particular due to variation in recruitment from dispersal phases, which can vary from place to place and year to year. It is also widely acknowledged, following the review by Hurlbert (1984), that ecological studies must be appropriately replicated, both sufficiently and at the appropriate scale for the organisms and habitats being studied, as well as the hypotheses being tested. At the same time, as Underwood and Chapman (2013) acknowledge, constraints of money, time, and equipment must also be considered. This issue is particularly acute when seeking to use fully quantitative techniques on coral reefs, as the work is time consuming and divers are only able to spend limited time underwater. As a result, reef scientists often find it difficult to collect sufficient data to detect changes in coral assemblages over more than a few restricted areas of reef, unless the changesminvolved are very marked and obvious. However, shallow water ROVS and submersibles combined with AI address these limitations.

Fig. 9.3 An example of a Wentworth grain size chart used to classify sediments (from Jeffress Williams, Matthew A. Arsenault, Brian J. Buczkowski, Jane A. Reid, James G. Flocks, Mark A. Kulp, Shea Penland, and Chris J. Jenkins, USGS, Public Domain, source Wikimedia)

Concerning coral monitoring protocols, Leujak and Ormond (2007) found that video transects were the most time-efficient method for determining overall coral cover while photo-quadrats would likely be the most effective for estimating the cover of separate taxa, since their better resolution allows for quicker identification of genera and species. In order to secure statistically significant results (i.e., to obtain the desired precision), they found that it is better to reduce the accuracy with which substrate cover is estimated within each quadrat or transect and spend the time saved by recording additional quadrats or lengths of transect albeit with less accuracy. One can better obtain the precision required by determining as few as 5 points per quadrat (rather than say 100) while increasing proportionally the number of quadrats sampled (Leujak & Ormond, 2007).

2.5 Fish Abundances

Estimates of the abundances of fish in shallow waters are usually obtained by snorkellers and divers using one or other forms of Underwater Visual Census (UVC) (see detailed reviews by Harmelin-Vivien et al., 1985). UVC Methods have the advantage that they are non-destructive and do not involve any capture. Most usually the fish are counted by snorkellers or divers along 10–200 m long transects, or in some studies by a diver observing the fish present for a fixed period of time within 5 or 10 m diameter circular sample areas (circle counts) (Bohnsack & Bannerot, 1986). On transects either all species other than small (<5 cm) cryptic ones are counted or more commonly only the easily observed and ecologically significant families, such as butterflyfishes (Chaetodontidae), damselfishes (Pomacentridae), groupers (Serranidae), and snappers (Lutjanidae), parrotfishes (Scaridae), and jacks (Carangidae). Another method consists of circle counts, usually undertaken for 10 or 15 min per circle, allowing more time for smaller individuals and cryptic species to reveal themselves, thus providing a better estimate of species diversity. However, it will not estimate the patchiness of fish distribution.

Fish identification guides are numerous according to regions. In particular, FAO has developed species identification guides for all regions in the world (https://www.fao.org), as well as Fishbase which enables to identify fish to the lowest taxonomic level (Froese & Pauly, 2011).

Increasingly, video cameras to which bait is attached, referred to as Baited Remote Underwater Videos Systems (BRUVS) (Cappo et al., 2001), are also widely used to assess fish assemblages, especially to monitor the numbers of larger species (such as sharks) (e.g., Brooks et al., 2011) which are only rarely encountered on transects or of a wider range of species at depths where SCUBA divers cannot operate. Increasingly pairs of cameras are being deployed to form stereo BRUVS that allow the sizes of fish to be estimated (Johansson et al., 2008).

Presently conventional fish counting methods are being progressively replaced by fields of sensors, machine learning and deep learning (Zhang et al., 2024) using infrared optical sensors (Ferrero et al., 2014), acoustic-based methods (Jing et al., 2017), resistance fish counting system (Sheppard & Bednarski, 2023), machine learning (Fan & Lui, 2013), density map regression (Zhang et al., 2020b), and eDNA metabarcoding (Valentini et al., 2016). However, these methods face limitations such as computational complexity, invasiveness, and high costs according to Zhang et al. (2024), who propose new models that may be transferred to deeper domains.

3 Rapid Environmental Assessment and Semi-Quantitative Scales

3.1 Use of Semi-Quantitative Scales

To address the need for landscape-scale surveys, various terrestrial and shallow-water marine programmes have developed protocols for Rapid Environmental Assessment (REA) using several semi-quantitative methods. Generally, observers use one or more semi-subjective scales to assess the relative abundance and health of different lifeforms or habitats.

Crisp & Southwood (1958) introduced the ACFOR scale, which was later expanded into the SACFOR scale, where species or lifeforms are assessed as being superabundant (or dominant), abundant, common, frequent, occasional, or rare (Fig. 9.4). This scale has been extensively used in marine habitats mapping and monitoring of UK intertidal and sublittoral zones (Connor et al., 1997a, 1997b).

Even though individual estimates may not be as accurate as fully quantitative counts, statistical methods can nevertheless be applied to the data, enabling quantitative changes or differences to be detected with some confidence (Strong & Johnson, 2020).

Fig. 9.4 Table formalising the interpretation of the SACFOR (superabundant-abundant-common-frequent-occasional-rare) scale as used to estimate the abundance of marine organisms of different size during Marine Nature Conservation Review (MNCR) surveys of Britain and Ireland (from Connor et al. 1997a p. 40). (Open Government Licence)

Already in the 1950s, marine habitats with associated faunal communities were identified and assessed based on a bionomic manual developed by Peres and Picard (1955, 1958, 1964) for the temperate European waters, in particular the Atlantic Ocean, the North Sea, and the Mediterranean Sea. Progressively scientists and managers have adapted their sampling strategies to formalise the use of semi-quantitative assessments and identify thresholds of impacts and environmental change to apply to the management of marine resources.

Since 1990, the United Kingdom conservation agencies used semi-quantitative methods for surveying terrestrial habitats for Phase 1 Habitat Surveys in which large terrestrial regions were surveyed on a field-by-field basis (JNCC, 2010). The method provides a relatively rapid, standardised system for classifying and mapping habitats based on the relative abundance of dominant species. Only if sites are identified as being of potential conservation interest, more detailed (Phase 2) quantitative surveys are undertaken using fully objective methods. This approach has been adopted by conservation agencies in European and Overseas countries and extended to intertidal and shallow-water subtidal habitats to provide a detailed classification of the seashore and seabed benthic habitats with their associated species communities (Connor et al., 1997a, 1997b, 2005).

Those methods have been incorporated into a wider EUNIS Marine Habitat Classification developed by the European Environment Agency (Davies & Moss, 2004; Galparsoro et al., 2012) and most recently revised in 2022.[4] Internationally there are now at least seven similar marine habitat mapping schemes in common use. Although, as emphasised by Strong et al. (2020), they differ significantly concerning environmental and ecological scope, spatial scale, structure, and compatibility with other mapping techniques.

In undertaking Phase 1 surveys of intertidal and sub-tidal zones around the UK, Connor et al. (1997a) provided clear guidance on the range of abundances corresponding to each level of the abundance scale. Further the abundance levels are adjusted to allow for the

4 https://www.eea.europa.eu/data-and-maps/data/eunis-habitat-classification-1/eunis-marine-habitat-classification-review-2022

size of the organisms involved, since for example the actual densities of a small invertebrate judged as “common” would obviously be much greater than that of a “common” large vertebrate (see Fig. 9.10).

3.2 Use of Scales in Tropical Marine Environments

A similar approach has been adopted by some scientists and managers surveying or monitoring extensive reef systems or coastlines. Thus, extensive surveys of much of the Great Barrier Reef, initially for assessing impacts of the coral-eating crown-of-thorns starfish, have been carried out by the Australian Institute of Marine Science (AIMS) since the 1980s, using a diver holding a manta-board (flat board with two handles for the diver to record data on survey sheets) towed at slow speed behind a boat.

In the Red Sea, Roads and Ormond (1971) surveyed the reefs with manta-boards in search of crown-of-thorns using a SACFOR scale for animal abundances and a percentage scale for coral cover (Bradbury et al., 1987). During another survey, the whole 2500 km eastern coastline of the Red Sea has been assessed over 3 years (IUCN/UNEP, 1985). A 6-point semi-log abundance scale and a percentage substrate cover scale were used to assess the quality of reef communities and the intertidal habitat at approximately 1 km intervals. The results were used principally to select candidate sites for nature conservation and detect areas subject to local environmental impact. A comprehensive statistical analysis of the resulting data enabled to identify sub-habitats and communities and detect regional differences (Price et al., 1998). Thirty years later a partial repeat survey using the same methods was able to detect statistically significant regional changes in coral cover and reef health (Price et al., 2014a). Such rapid assessment methods have also proven effective for ground-truthing maps of coral reefs and associated shallow-water habitats using multi-spectral imagery captured by cameras on a small plane or drone (e.g., Wilson et al., 2003).

In the framework of the National Parks of Egypt Protectorates Development Programmes project with the Government of Egypt and the European Commission, a monitoring programme has been developed. Additionally training of the national team of environmental officers has been achieved during 2002–2003 along the Egyptian coasts of the Gulf of Aqaba and the offshores islands. The establishment of a network of permanent monitoring stations has been achieved to assess the impact of recreational submarine tourism (Tilot, 2003b; Tilot et al., 2008; Jeudy de Grissac, 1999, 2002).

At each station coral cover was determined using a video survey method at depths of 3, 7, and 16 m, and fish abundance by underwater visual census at depths of 3 and 10 m. At selected stations, video monitoring proceeded along 50 m transects with the camera held perpendicular to the reef at a distance of approximately 20–25 cm from the substratum, moving at a speed of 7 mm/sec. To analyse the video record, a minimum of 90 still frames were sampled within each 50 m transect, and the substratum underlying five regularly arranged points was assessed for each frame. Hard and soft corals were identified to genus and growth form, but not always to species level. The results showed that the mean total coral cover was higher at shallower depths, 3 m (41%), 7 m (32%), and 16 m (38%). The diversity of hard coral genera over all stations was highest at 3 m with a mean 10.3 genera recorded per 50 m transect. Analyses confirmed differences in coral assemblage related to depth and wave exposure. Fish abundances and assemblages also varied with depth. Transects subject to greater tourist use did not segregate from those subject to less tourist use.

In the same framework, an exploratory survey of five offshore islands in the Egyptian Red Sea was undertaken in December 1997 to evaluate the importance of natural resources in terms of biodiversity and serve as a baseline study for monitoring studies prior to the opening of recreational diving in 1998. The results showed that these islands held a relatively high biodiversity and abundance of coral cover. The total cover of living hard and soft corals on the leeward reefs of each island ranged from 42% for El Zabarghad to 72% for Abu El Kizan island influenced by a great variability in soft coral cover. Analyses showed that the depth gradient was not a significant parameter controlling these coral assemblages (Tilot-de Grissac et al., 2000).

Similarly, in the southern Red Sea, an extensive survey in the Eritrean Dahlak archipelago was conducted within the framework of the Eritrea Coastal, Marine and Island Biodiversity (ECMIB) project with GEF funding. It was achieved to ensure the conservation and sustainable use of the globally significant biodiversity of the coastal, marine, and island ecosystems (Tilot et al., 2012; Jeudy de Grissac, 2006). Capacity building and in situ training were done with the national team of environmental officers (Tilot, 2007). A network of marine protected areas with permanent monitoring stations has been established (Tilot, 2006c). Other surveys using the same methodology were conducted in Seven Brothers archipelago, Djibouti (Jeudy de Grissac, 2003), Socotra Island, Yemen with UNDP/GEF (Tilot, 2002a), Hawar Islands, Bahrain (Jeudy de Grissac, 2004), Calamians Islands, Philippines (Tilot, 2003a), in the Daymaniyat Islands, Sultanate of Oman within a IUCN/RIOWA project (Tilot, 2016; Jeudy de Grissac, 2016), and in Alboran Sea (Spain Morocco, Algeria) and along the Lebanese coast up to 1000 m depth (Jeudy de Grissac, 2012a, 2012b).

With the NASA astrobiology team, a similar survey was operated in a high-altitude lake in Chile, to test a probe that would be sent to Titan by NASA (Cabrol et al., 2015; Parro et al., 2019). The survey led, in particular, to the discovery of the first freshwater polychaete tubeworm community associated to methane seeps (Tilot et al., 2019).

3.3 Indicator Species

Rapid Environmental assessment frequently uses indicator species or taxa, organisms selected as both relatively easy to monitor and likely indicative of the ecological status of the environment and habitats concerned (Whitfield & Elliot, 2002; Link & Browman, 2017). Reefcheck, for example, uses indicator species in some coral reef monitoring programmes (Hodgson et al., 2006).

Concerning coral reefs, butterflyfishes (FC) from the Chaetodontidae family are often selected as bioindicators. Indeed, they are relatively specialised concerning their trophic behaviour, e.g., nocturnal carnivore, diurnal carnivore, corallivore obligate (e.g., Chaetodon trifascatus), facultative corallivore (e.g., Chaetodon lunula), planktivorous, omnivore, and herbivore. Because of this trophic specialisation, on a healthy reef, butterflyfishes can serve as bioindicators.

For example, the video-monitoring survey along the Egyptian coasts of the Gulf of Aqaba (Tilot, 2002b, Table 7) has shown that the most notable differences of the state of the reefs were displayed by the lower abundance of butterflyfishes (Chaetodontidae) at 10 m depth when comparing 2002/2003 to 1996.

Some groups referred to as “Sentinel species” or “Sentinels of the Seabed (SOS) species” may also provide useful tools for providing insight into the ecological functioning of ecosystems or habitat status under the pressure of natural or anthropogenic disturbance. These Sentinel species are both typical of an environment and sensitive to environmental change (Serrano et al., 2022). Their selection can be undertaken formally by means of a two-stage process. In the first stage a “typical species set” is identified using intra-habitat similarity and frequency measures generated under reference or unimpacted conditions. The “sentinel species set” is then generated by selecting species that, by comparison with impacted sites, appear to be most sensitive to change.

The monitoring of “sentinel species” or “sentinel communities” should be adapted to the pelagic and abyssal domains and its different water layers where trophic processes and migrations occur. Using these sentinel species would also enable cost-effective monitoring of the state of the water column during deep-sea mining operations. Other indicator species in the water column are discussed further in Sect. 6, “Apex Migratory Predators and the Bioluminescence Producing Species”.

3.4 Metrics for Biodiversity and Environmental Sensitivity

Concerning measures of biodiversity, at both within-habitat (alpha-diversity) and within-region (beta-diversity) scales, it has become a priority to develop metrics that can be adapted to assess and monitor ecosystems where full species and habitat lists are unavailable, such as for many parts of the high seas, including the CCZ (Gray, 1997; Price, 1999; Izsak & Price, 2001). Also required are process-oriented metrics that account for ecosystem dynamics across temporal and spatial scales (Steneck, 2001; Price et al., 2007). Key functional groups, ecological roles, and species interactions (Hughes et al., 2005) can be meaningfully assessed by measures that are not too sensitive to sample size, such as average taxonomic distinctiveness (i.e., a measure of the mean taxonomic distance between species within a sample or study area) and variation in taxonomic distinctiveness (i.e., a measure of the variance in taxonomic distance between species) (Clarke & Warwick, 2001).

Complementing taxonomic distinctiveness, the concept of taxonomic similarity (i.e., the mean taxonomic distance between species from different samples or sites), can be used to compare across areas at different scales, since again this metric is not too sensitive to sampling intensity (Izsak & Price, 2001; Price, 2002; Price & Izsak, 2005).

Furthermore, diversity indices, such as environmental sensitivity or vulnerability indices, can be used to prioritise different conservation areas. These indices are often coupled with a Geographic Information System (GIS) to indicate biological sensitivity to different classes of impact and the likelihood of occurrence of these impacts (Cogan et al., 2009). Environmental sensitivity index (ESI) maps have been widely applied to the marine environment, initially for risk management of spills in the oil industry (Buckley, 1982; Tortell, 1992; Jensen et al., 1993). ESI maps, when complemented by maps representing socioeconomic data, can sharpen collaboration between managers and stakeholders to identify vulnerable locations, establish protection priorities, and identify strategies to minimise undesirable consequences. The value of habitats as surrogates for ecosystem services needs further consideration, as there can be important co-benefits or, alternatively, opportunity costs associated either with their conservation or their loss (Fraschetti, 2012; Fourchault et al., 2024).

In the case of the Gulf of Aqaba monitoring Programme (Tilot, 2003b), “vulnerability indexes/need for protection indexes” have been calculated on the basis of % of morphological characteristics of the coral colonies (branched, tabular), the sand cover, slope characteristics, hard coral cover, coral diversity, coral Shannon Wiener H′, Fish abundance, fish diversity, and fish Shannon Wiener H′. Table 9.1 shows that the stations most vulnerable are those with the highest coral cover, in particular branching, high coral diversity, high fish abundance and diversity, along shallow slopes and thus have a higher need for protection (Tilot, 2002b; Tilot et al., 2008).

Multi-metric management indices, such as included in the Ocean Health index (Halpern, 2020) are currently applied for different purposes. Some include indicators of physical stress, water quality, the presence of biological invaders, and over-fishing. Conservation classes can be assigned to reef communities based on their growth forms and their life strategies (stress tolerances, adaptation to disturbances, competitive dominance, etc.) making it possible to assess the complexity of habitats and associated epibenthic communities. The objective is to assign a biodiversity value and a bioconstruction value (measuring replacement time for each colony) and to evaluate the structure and dynamics of populations (recruitment and mortality).

Table 9.1 “Vulnerability index” and “need for protection index” of the coral monitoring stations of the Gulf of Aqaba (adapted from Tilot, 2002; Tilot et al., 2008)

Sustainability indices, such as the Human Well-being and Sustainable Livelihood (HWSL)* indicator, have been proposed to determine the effectiveness of policy measures in resource management (Dahl, 2012a; Sterling et al., 2020). Sets of indicators that capture both ecological and social-cultural factors, and the feedbacks between them, can underpin cross-scale linkages that help bridge local and global scale initiatives to increase the resilience of both humans and ecosystems (Sterling et al., 2020).

UNDP has produced the Human Development Index (http://hdr.undp.org/en/) to focus on this in a collective way at the national level, but this hides significant disparities within countries. What is lacking is a way to operationalise the development concepts that achieve well-being at the individual level (Dahl, 2012a).

The recognition of the need to look beyond GDP (Stiglitz, 2009)) has been acknowledged by governments at the United Nations Conference on Sustainable Development (Rio + 20) (Dahl, 2012b). Ideally, a well-being indicator would integrate the economic, social, and environmental dimensions, enabling every human being to fulfil their potential in life both by cultivating individual qualities, personality and capacities and by contributing to the advancement of society (Baha’i International Community, 2010). Bhutan was the first country to assess the purpose of development through Gross National Happiness (Ura et al. 2012, http://www.gnhc.gov.bt/). The OECD, the European Environment Agency, and several countries and international organisations have developed international standards for measures of well-being and happiness. Thus, placing focus on the individual makes sustainable development relevant, triggering a positive impact on their relationships and local communities and leading to improvements on a larger sphere (Dahl, 2012b).

4 Environmental Surveys, Monitoring, and Rapid Environmental Assessments in the Deep Ocean

4.1 Platforms for Deep Sea Benthic Surveys

The deep sea has been first explored by bottom trawls and grab samples (Belyaev, 1989), and underwater cameras (Heezen & Hollister, 1971; Lemche et al., 1976). Presently new imaging technologies and the use of submersibles enable a more holistic analysis of undisturbed deep sea communities and a greater understanding of the structure and processes of the marine benthic and pelagic ecosystems (Solan et al., 2003).

The Remotely Operated Vehicles (ROVs), carrying still and/or video cameras, are tethered underwater vehicles controlled from the surface which enable video/photo monitoring and sample collecting. ROVs are one of the best-known tools for the exploring and monitoring of deep-sea habitats. Originally developed for industrial applications such as oil and gas exploration, ROVs have been adapted for scientific purposes and proven capable of recording high-resolution still and video imagery as well as collecting occasional specimens and other environmental data (Macreadie et al., 2018; McLean et al., 2020). In this way, the most advanced ROVs have demonstrated the ability to collect extensive data on species diversity and abundance, providing valuable insights for habitat monitoring and conservation efforts. However, despite being most effective for surveying only limited areas, one of the biggest challenges in employing ROVs is the huge amount of data that can be generated, particularly as video footage often requires considerable, time-consuming post-deployment processing by one or more skilled taxonomists (Macreadie et al., 2018). Because of the limited length of their umbilical cable at great depths, their heavily reinforced hull, and restricted speed and manoeuvrability, ROVs are best used for targeted surveys of restricted sites, rather than for landscape-scale mapping, for which autonomous underwater vehicles (AUVs) may be more effective (McLean et al., 2020).

ROVs are also used in midwater such as with the Monterey Bay Aquarium Research Institute where they have been co-evolving for 30 years. Robison et al. (2017) developed a time-series programme based on quantitative video transects with the ROV flying at a constant depth and speed (55 cm s–1) for 10 min. The first transect is run at a depth of 50 m, the second at 100 m, and subsequently at 100 m intervals down to 1000 m. The main video camera is set at its widest angle, and the distance travelled during each transect is measured with an acoustic current meter. Lighting is configured to fully illuminate the entire viewing angle of the lens, up to 3 m in front of the vehicle.

Operating ROVs at depths of 1000–6000 m also requires significant financial and logistical resources. Industrial-grade ROVs designed for heavy-duty tasks can cost hundreds of thousands of dollars and require significant technical expertise and funding to operate (McLean et al., 2020). In addition, ROV deployments require surface vessels and experienced personnel, increasing the cost of a survey campaign (Elvander & Hawkes, 2012). Nevertheless, ROV technology is advancing rapidly, nd smaller, more cost-effective models are being developed which operate at greater speeds and depths (Elvander & Hawkes, 2012).

Autonomous underwater vehicles (AUVs) differ from ROVs in that they are not reliant on an umbilical line/tether but are deployed to operate autonomously away from the parent vessel. They have the ability to operate over long periods of time without direct human control and can cover large areas of ocean without the limitations of tethers. AUVs are task controllers that integrate artificial intelligence and other advanced computing technologies. Hence AUVs can explore bathypelagic and abyssopelagic seabed areas more effectively than ROVs, making them potentially more useful for habitat mapping and environmental monitoring on a larger scale (Huvenne et al., 2018; McPhail, 2009). Nevertheless, they have a restricted ability to respond to biological cues.

AUVs can use path planning and obstacle avoidance techniques, as described by Zhang et al. (2021), to navigate complex underwater terrain while minimising energy consumption. While these techniques improve the efficiency of data collection, they do not address the limitations of video resolution and species recognition (Ridao et al., 2015). Horton et al. (2021), for instance, have stressed that while the newer AUV models show improvements in image quality, species identification is still limited to higher taxonomic levels due to poor image resolution.

Ongoing efforts are underway to develop a new generation of “Intervention AUVs” (I-AUVs), as described by Ridao et al. (2015). These vehicles are designed for intervention tasks such as sampling and manipulation, but although some have shown promise in controlled environments, they are still at an early stage of development. While these advancements improve the operational efficiency of AUVs, they do not necessarily improve the quality of the data collected. In environments with complex terrain or poor visibility, AUVs still struggle to capture high-resolution images. Furthermore, the computational complexity of real-time path planning algorithms may limit their application in highly dynamic environments that require rapid decision-making (Zhang et al., 2021).

Presently, swarm robotics have great potential as cooperative navigation can improve agents’ positioning and navigation performance through information sharing among AUVs (Cai et al., 2023).

Drop cameras and seabed landers are known to capture high-resolution images of the seabed substrate and biota, allowing for much better identification of habitats and species (de Mendoça & Metaxes, 2021). Drop cameras, lowered by a tether from a vessel, and seabed landers, released without a tether but retrieved when a flotation device is triggered, are being increasingly used for both deep-reef and deep-sea studies. Some landers, designed for use at extreme depth, where pressure and the absence of light limit the durability of the equipment and image quality, are able to withstand pressures of over 1100 atmospheres and have been deployed to depths of up to 11,000 meters (Hardy et al., 2013). Stoner et al. (2008) have outlined the necessity to adapt the survey gear and sampling method according to the knowledge of the behaviour of deep water fish to study.

Towed camera systems or benthic sledges have become the most important tools for deep-sea ecological studies, providing visual access to benthic habitats that are otherwise difficult to monitor. These systems consist of digital cameras attached to a sled or frame that is towed behind a vessel, usually via a coaxial or fibre-optic cable permitting real-time viewing and image recording of the seafloor (Foell and Pawson, 1985; Fornari, 2003; Kelley et al., 2016). Towed cameras, designed to capture high-resolution images and, in some cases, video, enable researchers to view benthic habitats in regions where the benthic substrate is relatively flat, but they are less practicable on reefs or steep slopes. The depth of deployment, resolution, image-analysis capabilities, and environmental adaptability of different systems vary significantly, thus influencing image quality and the scientific insights to be gained.

Purser et al. (2019) used a towed video system designed to capture high-resolution images at depths of more than 1000 m in order to investigate the distribution and abundance of benthic organisms in hydrothermal vent environments. The resolution, although not specified, was sufficient to identify large benthic organisms and compare the composition of fauna at different sites, but not to capture smaller organisms and finer ecological details. Also, when comparing trawl and video camera surveys, visual surveys can reveal rather different “pictures” of fish densities (McIntyre et al., 2015). Adapted towed camera systems and sampling strategies have been designed to upgrade the sampling efficiency (McIntyre et al., 2015).

A combination of the different types of platforms may be employed to operate most effectively depending on the topic of research, the geomorphology and ecology of the seafloor, the characteristics of the water column, as well as the extent of the area to be studied. However, the use of multiple platforms obviously requires a large vessel and considerable resources.

Recent advances in machine learning (AI) enable fast, sophisticated analysis of visual data but have still limited success in the ocean due to lack of data standardisation, insufficient formatting, and the need for large, labelled datasets. AI has proven highly effective in processing photographic and video data in the deep sea by automating taxonomic and habitat identification, as well as assessing state of health, a process that continues to evolve with the acquisition of new identifications (Lopez-Vazquez et al., 2023).

The National Geographic Society's Exploration Technology (NGSET) Lab deployed between 2010 and 2020 its autonomous benthic lander platform (the Deep Sea Camera System, DSCS) to collect video data from locations in all ocean basins (Giddens et al., 2020). A total of 594 deployments collected videos at depths ranging from 28 m to 10,641 m in numerous habitats including trenches, abyssal plains, seamounts, arctic, shelves, straits, and canyons. The videos from these deployments have subsequently been ingested into an AI-powered, cloud-based collaborative analysis platform, where they are annotated by experts at the University of Hawaii.

The Monterey Bay Aquarium Research Institute (MBARI) has demonstrated that machine learning can also accelerate behavioural observations (https://www.mbari.org/wp-content/uploads/Kaplan_Kira.pdf). Egbert et al. (2020) have been developing software that can be used to annotate underwater video recordings and can, by machine learning, identify species previously encountered. Their website was initialised with 50 h of high-resolution underwater videos from the Monterey Bay Aquarium Research Institute (MBARI). Biology students contributed more than 30,000 annotations, of which approximately 10% were verified by experts. Recently, MBARI built an open-source image database, "FathomNet", that standardises and aggregates identifications and associated information from experts.

This allows future contributions from distributed data sources, accelerating video data processing. It reduces annotation effort and enables automated tracking of underwater concepts when integrated with robotic vehicles. FathomNet can also be used to train and deploy models on other institutional videos as demonstrated with the NGSET Lab data (Katija et al., 2022).

5 Image and Video Sampling and Analysis

5.1 Cost-Effectiveness of Photo and Video Sampling in the Deep Sea

On both coral reefs and the deep seabed, questions arise: how many sites need to be inspected, how many images or hours of video studied and, in particular, should as many images or sequences as possible be studied in as much detail as practical, or is it better to undertake a more rapid assessment of a greater amount of imagery? Given the variability of the seabed and in the deep ocean the scarcity of large or readily identifiable organisms, much greater statistical precision may be achieved by using various measures to analyse much larger numbers of images or sequences, rather than by undertaking time-consuming highly detailed analysis of only a limited amount of imagery. These measures may include the use of semi-quantitative measures as discussed further below.

Among the largest challenges associated with camera-based surveys of the seabed is the amount of time required to process the large quantity of digital imagery collected in addition to other data collected from multi-parametric platforms to form "holistic monitoring" (see further, 7). Analysis of video data is very labour intensive and, depending on the level of taxonomic analysis attempted, may require considerable and multiple expertise. Further, the resolution of the video footage is often inadequate (Wynn et al., 2014), given that high-resolution imagery is essential for documenting subtle taxon-specific features, behaviours of megafauna, and ecological processes. Additionally, in some habitats, sightings of megafauna may be few and far between.

A number of annotation tools have been developed to manage and analyse visual data, resulting in many software solutions that can be deployed on computers onsite during field expeditions or on the World Wide Web (Gomes-Pereira et al., 2016). However, the limited availability of experts and the prohibitive costs to annotate and store footage have induced the development and deployment of novel methods for automated annotation of marine visual data using artificial intelligence and data science tools for ocean ecology (Katija et al., 2022).

The processing times and methods for video and image data vary between the platforms and cameras frequently influence the choice of equipment, given the scope and timeline of projects. de Mendoça and Metaxes (2021) described the use of ImageJ software to analyse drop camera images, focusing on manual quality control to ensure that images are representative of benthic communities. They assessed images judged unsuitable due to poor lighting or sediment clouds, and these were either excluded or cropped, enabling more targeted species identification. However, these images can also inform on trait characteristics and bioluminescence of pelagic and benthic species (Tilot et al., 2024b). This approach to quality control can significantly increase processing times, especially when high-resolution images are used, but it may improve data accuracy. This poses the question of whether the time and cost of expensive of platforms and vessels are justified on the terms of the limited quality of the considerable amounts of data they can generate.

Where still imagery is obtained, for example, using a drop camera or bottom lander, once preliminary estimates of standard error are possible, one can undertake a statistical power analysis to determine (a) what is the number of replicates required to distinguish between different sites or areas?, and (b) whether it is preferable to assess individual images with greater accuracy, or by using less accurate estimates to increase the number of images analysed? Details of the issues and calculations can be found in Ecological Census Techniques edited by Sutherland (2006); although to determine the best use of time, it is also necessary to estimate the time required both to acquire additional images and to assess them to different levels of accuracy. In the same perspective, more recently, Swanborn et al. (2002) have reviewed the application of seascape ecology, linked to landscape ecology and habitat mapping, to the deep sea and extracted the pattern-oriented concepts, multiple-scale tools and techniques relevant to spatial management of Barents Sea, as case study.

From a statistical perspective, it would be desirable to space out randomly all the sampling stations over the region being monitored (which may be thousands of km2 of seabed); in practice, logistical constraints often make it more feasible to take sets of replicate samples in one area prior to moving to another for additional replicates. This limitation arises because significant time may be required to move beyond short distances (by vessel or by vehicle) making continuous sampling impractical. Consequently, this results in what is termed “cluster sampling”. While the term might suggest an undesirable approach, power analysis can be applied to an optimum number of samples to be taken from each cluster and the distance between clusters to achieve the highest precision (Greenwood & Robinson, 2006).

Because the abundances of benthic biota typically vary at a range of spatial scales, the size of the sampling unit selected to sample a habitat or population is very important for distinguishing patterns of distribution with any confidence (Underwood & Chapman, 2013). Where single sites or areas on a scale of tens of km are to be surveyed, or the biodiversity or ecology of limited areas investigated, much smaller numbers of images or videos may be adequate, or certainly preferred if the resolution achieved is greater. Conversely when there is a need for landscape monitoring of the seabed in where very large sample sizes are required, as for polymetallic nodule fields or climate change impacts, sets of large number of sample images are necessary to achieve reasonable precision.

Current platforms are able to supply two forms of visual data collected at the deep seabed. Single images comparable to photo-quadrats are captured by seabed landers and may be of high resolution; although if only modest numbers of images can be acquired, the extent to which the spots sampled are representative of the wider region is impossible to determine with any certainty. In contrast benthic sledges and drones may complete very long excursions and generate either thousands of still images or hundreds of hours of video recordings that are not only difficult but immensely time-consuming to interpret.

The sampling intervals should also be considered according to the biota and substrate to be assessed. During long deployments such as those performed in the water column and the abyssal and hadal zones, sampling intervals are variable, e.g., 20 min sequences to monitor deep mesopelagic boundary communities in the Monterey submarine canyon (Leitner et al., 2021) or more than 30 min sequences on abyssal hills versus 2 or 5 min sequences in coral reefs. Setting sampling intervals may be the optimal approach if too many video sequences have been recorded for all of them to be analysed in full, although AI can increasingly help in identification and quantitative analysis processes.

Once interpretation of adequate video or still imagery has been undertaken, a variety of standard multi-variate statistical approaches are available to detect spatial or temporal trends or to detect more complex patterns. A study by D’Onghia et al. (2011) deployed a towed camera to study benthic fauna and employed multiple correspondence analysis (MCA) to correlate the distribution of species with substrate types. Kelley et al. (2016) used multivariate statistical techniques, including non-metric multidimensional scaling (NMDS) and analysis of similarity (ANOSIM), to compare fauna on natural and artificial substrates. de Mendoça and Metaxes (2021) used one-way analysis of variance (ANOVA) to compare the abundances of the most abundant taxa, and NMDS and permutational multivariate analysis of variance (PERMANOVA) to explore patterns in the benthic assemblages across sampling designs. However, such analyses will be more revealing and more likely to reveal significant trends if some form of power analysis is used to assess the optimum sample size and number and the accuracy of analysis required to achieve the desired precision.

5.2 Analysis of Deep Sea Imagery in the CCZ in View of DSM

In the context of assessing deep sea (bathyal, abyssal) mineral resources of commercial interest (polymetallic nodules, hydrothermal sulphides, cobalt-rich crusts) and associated megafauna (visible on imagery), specific Deep Sea Semi-Quantitative (DSSQ) scales and annotated lifeform (taxa) photographic guides have been developed. These tools facilitate the identification of different mineral and sediment “facies” as well as megafaunal assemblages, particularly for polymetallic nodules.

Concerning the polymetallic nodule fields, Specific Deep Sea Semi-Quantitative (DSSQ) scales and an annotated lifeform (taxa) photographic guide reviewed by world taxonomists have been developed to identify different mineral and sediment “nodule-facies” and megafaunal organisms as well as assess abundances and topographic and sedimentary features (Tilot, 2006a). These were constituted from samples, videos, and photos collected by IFREMER with a wide range of platforms which were often combined (the towed benthic sledge “Troika”, free-fall grabs with photo camera, the towed “R.A.I.E.” (the remotely controlled unmanned submersible “Épaulard”), the manned submersible “Nautile” (Tilot, 2006b; Tilot et al., 2018), and now the regularly upgraded ROV Victor 6000 (https://www.ifremer.fr/fr/flotte-oceanographique-francaise/decouvrez-les-navires-de-la-flotte-oceanographique-francaise/victor-6000).

The DSSQ scale for polymetallic nodule fields was established by an expert team of sedimentologists and geologists at IFREMER/AFERNOD based on the parameters of their geological environment in particular, the topography, erosion by deep ocean currents, and the model of regional deposition. It has led to the differentiation of nodule fields and the recognition of “nodule-facies” (Hein and Voisset, 1978) based on a photographic study of the seabed combined with samples and morphological and geochemical measurements of different types of nodule-facies. The current classification defines five principal nodule-facies (Table 9.2).

Table 9.2 Summary of the five types of nodule-facies based on visual observations and photographs of the seabed (Hoffert & Saget, 2004), adapted from Tilot, 2019© Ifremer

For example, nodule-facies C consists of large nodules (6–15 cm diameter or sometimes larger) that are deeply embedded in surface sediments, if not completely buried (60–100%). These nodules have a hummocky and heterogeneous (sometimes granular) surface. They are typically elliptical in shape. A visible equatorial thickening can be observed due to the sediment distribution partially covering the nodule. They are generally set deep within the surface sediments, and fracture fissures are often present. Attached organisms are common. The superficial texture ranges from smooth to rough, with a dominant botryoidal structure.

The nodules are non-coherent. The concentration of nodules decreases (2–10 kg/m2) as nodule size increases. They cover only 15–20% of the ocean floor with a maximum abundance of 8 kg/m2. Higher values in some samples may result from complete burial of nodules. A slope factor > 15% has been studied on nodule-facies C 30%. These nodules are always present in the southern part of the AFERNOD zone but coexist with nodule-facies B further north. The underlying silts are less clayey and richer in radiolarians. They have a high water content with very weak cohesion in the first few centimetres, forming a semi-liquid layer. These nodules contain high levels of manganese (30%, with a ratio to iron of 6), nickel (1.4%), and copper (1.2%). The concentric layers of hydroxides of manganese are well crystallised (Tilot, 2006a).

Nodule-facies C+ has slightly smaller nodules (7.5 cm diameter), less sunken into the silt (30–60%), homogeneous in size, with a hummocky surface, a predominantly mottleddendritic internal structure, and found at concentrations of 8–20 kg/m2. The superficial texture is smooth to rough, and like nodule-facies C, they are non-coherent. Nodule-facies C is the one (the other is facies BP) targeted for commercial deep-sea mining due to its valuable mineral contents.

Taxon identification can be achieved by world identification databases progressively growing, in reference to the World Register of Marine Species (WoRMS)[5] (Boxshall et al., 2014) and to the Ocean Biodiversity Information System (OBIS) data standard formats for image-based marine biology (De Pooter et al., 2017; Giddens et al., 2020).

Nevertheless, a taxon identification guide, annotated with trait characteristics per taxa (when possible), was developed specifically, based on the analysis of more than 200,000 photographs (each covering approximately 12 m2) and 22 h of film footage. All identifications and annotations on trait characteristics have been subsequently reviewed by world taxonomists (Tilot, 2006a). Selected photographs of identified megafauna and nodule-facies are displayed in Fig. 9.5. Additionally, an annotated photographic atlas of echinoderms from the Clarion-Clipperton Fracture Zone was published in the UNESCO Intergovernmental Oceanographic Commission Technical Series (Tilot, 2006b).

Fig. 9.5 Images of megafauna in the nodule ecosystem at three French study sites in the CCZ taken by the platforms Épaulard, RAIE, and TROIKA, and from the manned submersible Nautile.

From the beginning of deep-sea ecology, the importance of megafauna in the functioning of deep ocean environments was evident (Rex, 1981; Smith & Hamilton, 1983). The first quantitative analyses of the epibenthos and records of megafaunal animal behaviour were based on photographic and video data taken by towed or fixed devices and manned submersibles starting in the 1960s and 1970s (Owen & Sanders Hessler, 1967; Rowe, 1971; Grassle et al., 1975; Lemche et al., 1976; Cohen & Pawson, 1977). These surveys progressively expanded with the development of diverse platforms (Ohta, 1985; Laubier et al., 1985; Foell, 1988; Foell et al., 1992; Pawson, 1988a, 1988b; Tilot et al., 1988; Foell & Pawson, 1989; Kaufmann et al., 1989; Wheatcroft et al., 1989; Bluhm, 1991; Thiel et al., 1991; Sharma and Rao, 1992; Christiansen & Thiel, 1992; Smith et al., 1992; Christiansen, 1993; Bluhm, 1994; Bluhm & Thiel, 1996; Lauermann et al., 1996; Piepenburg & Schmid, 1997; Hughes & Atkinson, 1997; Fukushima & Imajima, 1997; Kaufmann & Smith Jr, 1997; Kotlinski & Tkatchenko, 1997; Matsui et al., 1997). These works have been referenced in comparative analyses in the three volumes published in the UNESCO Intergovernmental Oceanographic Commission Technical Series (Tilot, 2006a, 2006b, 2010) and in Tilot et al. (2018).

In the case of polymetallic nodule fields, megafauna can be used as an indicator of recolonisation on the seabed after the impact of deep-sea mining (Tilot, 1988; Bluhm, 1997) or of the variation in flux of particulate organic carbon (Smith et al., 1997). The megafauna is also one of the principal agents of bioturbation of the deep sea benthos (Mauviel, 1982; Mauviel & Sibuet, 1985; Levin et al., 1986) and so can influence many other biological and geochemical components of the ocean depths (Sharma & Rao, 1992). Because of its scattered distribution, the study of the megafauna requires the sampling of large areas (Rice et al., 1982).

Concerning the Clarion Clipperton Fracture Zone, the first extensive study of the benthic megafauna was conducted on five sites (French NORIA, NIXO 45, NIXO 41, American

5 https://www.marinespecies.org/

ECHO1, the consortium site IOM BIE (Russia, Bulgaria, Cuba, Poland, Czech Republic, Slovakia). Comparisons were also made with the fauna from the Peruvian Basin (southern Pacific Ocean), using data from the German DISCOL cruise (Tilot, 1989), to identify potential morphospecies or taxa common to both regions (tropical north-eastern and south-eastern Pacific). The analysis was based on more than 200,000 photographs and 55 hours of video footage collected since 1975, by various submarine devices, the towed "RAIE", the sledge Troika, the camera coupled free sampler "ED1", the autonomous unmanned "Épaulard," the manned submersible "the Nautile" for the French areas, the American "Deep Tow Instrumentation System" for ECHO I site and for the IOM BIE site, the Russian towed cameras MIR-1 and NEPTUN, and also box cores with photo cameras, the "Ocean Floor Observatory System (OFOS) in the German impact study sites of DISCOL in the Peru basin, south Pacific (Tilot, 1989; Tilot, 2006a, 2006b; Tilot, 2010; Tilot et al., 2018).

In this extensive survey, video and photographic transects were analysed across various nodule-facies with percentage cover recorded as follows: A30%, C5%, C10%, C15%, C20%, C30%, C40%, B40%, B50%, BP35%, BP50%, O old sediments, O recent sediments. The size, tracks, behaviour, and habitats permitted the compilation of an exhaustive database.

In a pilot study at NIXO 45 (Fig. 9.6), a random selection of 48,100 photographs, covering an area of about 76,000 m2, collected by the "Épaulard" and the R.A.I.E. have been analysed by units of 200 photographs. The surface areas photographed by the "Épaulard" and "R.A.I.E" were similar. Data was analysed using a program developed by Ifremer for studying the spatial distribution of megafauna photographed in situ (Sibuet, 1987). This computer program adds by successive increments the surface area of each photograph calculated from the elevation of the camera. Results displayed a total of 159 taxa.

Fig. 9.6 Pilot Site NIXO 45 where REA and management indices have been applied. GIS, Bathymetry, currents, and the four main facies represented. The areas with slope bigger than 15% are highlighted in green. Bottom current directions derived from the analysis of imagery of the seabed, in particular of the direction of suspension feeders (in Tilot, 2019)

The results of our analysis revealed significant taxonomic richness among several phyla, ranked by the number of identified taxa, such as (in order of importance) cnidarians (59 taxa), echinoderms (46 taxa), sponges (38 taxa), and chordates (27 taxa). Preferential habitats have been identified and summarised in Table 9.3.

Table 9.3 Faunal communities associated to preferential habitats identified during the extensive survey of megafaunal in the CCZ (adapted from Tilot, 2006a, Tilot et al., 2018)

Nodule-facies O on recent sediments characterised by an abundance of mobile fauna, mainly detritus feeders and carnivores, mainly isopods Munnopsidae, asteroids Porcellenasteridae, Ophidioid, and Ipnopid fish. Suspension feeders on this facies are sedentary polychaete worms responsible for a particular form of disturbance known in the literature as "witches rings" (Heezen & Hollister, 1971). Taxa exclusive to this facies are sedentary polychaetes Cirratulidae, holothurians Psychropotes longicauda, Psychropotidae and siphonophores Physonectes sp.
Nodule-facies O on ancient sediments is characterised by a majority of suspension feeders, octocoralliarids Primnoidae, Isididae, Hyocrinidae, and actinids Hormathiidae, Actinoscyphiidae. Detritus feeders, asteroids mainly Pterasteridae, holothurians, mostly, Synallactes aenigma and Benthodytes lingua, and peracarids, Cumacea. Taxa unique to this facies are suspension feeders Demospongia Cladorhizidae and sedentary polychaetes with typical round mounds with a circle of holes. Exclusive detritus are asteroids Hymenaster violaceus and holothurians Benthodytes lingua, and exclusive carnivores such as gastropods Pterotracheidae and Liparid fish.
Nodule-facies C+ 2–5% is characterised by fixed fauna, predominantly suspension feeders, alveolate hexactinellid sponges, unique to this facies, octocoralliarids Isididae, Umbellulidae, and corallimorpharids

Sideractiidae. The most abundant detritus feeders are Munnopsid isopods and holothurians Peniagone gracilis. Ophidioid fish are the most abundant carnivores. Vesicomyidae bivalves are exclusive to this facies
Nodule-facies C+ 10% with mostly mobile detritus feeders, sipunculids Nephasoma elisae, echinoids Plesiodiadema globulosum and holothurians Mesothuria murrayi, Paelopatides sp., and Pannychia moseleyi. The most abundant suspension feeders are hexactinellid sponges Hyalonematidae and octocoralliarids Primnoidae. Carnivores are medusas Trachynemidae. Decapods Plesiopenaeus sp. are exclusive to this facies, as demospongian sponges Phakellia sp.
Nodule-facies C+ 15% with essentially mobile suspension-feeders with actinids Liponema and Actinoscyphia sp. and polychaete Sabellidae, detritus feeders with echinoids Aeropsidae with a characteristic sinuous trail and holothurians Peniagone vitrea. Abundant carnivores are archaeo-gastropods and siphonophores Rhodaliidae. Taxa exclusive are carnivores as Physonectes siphonophores, neogastropod Turridae, polychaetes Polynoidae or Aphroditidae and fish Coryphaenoides yaquinae. Suspension feeders unique to the facies are sponges Caulophacidae
Nodule-facies C+ 20% mainly mobile organisms: Suspension feeders such as sponges Poecillastra sp. and free crinoids Antedonidae. Detritus feeders are peracarids Cumacea and holothurians Meseres macdonaldi. Unique taxa are sponges Hyalonema sp., Chiroteuthid cephalopods, and Galatheids
Nodule-facies C+ 30% with an abundant mobile fauna, largely carnivorous, cephalopods Benthesicymus sp. (unique to this facies), and medusas Nausithoidae. Detritus feeders are the swimming holothurians Enypniastes eximia
Nodule-facies C+ 20–40% on slopes > 15° with mainly fixed suspension feeders Hexactinellid sponges Rossellidae, Euretidae and Demospongia Cladorhizidae and sedentary polychaetes. Detritus feeders are mainly echinoderms Brisingidae and holothurians Benthodytes sp. carnivores with decapods Nematocarcinus sp., peracarids Tanaidacea and Bythitidae fish Typhlonus sp
Nodule-facies C+ 40% with mainly carnivorous polychaetes Hesionidae and Aphroditidae and swimming polychaetes. Detritus feeders, such as holothurians Peniagone intermedia and suspension feeders, sponges Euplectella sp. and ophiuroids Ophiomusium sp. the unique taxa are holothurians Orphnurgus sp., Amperima naresi, and ascidians
Nodule-facies B 40% with mainly mobile detritus feeders, holothurians Psychronaetes hanseni and Benthodytes typica and asteroids Hymenaster sp. suspension feeders are the antipatharids Bathypates patula and Bathypates lyra, Brisingidae Freyella sp. unique to this facies are peracarid amphipods and holothurians Elpidiidae, Deimatidae Deima validum and octocoralliarid Umbellulidae
Nodule-facies B 50% mainly suspension feeder actinids Sincyonis tuberculata and carnivore swimming aphroditid polychaetes. Are unique, antipatharid Schizopathes crassa
Nodule-facies BP 35% mostly suspension feeders sponges Poecillastra sp., actinids Bolocera sp. and Actinoscyphia, ophiuroids Ophiomusium armatum. A two-horned ring-shaped Hexactinellid sponge is exclusive to this facies
Nodule-facies BP 50% mostly holothurians Synallactes aenigma, Synallactes profundi, Peniagone leanderas, Benthodytes sp. and polynoid polychaetes

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In terms of taxonomic richness, sessile suspension feeders (84 sessile taxa and 10 mobile taxa) dominate over carnivores/scavengers (72 taxa) and detritus feeders, which consist essentially of mobile taxa (61 taxa). The taxonomic richness of suspension feeders is principally represented by the cnidarians (35 taxa) and by all the sponges observed in the study zone (38 taxa). As for detritus feeders, their taxonomic richness is principally due to holothurians (31 taxa).

A multivariate analysis of relationships between the megafaunal morphospecies (or taxa) and the nodule-facies demonstrates that the intra-facies heterogeneity was significantly lower than inter-facies heterogeneity. Results show that suspension feeding assemblages were predominantly found on facies B and C in the eastern and western hills in the CCZ, particularly in facies C 5–10% and C 15–20% with slope > 15° (Tilot, 2006a), a distribution indicating the general direction of N40–N50 bottom currents (Tilot et al., 2018).

The multivariate analysis showed that the identity of the selected facies has been preserved during sampling and the analysis of relationships enables to identify preferential habitats which are displayed in Table 9.3.

Based on data from the UNESCO/IOC baseline survey of NIXO 45, a sensitivity index was developed, using a range of ecological and environmental indicators (Tilot et al., 2018). The typical (modal) values of each indicator were assessed, leading to the sensitivity index values shown in Tables 9.3 and 9.4 (Tilot et al., 2019). The results indicate that overall facies C, followed by facies C 25–40% and facies BP 35–50% may be considered the most vulnerable, with facies C 5–10% and facies C 15–20% on a slope of >15% obtaining very similar scores. These facies are hosting the most diverse and abundant megafauna according to the multivariate analysis. Facies O with recent sediments adjacent to facies C have the second highest richness and abundance values. Notably, mining interests are primarily focused on facies C and facies BP (where nodules are most abundant) (Tilot, 2006a).

Table 9.4 Summary of the sensitivity to natural and anthropogenic impacts of certain ecological characteristics of the megafaunal communities associated with polymetallic nodules in the CCZ (adapted from Tilot et al., 2018, Table 9.4)

5.3 Indicator Species of the Abyssopelagic Layer

Into the abyssopelagic layer, sometimes several hundreds of meters above the seabed occur benthopelagic holothurians such as bioluminescent Enypniastes eximia and Peniagone leander (Ohta, 1983; Tilot, 2006a, 2006b, 2010; Rogacheva et al., 2010), which can serve as indicator species for near-bottom regions.

As well, Bonnelid worms with about 2 m well-recognisable mounds are indicators of certain polymetallic nodule habitats (Tilot, 2006a, 2006b), which are also targeted by mining. Fixed organisms are also good indicators of bottom currents as utilised in the analysis of NIXO 45 (Tilot, 2006a, 2006b; Tilot et al., 2018).

6 Rapid Environmental Assessment in the Water Column

Because pelagic ecosystems are dynamic and subject to vertical as well as horizontal movement and seasonal migrations, the environmental effects of mining will likely be dispersed over considerable areas, both through Lagrangian (drifting) dispersion and through geophysical processes and oceanic currents. Rapid Environmental Assessment (REA) of the benthic environment does not rule out complimentary monitoring of the water column, especially since some parameters can be monitored rapidly or continuously using sensors and software. Such key water column parameters include temperature, salinity, dissolved oxygen, primary chlorophyll concentration, pH, alkalinity, turbidity, and currents. The collection of data can be extended by the deployment of Moving Vessel Profilers (MVPs), which can report data on water column profiles and sediment classification within the water column (Paka et al., 2020). Determining current speeds at depths of more than 1000 m, which ship-borne acoustic current profilers cannot reach, is more challenging. Currents can be determined at greater depths using Lowered Acoustic Doppler Current Profilers (LADCPs), although the software processing required to generate velocity profiles from the raw data is quite complex (Visbeck, 2002; Komaki & Kawabe, 2007).

Also in the water column it is useful to select indicator or sentinel species or taxa for monitoring purposes. Species producing bioluminescence, which include myctophid fishes, gelatinous zooplankton, crustaceans, jellyfishes and cephalopods (Irigoien et al., 2014) may be particularly appropriate, given their enhanced visibility in a healthy ecosystem. Moreover, their diel vertical migration (DVM), typically between photic and aphotic realms, likely plays a major role in benthopelagic coupling. On their death they act as carriers in the transport of carbon and nutrients to deep epi-benthic communities, thus serving a key function within the ocean's biological pump. It should be possible to detect and quantify these different forms of bioluminescence, using ultra-low light cameras, and the results integrated into large-scale monitoring programmes. This could facilitate the measurement of mass and energy transfer from the upper ocean to the seabed and consequently the effects of climate change (Levin et al., 2019). It could also enable to monitor the role of deep scattering DVMs in the behavioural responses and abundance and structure of deep-sea benthic communities (Chatzievangelou et al., 2021).

The "Tagging of Pelagic Predators" (TOPP) framework, with its various tools, logistics, and oceanographic integration, is a key asset for monitoring biodiversity, critical habitats, distribution, and abundance of key species, Apex predators, in the high seas worldwide, in particular in the CCZ (Costa et al., 2010; Tilot et al., 2024a) (Fig. 9.7).

Fig. 9.7 Apex predators and charismatic megafauna tracks recorded monthly each year throughout the "Tagging of Pelagic Predators' Program" (TOPP) in the Pacific (Costa et al., 2010) in (Tilot et al., 2024b)

The CCZ is framed in red. The map shows that the CCZ is a major open ocean habitat for migratory marine species. Several whale species follow migratory routes through the CCZ to feed, driven by surface water temperature and primary productivity (Block et al., 2011; Tilot et al., 2023). The TOPP collects time-series data from key species such as air-breathing vertebrates, birds, marine mammals (including pink and sooty shearwaters, Laysan albatross, black-footed albatross, leatherback turtles, loggerhead turtles, blue whales, fin whales and humpback whales, and sperm whales), and other marine vertebrates (such as bluefin tuna, yellowfin tuna, white shark, salmon shark, mako shark, blue shark, Dosidicus gigas squid, Mola mola). These key migratory species carry a variety of archival electronic tags (pressure time at depth, temperature, external environment, internal body, light level, and geolocation) that allow researchers to record physical and chemical parameters and deduce biochemical and productivity parameters (Block et al., 2003; see in Census of Marine life: http://www.coml.org/projects/tagging-pacific-predators-topp.html). Importantly, many of these Apex predators and marine megafauna can be considered ecosystem engineers.

In this perspective, the targeted key species could be used as sentinel species. The programme can assess the uniqueness of species assemblages, critical habitats, and the range of the movements into and out of the ecosystem (Block et al., 2003; Harrison et al., 2018). Further, standardised methods could enable comparisons between oceanic regions of the world thus facilitating the assessment and management of marine resources.

Monitoring of currents in the water column and close to the seafloor can be particularly important for some of the target species. Current patterns can be investigated not only with the aid of Moving Vessel Profilers (MVPs) and Lowered Acoustic Doppler Current Profilers (LADCPs) but also with a new generation of scientific instruments that incorporate optical methodologies and image analysis, constituting a significant advance in the field of hydrological observations.

Large- Scale Particle Image Velocimetry (LSPIV) for example is an innovative technique that is being increasingly and widely adopted to estimate the surface flow velocity field of water bodies. Another image-based approach, Particle Tracking Velocimetry (PTV), is a particle tracking procedure that can be used to identify the trajectories and velocity of objects, including sediment particles, passing through the field of view of a video camera or recording. This method can be applied not only in deep water but also to a sequence of still images recorded at a fixed acquisition frequency with a still or video camera.

The data from such sensors can be used with mathematical and geospatial tools, including local 3D eco-hydrodynamic models, particle dispersion models, global ocean models, as well as species distribution models. These, in conjunction with Geographic Information Systems (GIS), represent a well-established methodology for elucidating the influence of topography, hydrography, and species ecology on the structure of faunal assemblages (Nihoul, 1981; Nihoul & Djenidi, 1991; Harkema & Weatherly, 1996; Moreno-Navas et al., 2014). Furthermore, ecohydrodynamic modelling can facilitate a more comprehensive understanding of the interrelationships between a wide range of environmental variables, including food supply, larval transport, community composition, sediment dynamics, and pollutant dispersion. The interactions are shaped by hydrological processes such as currents, internal waves, upwelling, downwelling, along with bottom topography (Henry et al., 2013).

7 Holistic Ecosystem Monitoring

Holistic ecosystem monitoring with ecosystem surveys, in particular when combined with numerical modelling, has proven to be more efficient to resolve ecosystem status and changes in the full temporal and spatial domains, but there will always be a need for single species/group surveys to obtain higher precision for some stock estimations (Eriksen et al., 2018). A holistic ecosystem monitoring adapts well to a cross-sectoral approach for systematic conservation planning throughout the water column combining ecological and socioeconomic data from the fishing, shipping and deep-sea mining sectors, such as fostered by Fourchault et al. (2024).

This approach has proven to meet the same conservation targets at a lower overall cost using a smaller area compared with sector-specific plans implemented simultaneously and could serve in the implementation of the recently signed new agreement on the conservation and sustainable use of marine biodiversity beyond national jurisdiction—also known as the BBNJ treaty or high seas treaty.

To recall that this High Seas Treaty has for general objective “to ensure the conservation and sustainable use of marine biological diversity of areas beyond national jurisdiction, for the present and in the long term, through effective implementation of the relevant provisions of the Convention and further international cooperation and coordination” and more specifically for:
– “the definition of a regulatory framework;
– the recognition of a common heritage of humanity;
– the internationalisation of decisions on environmental impact studies;
– the fair and equitable sharing of benefits arising from marine genetic resources;
– the creation of protection and conservation areas in the marine environment in order to preserve, restore and maintain biodiversity;
– the production of knowledge, technical innovations and scientific understanding”.

In particular, all principles and approaches of Article 5 (Box 1) will need to be supported by a proper baseline and regular monitoring of the quality of the marine environment, in particular the sea bottom and the water column, as recommended in the previous sections. In addition, the realisation of Environmental Impact Assessment for any activities, or simply for deciding on the areas to be protected or conserved for the future will also need Rapid Environmental Assessments, as the sites concerned will be extensive and generally presenting limited knowledge.

Box 9.1 Article 5 of the High Seas Treaty
Article 5: General principles and approaches
In order to achieve the objectives of this Agreement, Parties shall be guided by the following principles and approaches:
(a) The polluter-pays principle;
(b) The principle of the common heritage of humankind which is set out in the Convention;
(c) The freedom of marine scientific research, together with other freedoms of the high seas;
(d) The principle of equity, and the fair and equitable sharing of benefits;
(e) Precautionary principle or precautionary approach, as appropriate;
(f) An ecosystem approach;
(g) An integrated approach to ocean management;
(h) An approach that builds ecosystems resilience, including to adverse effects of climate change and ocean acidification, and also maintains and restores ecosystem integrity, including the carbon cycling services that underpin the ocean's role in climate;
(i) The use of the best available science and scientific information;
(j) The use of relevant traditional knowledge of Indigenous Peoples and local communities, where available;
(k) The respect, promotion and consideration of their respective obligations, as applicable, relating to the rights of Indigenous Peoples or of, as appropriate, local communities when taking action to address the conservation and sustainable use of marine biological diversity of areas beyond national jurisdiction;
(l) The non-transfer, directly or indirectly, of damage or hazards from one area to another and the non-transformation of one type of pollution into another, in taking measures to prevent reduce, and control pollution of the marine environment;
(m) Full recognition of the special circumstances of small island developing States and of least developed countries;
(n) Acknowledgement of the special interests and needs of landlocked developing countries.

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8 Next-Generation Ocean Monitoring in the High Seas and Deep Sea

8.1 Integration with Other Technologies

Before concluding, however, we should emphasise that there are a number of other technologies, some only just emerging, which may provide powerful tools for monitoring the deep ocean at the scale required, such as acoustics, genomics, and machine learning (AI). We will briefly discuss three of these, which may have the potential to warn of regional-scale changes in water quality and biodiversity.

First, passive acoustic monitoring of coral reefs and fish (Servick, 2014; Elise et al., 2019) is increasingly being used to complement traditional visual survey methods. Sound-pressure levels have been shown to correlate positively with coral cover, invertebrate abundance, and fish diversity (Kaplan et al., 2015). Both diel and seasonal variation in soundscapes have been recorded in reef ecosystems (Staaterman et al., 2017), and there is increasing evidence that coral reef health can be assessed by acoustic as well as visual indicators (Freeman & Freeman, 2016) at different spatial and temporal scales (Nedelec et al., 2015). Specialists in the field consider ocean acoustics a powerful tool for monitoring global activity, especially in the high seas and deep-sea environments, and there are plans for a global network of acoustic recorders to be deployed to track animal distributions and behaviour, in relation to human activities, across the oceans. Acoustics allow for observations of population- and group-scale dynamics; however, individual-scale observations, especially the determination of animals down to lower taxonomic groups like species, are challenging tasks (Benoit-Bird & Lawson, 2016).

Second, remarkable advances in our ability to replicate and sequence DNA, as a result of the development of ultra-fast automated sequencing techniques have opened the field of environmental DNA (e-DNA) (Bohmann et al., 2014). It is proving possible to identify, among other taxa, the majority of fish present in a reef area from a limited number of water samples (DiBattista et al., 2017; Stat et al., 2019; West et al., 2020). DNA samples may not detect all the species observed by standard methods, but they do detect species not otherwise recorded (Stat et al., 2019) and replicate or reveal patterns of species distributions between habitats (West et al., 2020) and between coral reef regions (Mathon et al., 2022).

Parallel advances in DNA technology have permitted the detection and identification of hundreds of microorganisms present both in water samples and within the microbiomes of marine organisms. The bacteria dominant in the water column are largely distinct from those in the microbiomes of local organisms and are likely dispersed at broader landscape scale, while also responding detectably to environmental change (Zhang et al., 2020a). Moreover, the microbiomes of at least some marine organisms show changes in response to environmental conditions, e.g., corals show shifts in species composition in response to environmental impact ahead of visible changes in the corals themselves (Ziegler et al., 2019). It seems likely that water sampling at a range of depths for e-DNA and bacterial communities could provide a means for monitoring environmental conditions at an appropriate scale, although as yet the sources and distribution of eDNA in the deep sea remain poorly understood (Laroche et al., 2020). While eDNA studies provide broad-scale views of biological communities from only a few discrete samples, determining the spatial source of the DNA, relating the measurements to population sizes and age, and the presence of confounding non-marine biological markers in samples are active areas of research that still need to be addressed (Chavez et al., 2021).

Both genomics and acoustics rely on imaging for visual verification. Imaging, a non-extractive method for ocean observation, enables the identification of many animals at the species level, elucidates community structure and spatial relationships in a variety of habitats, and reveals fine-scale behaviour of animal groups (Durden et al., 2016).

Finally, Artificial Intelligence (AI) is transforming deep-sea research by automating real-time imagery analysis of underwater imagery and environmental information, identifying species, and mapping habitats using advanced computer vision algorithms. By deploying Autonomous Underwater Vehicles (AUVs) and Remotely Operated Vehicles (ROVs) equipped with specialised sensors, AI facilitates accurate biodiversity monitoring, enabling the identification of marine species, population estimations, and the detection of anomalies that may indicate ecosystem changes or new discoveries. AI processes vast datasets collected by underwater vehicles, integrating sonar, imaging, and environmental data to uncover patterns and predict biodiversity hotspots. Additionally, AI enhances exploration through efficient navigation, real-time data analysis, and predictive modelling, supporting conservation efforts by evaluating human impacts such as fishing and pollution. By improving the precision and efficiency of deep-sea surveys and long-term monitoring, AI reduces operational costs and minimises reliance on human intervention. As a result, researchers can gain deeper insights into marine ecosystems over extensive ocean areas in the face of environmental change while fostering sustainable exploration practices.

8.2 Role of Citizen Scientists and Participatory Approach with Indigenous Peoples

As mentioned in the section on coral reef monitoring, volunteers and citizen scientists have played an important role in assisting coral reef monitoring programmes since the 1980s, particularly those employing REA or other less rigorous monitoring techniques. Reef Check[6] is by far the best known of these programmes, and both local and international volunteers have become widely involved in national reef monitoring programmes.

Reef monitoring programmes have increasingly involved local people, particularly students or fishers, in snorkel and SCUBA diving-based reef survey work, in most countries of the coral reef belt of the world, in particular in developing countries with specialised initiatives from UN agencies, international NGOs, regional and local NGOs, and universities. Reef monitoring handbooks have been developed specifically for indigenous people (e.g., Dahl, 1981b). Many local participants later continue such survey work as paid staff members of national conservation agencies or local initiatives. Moreover, the development of indicator metrics has extended beyond the scientific framework, involving local people directly in the selection of such indicators in a more inclusive and participatory manner (Duxbury & Gillette, 2007).

Pacific Islanders have long embraced their role as custodians of the stocks on which they depend and of the wider ecosystems on which these stocks in turn depend (Johannes, 1978, 2002; Tilot et al., 2021c). In the Pacific, traditional knowledge and customary practices have been applied by local communities to conserve and manage sustainably a range of marine areas and species[7] (Tilot et al., 2021b). Such protected areas are now among the best-known examples of Locally Managed Marine Areas (LMMAs), which often include deep-sea habitats. A dedicated network[8] of LMMAs has been set up to enable the exchange of knowledge and experience (Jupiter et al., 2014; Tilot et al., 2024a). The success of individual LMMAs is measured by the health of the ecosystem, habitats, and species, the reduction of perceived threats, and by estimating the well-being of the people who use the resources. LMMAs offer unique opportunities to protect cultural seascapes, enabling indigenous communities to maintain their traditional practices while actively participating in marine management (Bambridge & D’Arcy, 2014; Lewis et al., 2017).

Citizen science is an innovative research approach derived from ecological, environmental, and earth observation sciences (Peter et al., 2019). It integrates human and societal perspectives into scientific research (Adler et al., 2020). It develops new scientific knowledge (Dickinson et al., 2012; Larson et al., 2020) through bidirectional trade-offs of information and knowledge between scientists, experts, stakeholders, and volunteers or the public (Ryan et al., 2018). In this perspective, scientists can connect volunteers’ contributions and information to broader scientific literature and contributors can guide scientists in data interpretation and discover common concerns to which science can give answers and solutions through constant bottom-up observation of phenomena under investigation (Ryan et al., 2018). Volunteered users’ interactions may contribute to create large database, address the lack of information, and support large-scale surveys and interviews (Zheng et al., 2018). Citizen science differs from other participatory approaches as amateur volunteers, according to their level of expertise, are involved in the research design, data collection and interpretation processes together with experts (Bonney et al., 2009; Njue et al., 2019), while traditional participatory approaches, generally more passive, tend to leverage volunteers regardless of their own skills.

6 https://www.reefcheck.org/
7 https://www.sprep.org
8 https://www.lmmanetwork.org

Research monitoring programmes are usually approached from a top-down data-centric perspective (Kühl et al., 2020). Engagement of people with a different level of expertise and knowledge can contribute to enrich point of views in research activities and enlarge research perspectives by paving the way to citizen science approaches that also consider socio-cultural perspectives (Kühl et al., 2020). Therefore citizen science is a tool that coordinates citizens to action (Groulx et al., 2017), combining scientific purposes with decision and policy-making processes under a community-based approach (Adler et al., 2020).

Capacity-building programmes involving participatory approaches and citizen scientists have standardized protocols often replicated by EU, UN agencies, international NGOs, and local governments in developing countries, in particular for deep-sea fisheries and conservation topics (Tilot, 2013).

8.3 Operational Observation of the Oceans

Rapid monitoring of water column parameters can be further facilitated through the use of Operational Oceanography (Le Traon et al., 2001). This encompasses both measurements of ocean parameters by satellite-borne sensors and in situ measurements of physico-chemical parameters by buoys, moorings, and floats. The resulting data are transmitted in real time to base stations for analysis, including input to digital models (or "Digital twin for the Ocean"). Obviously, such a system can have multiple applications, including the monitoring of the marine environment and of the impact of climate change as well as the impact of deep-sea mining and similar anthropogenic activities on sediment transport and ocean productivity.

The collation of in situ offshore oceanographic data was initially developed at the global and regional level within the international framework of the Global Ocean Observing System (GOOS), sponsored by the Intergovernmental Oceanographic Commission (IOC) of UNESCO*. The need for routine monitoring of ocean conditions to inform science, protect ocean health, generate climate projections, and provide early warning systems has led to the identification of what have been termed Essential Ocean Variables (OEVs) by GOOS expert panels.[9] The collection of information from fixed stations has been supplemented by the transmission data via the international ARGO network from a fleet of floats and robotic instruments. These instruments include profiler floats, which not only drift with ocean currents but move vertically between the surface and one or more pre-set depths of up to 4000 m or more (Fig. 9.8a) (Estes et al., 2021)

Fig. 9.8 (a) Above: diagram illustrating the operation of a profiler float together with its mode of transmission home of data via the Argo data transmission system (figure courtesy Thomas Haessig - Ifremer). (b) Below: the location of the EMSO observatories and of the other comparable networks (figure by EMSO). https://emso.eu/observatories/

Specifically, it is anticipated that a global system of deep-sea observatories will be established that can be linked for data transmission to the extensive system of seabed telecom cables.

9 https://goosocean.org/what-we-do/framework/essential-ocean-variables/ *The UNESCO-IOC Global Ocean Observing System (GOOS) programme is organized around globally coordinated observing networks and a heterogeneous set of regional alliances established around regional groupings of nations with common interests.

The European Multidisciplinary Seafloor and water column Observatory (EMSO) actioned by eight Member States (Italy, Spain, Portugal, France, Ireland, Norway, Greece, and Romania) is a unique marine multidisciplinary, distributed research infrastructure, with the goal to explore, monitor, and better understand the phenomena happening within and below the oceans and their critical impact on the Earth (https://www.ifremer.fr/fr/infrastructures-de-recherche/observerles-fonds-marins-sur-le-long-terme). EMSO research infrastructure provides relevant information for defining environmental policies based on scientific data, covering a wide range of interdisciplinary areas, from polar to tropical environments, down to the abyss. EMSO includes the EMSO-Azores deep-sea observatory node (Fig. 9.9) which is set atop an active volcano that hosts one of the largest active ridge hydrothermal vent sites of the Mid-Ocean ridges.

Fig. 9.9 Existing models of seafloor and seabed observatory platforms within the European Multidisciplinary Seafloor and water column Observatory (EMSO) Azores. It includes 14 observatory and test sites with multiple sensors secured through the water column and on the seabed, which continuously measure a wide range of biogeochemical and physical parameters which are transmitted directly to a large and diverse group of users. https://emso.eu/observatories- node/azores- islands/

Thus, EMSO brings together diverse and numerous scientific partners, Institutes and Research Centres operating in key sites in the European Sea in a common strategic framework of scientific facilities to promote and drive advances in marine science and technology while enabling access to its services, facilities, and technology platforms, from polar to tropical environments, down to the abyss. Similarly, in the USA, the Monterey Bay Research Institute (MBARI) has constructed the Monterey Accelerated Research System (MARS)[10] which at a depth of 900 m provides a seabed platform connected to the shore by a 51-km-long power and fibre-optic cable. Through the platform's "science node" up to eight long-term experiments can report in real time. In total at least six regional networks are being developed serving engineering, science, and education (Fig. 9.10).

Fig. 9.10 The Monterey Accelerated Research System (MARS) (https://www.mbari.org/technology/monterey-accelerated-research-system-mars/) set up at 900 m by the Monterey Bay Research Institute (MBARI). https://www.mbari.org/technology/monterey-accelerated-research-system-mars/

Another solution is to use existing telecom cables to ensure monitoring at a global scale with data transmission and power, see: https://resources.geant.org/wp-content/uploads/2022/10/2210_Geolab_EllaLink_rev.pdf and https://ihr.iho.int/articles/science-monitoring-and-reliable-technology-smart-to-monitor-the-ocean-using-submarine-cables/.

As well there are regional cabled continental margin arrays. As an example, on the north-western American continental margin, an infrastructure is set up off the continental slope near the Cascadia subduction zone, near an area with methane hydrates. As illustrated, it connects to other offshore and shelf sites (Fig. 9.11).

Fig. 9.11 Map of the regional cabled continental margin array located just off the continental slope near the Cascadia subduction zone, on the continental slope at Southern Hydrate Ridge (an area with methane hydrates), and then connects further up the slope to the Endurance Array Oregon Line at the offshore and shelf sites (https://oceanobservatories.org/array/cabled-continental-margin-array/)

These Internet-connected observatories support research on complex Earth processes in ways not previously possible, providing scientists, communities, and decision-makers with access to free, open, continuous, long-time series data from anywhere in the world, in real time.

10 https://www.mbari.org/technology/monterey-accelerated-research-system-mars/

9 Conclusions

In conclusion, our experience suggests that surveillance of the extensive areas of deep seabed (and the huge volume of the water column above the seabed) that need to be monitored as a safeguard against the effects of deep-sea mining is best achieved by making extensive use of a range of methods that have become commonplace in rapid environmental monitoring of other terrestrial and marine environments.

These include the use of standardised semi-quantitative measures and a focus on selected taxa and indicator species. Power analysis of the data often justifies this approach, indicating that because of the patchiness of the biota and habitats at multiple scales, the precision of surveys tends to be limited not by the accuracy with which individual images or samples are analysed but by the number of images or samples inspected. Assuming a finite resource and manpower available to undertake the work, although AI can increasingly help, it is better and more achievable to make use of rapid assessment techniques to secure the large sample sizes required.

A major advantage of survey work based on still and video imagery is that that imagery can be retained and archived if required for further reference. If an REA type of analysis of the imagery suggests that a site or area may be of particular interest or concern when comparing it to the norm across large areas, then that data can be re-examined and analysed in greater detail (i.e., with greater accuracy). This is clearly a better strategy than finding that it is only possible to examine an inadequate number of stations or to analyse the imagery from only a portion of the stations recorded.

9.1 Recommendations for Use of Video and Still Imagery for Deep Seabed

Based on the considerations above, we propose a number of recommendations for the analysis of still and video imagery linked to broadscale surveys of deep reefs and deep seabed. These are not intended for detailed studies of a particular site or areas undertaken for specialised ecological or taxonomic purposes where other considerations will apply. Rather they are intended to be of value where very large areas of thousands of km2 need to be assessed against environmental change. The following steps are proposed for analysing video and still imagery data:

1. The size of sampling units should be standardised for particular types of habitat so that data can more easily be compared between research teams and study areas.
2. For rapid analysis of substrate cover in video we recommend the annotation of data for discrete periods of time, e.g., for successive two-minute periods for shallow water, e.g., coral reefs, and 20–30 min periods, e.g., abyssal plains (after statistically testing the necessary period) considering the dispersal of megafaunal communities and of differentiated substrate.
3. Where very large quantum of video data has been recorded, it may be most effective to separate analysed segments by sections that remain unexamined, at least to begin with, e.g., by analysing the video imagery for 2 min once every 10 min for shallow habitats or deep habitats where megafauna is concentrated such as the summits of seamounts, hydrothermal vents, and longer sequences (20–30 min every 60–90 min, for example, after statistical testing).
4. Within each unit of analysis, we recommend that a set of characteristics be annotated using a semi-quantitative scale, such as the Deep Sea Semi Quantitative (DSSQ) scale developed for nodule-facies, where the values are judged visually with respect to each set

period; these characteristics include slope, rugosity, substrate character, grain size, and current strength which should be judged using defined scales ranging from five to ten points, or more after if statistically tested for each deep-sea habitat.
5. To set a lifeform taxonomic identification guide for deep-sea benthic megafaunal assemblages associated with an identified deep-sea ecosystem (hydrothermal vents, cobalt-rich crusts, polymetallic nodules, cold seeps, abyssal hills, seamounts, canyons, trenches, etc.), such as the one assembled for polymetallic nodule ecosystem (Tilot, 2006b).
6. Also within the DSSQ scale, to assess the abundance of major categories of lifeforms, such as sponges, actinians, echinoderms, molluscs, fish, dominant trait characteristics, substrates, and physical parameters.
7. The use of semi-quantitative scales (such as the DSSQ scale) does not preclude the fully quantitative counting of limited numbers of features or taxa of special interest, such as those identified as indicators of specific habitats or environmental conditions.
8. To adapt spot sampling to the deep sea context, with photo and video transects for sampling. In the coral reef context, the analysis of close-up video and still images indicates that five spot samples (recording the biota or substrate underneath five fixed positions within the image) are usually sufficient from a statistical point of view; otherwise, ten spot samples may be required to ensure accuracy. It is necessary to test this technique in the deep sea and assess the number of spot samples needed to enable a statistically valid assessment of deep-sea benthic communities.
9. We recommend that, as in various coral reef programmes, a hierarchical system of taxon identification be used for deep sea individuals and communities, whereby less experienced observers principally record biota as lifeforms and more experienced observers record biota by reference to orders, families, or perhaps genera so that specialist taxonomists can still work at species level where desirable or available.
10. We also support many of the recommendations of Horton et al. (2021) concerning the use of a system of Open Nomenclature (ON).
11. It would be beneficial if the DSSQ scales used for assessing different physical parameters, habitats, and lifeforms were standardised, and the descriptors of the different grades were agreed upon. This would allow more robust statistical comparisons of the data obtained by different studies and research teams in different regions of the world's oceans or at different times.

9.2 Recommendations for the Water Column

We emphasise that the adoption of REA principles for the seabed does not preclude the necessity for monitoring key water column parameters such as turbidity, current speed, dissolved oxygen concentration, primary production (chlorophyll), and alkalinity. It is fundamental that a long-term, three-dimensional monitoring system recording these environmental variables be established not only on the seafloor but also in its subsurface, throughout the water column, the ocean surface and above.

Monitoring of currents in the water column and close to the seafloor is particularly important. Ecosystem monitoring is a holistic approach that enables to monitor the spatial-temporal connections between physical and chemical oceanography, biogeochemistry, marine geology, plankton, nekton and benthos ecology and biology, food web dynamics, and marine biogeography defining highly integrated Cells of Ecosystem Functioning (CEF) (Boero et al., 2024).

9.3 Other Recommendations

Holistic ecosystem monitoring adapts well to a cross-sectoral approach for systematic conservation planning from the seafloor throughout the water column combining ecological and socioeconomic data from the fishing, shipping, and deep-sea mining sectors. This integrative approach serves well in the implementation of the new High Seas Treaty.

Nevertheless, it is clearly desirable that where possible such Rapid Environmental Assessment (REA) techniques be combined with the use of other forms of monitoring that can be applied on a broad scale. These range from collation with data from satellite-born sensors to the use of operational oceanographic methods and systems such as GOOS, as well as in situ measurement of key water column parameters and eco-hydrodynamic modelling which can facilitate a more comprehensive understanding of the interrelationships between a wide range of environmental variables, including food supply, larval transport, community composition, sediment dynamics, and pollutant dispersion.

We also suspect that acoustic monitoring, tagging of apex predators with sensors (such as the TOPP), using bioluminescence as a health and migration indicator, and using genomics in particular environmental DNA sampling with the help machine learning (AI) will become a cost-effective means of broadscale monitoring of ocean health.

At the same time, resource and manpower limitations and deep sea awareness can be addressed by making use of a participatory approach of stakeholders, citizen scientists, and student volunteers, although this must always be subject to effective training and regular testing. In particular we encourage the full and equitable participation of indigenous people and local communities (IPLCs), both in monitoring and research and in governance processes.

To conclude, the standardisation of REA worldwide through Multidisciplinary Seafloor and Water Column Observatory Platforms would enable joint management decisions at the regional and global level, aligned with the scale of oceanographic processes and management decisions. Such an approach should be of value in surveying and monitoring broad ocean landscapes not only in response to the potential impacts of deep-sea mining but also in assessing areas for inclusion within the 30% of ocean areas intended to be protected under the Convention on Biological Diversity. This aligns with the new High Seas Treaty.

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Dr Virginie Tilot holds a PhD in Biological Oceanography and an ecological engineering degree (conservation and fisheries). She is honorary Attaché of the National Museum of Natural History, Centre d'Écologie et des Sciences de la Conservation (CESCO), Paris (France), and is titulary member of the Académie Royales des Sciences d'Outre-Mer (ARSOM), Belgium, and a corresponding member of the Académie des Sciences d'Outre-Mer (ASOM), France.
Presently she is a research scholar and senior adviser in sustainability sciences, marine biodiversity, and ecology, specialised in coral reefs, deep sea and high seas, marine spatial management, and planning and conservation. Dr Tilot has more than 35 years of experience and has participated in 43 regional projects in the Atlantic, Pacific, and Indian oceans, the Red and Mediterranean seas, and more than 50 countries in the world, mainly in the Indian and Pacific oceans and the Red Sea for diverse national and international organisations, institutions, universities in particular for agencies of the United Nations (UNEP, UNDP, FAO, UNESCO/IOC, etc.) and EU. She has a total of 150 publications and 78 communications. Concerning the domain of deep-sea mining, Dr Tilot has issued recently four peer-reviewed publications, several communications in international meetings, three volumes in the UNESCO/IOC Technical Series on “Biodiversity and Distribution of faunal assemblages and options for the management and conservation of the nodule ecosystem in the Clarion Clipperton Fracture Zone” and chapters in four Springer book Series on deep-sea mining, Ed. R. Sharma, Environmental Issues of Deep-Sea Mining: Impacts, Consequences and Policy Perspectives, Perspectives in Deep-Sea Mining, Deep-Sea Mining and the Water Column, and Deep-Sea Mining: Management, Policy and Regulation (this volume).

Dr Nadia Aarab is a renowned expert in environmental sustainability, with over 23 years of experience in the oil and gas industry. She currently serves as Group Sustainability Manager at DeepOcean Group, where she has structured sustainability programmes and leads the implementation of the Corporate Sustainability Reporting Directive (CSRD). She is also responsible for executing the company's net-zero strategy and integrating sustainability as a core business function. Her career spans marine environmental protection, aquaculture, oil spill preparedness, and energy transition. She has held key roles, including Senior Marine Scientist and Senior HSE Professional in Emergency Preparedness. Dr. Aarab holds a PhD in Oceanography from CNRS-University of Bordeaux 1 and a master's in Biosciences from the Oceanography Centre of Marseille, France. She is also a Lean Six Sigma Black Belt, specialising in strategic roadmaps for sustainability in the energy sector. In 2024, she was elected Chair of the Environmental Sustainability Committee at IMCA, where she plays a key role in shaping global sustainability guidelines. She also leads deep-sea mineral sustainability initiatives and contributed to the ESG handbook for marine minerals in collaboration with DNV and other institutions.

Beyond her corporate role, Dr. Aarab is the co-founder of Clean Shores, a nonprofit raising awareness about plastic pollution and the importance of shoreline cleaning. A recognised thought leader, Dr. Aarab is frequently invited to speak at IMO, COP28, the International Seabed Authority (ISA), the UN Global Compact, and the UN Marine Program (UNMP), contributing to global sustainability efforts.

Dr Juan Moreno Navas has an honours degree in Marine Science from Cadiz University (Spain) and a PhD in Marine Aquaculture Spatial Planning from Stirling University (UK). His PhD degree focused on marine vulnerability, spatial analysis, and hydrodynamic and ecological models. He was a postdoctoral research scientist, working mainly on GIS, spatial analysis, and biodiversity in cold water coral habitats at Heriot-Watt University (UK). He joined the Physical Oceanography Research Group at Málaga University (Spain) for 2 years as research associate, working on hydrodynamic modelling. He is currently working as a research technician in artisanal fisheries management and modelling at the Spanish Institute of Oceanography (CNIEO-CSIC). Concerning the domain of deep-sea mining, he is coauthor of a publication on "The benthic megafaunal assemblages of the CCZ (Clarion Clipperton Zone, Eastern Pacific) and an approach to their management in the face of threatened anthropogenic impacts."

Dr Arthur Dahl is a doctorate in Marine Biology specialising originally in monitoring and modelling complex coral reef ecosystems, before moving to New Caledonia to become the Regional Ecological Advisor to all the Pacific Island Countries, assisting with nature conservation and environmental management integrating indigenous knowledge, and organising the Secretariat of the Pacific Regional Environment Programme (SPREP). Dr. Dahl then joined the United Nations Environment Programme (UNEP) in Nairobi, Kenya, as Deputy Director of Oceans and Coastal Areas, responsible for the Regional Seas Programmes, and finalised the Oceans and Coastal Areas chapter of Agenda 21, before becoming coordinator of the UN System-wide Earthwatch in Geneva, Switzerland, covering all monitoring and assessment of the global environment. As a scientist of complex systems, he is interested in the interface between science, values, and global governance as we try to transition to a more just and sustainable society. His most recent book (with Augusto Lopez-Claros and Maja Groff) is entitled *Global Governance and the Emergence of Global Institutions for the twenty-first Century*.

Dr Alain Jeudy de Grissac holds a PhD in Marine Geology, Sedimentology, and Oceanology from the University of Marseille (France). With more than 40 years of experience with different international organisations or institutions, he is presently a corresponding member of the Académie des Sciences d'Outremer (France) and is still active as an independent expert specialised in projects related to marine and coastal conservation, natural and cultural heritage, local community development, and planning for sustainable development. His main focus is on marine and coastal protected areas, from the identification, creation, management, and evaluation of individual sites or networks of sites to the development of national policies or action plans, improvement of national legislation, and training of national administrations. He participated in the creation of more than 70 marine protected areas in the world and managed some of them. Concerning the domain of deep-sea mining, Dr. Jeudy de Grissac was mandated by IFREMER to participate in research cruises in the Pacific to assess polymetallic nodule deposits in the CCZ. He has also participated in different technical meetings and the drafting of guidelines on deep-sea conservation and management in the world oceans.