Minor Dissertation

Assessing the utility of open-source data in exploring benthic biodiversity in mining concessions off the South African coast

**Picture by Peter Chadwick, sourced from SANBI article (2019).*

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Abstract

Extractive activity in the marine realm is a well-recognised pressure on the marine environment, particularly for the preservation of biodiversity (Majiedt et al. 2019). Data that are openly available, from repositories, guides and within other studies, are a growing resource available to researchers, with the benefits including accessibility, cost-effectiveness, and access to long-term data. Data were sourced from the Ocean Biodiversity Information System (OBIS), Offshore Invertebrate Field Guide (Atkinson and Sink) and mining impact datasets (Steffani and Pulfrich 2008, Cook 1995, 1996, 1997 and 1999) to explore the utility of openly available data in exploring benthic biodiversity within two mining concessions on the west and south coasts of South Africa. Lists of benthic taxa were generated, and biodiversity quantified using quantitative measures (species richness, Shannon-Wiener index) and multivariate analyses, where possible. Each dataset provided a different aspect of the benthic biota within the concessions, including taxonomic data (i.e., species, genera, class, phylum) that are easily quantified within a particular region. Long-term data available from OBIS allowed for patterns to be visualised over time, although this was constrained by data gaps, differences in methodology and lack of metadata, for instance. For the purposes of assessing how anthropogenic pressure impacts biodiversity, the utility of open-source data was limited to datasets that considered the impacts of mining in particular. To consider the impacts different types of extractive activity have on biodiversity at a finer scale, in-situ sampling of the proposed area is substantially more beneficial than open-source data in assessing the particular ways ecosystems are impacted by anthropogenic activity.

Key words: Open-source, biodiversity, mining, benthic

Acronyms

AfrObis	African Ocean Biodiversity Information System
CO_2	Carbon Dioxide
DAFF	Department of Agriculture, Forest & Fisheries
DEA	Department of Environmental Affairs
DFFE	Department of Forestry, Fisheries and Environment
DMRE	Department of Mineral Resources and Energy
EBSA	Ecologically and Biologically Significant Area
EEZ	Exclusive Economic Zone
EIA	Environmental Impact Assessment
H_2S	Hydrogen sulphide
ISA	International Seabed Authority
IUCN	International Union for Conservation and Nature
MCM	Marine and Coastal Management Department
MDS	Multidimensional scaling
mM	Millimoles
MPA	Marine Protected Area
OBIS	Ocean Biodiversity Information System
PERMANOVA	Permutational Multivariate Analysis of Variance
PERMDISP	Permutational Analysis of Multivariate Dispersions
PRIMER	Plymouth Routines In Multivariate Ecological Research
QGIS	Quantum Geographic Information System
SAEON	South African Environmental Observation Network
SAMC	South African Museum
SANBI	South African National Biodiversity Institute
SIMPER	Similarity Percentage Analysis
SIMPROF	Similarity Profile
TCP	Technical Co-operation Permit
UN	United Nations
UNCLOS	United Nations Convention on the Law of the Sea
UNESCO	United Nations Educational, Scientific and Cultural Organisation

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1 Introduction

1.1 Resource exploitation

After rapid growth during the 20th century, 2011 saw the human population surpass 7 billion. With growth rates currently estimated at 1.05% (Roser et al. 2013), down from an all-time high of 2.1% in the latter half of the 1970's, the UN has projected the human population will reach 9.3 billion by 2050 (Lee 2011). Exponential population growth in addition to increased demand for rare earth elements has placed unprecedented pressure on the availability of finite natural resources, particularly within the terrestrial realm (Kaikkonen et al. 2018; Van Bavel 2013). With the gradual depletion of land-based resources, technological advances and rising demand for raw materials have led many coastal countries to shift their focus to greater depths than ever before (Kaikkonen et al. 2018). Despite recent interest being placed on extracting mineral resources from the deep-sea (Jones et al. 2017), mineral mining in the deep-sea has yet to commence (Miller et al. 2018a). For the most part, minerals are extracted from reasonably shallow depths of up to 50m on the continental shelf (Baker et al. 2016). However, in diamond mining, ore is regularly extracted from depths of up to 150m off the Namibia coast, using specialist vessels and equipment (Schneider 2020).

1.2 Marine mining

A number of materials including aggregates, gold, tin and diamonds have been mined from coast to the shallow depths of the continental shelf for many decades (Kaikkonen et al. 2018). Despite exploration licenses for deep-sea mining having been issued by the International Seabed Authority (ISA), the authoritative body formed to oversee matters relating to the seabed of the high seas, mining of minerals from the deep-sea has yet to commence due to the logistical and financial difficulties associated with exploration at great depths, in addition to increasing environmental concerns (Miller et al. 2018a). However, the depth at which hydrocarbons are extracted from the ocean has continually increased from the first extraction in 1897 in the shallow marine environment to depths greater than 3000m today (Cordes et al. 2016; Levin et al. 2020). Currently, there are three oceanic environments in which the extraction of raw materials is carried out, namely: coastal plains (i.e., coastal dunes, beaches), shallow depths on the continental shelf, and the deep-sea, defined as depths of more than 200m (Miller et al. 2018b).

1.3 Coastal plains

The type of material to be extracted depends wholly on resources available within a particular region. Aggregates are the most commonly-mined material around the world, used primarily within the construction industry (Baker et al. 2016, Mankelow et al. 2008). Heavy mineral sands, containing zirconium and titanium, are mined from coastal dunes and beaches, in addition to gold and diamond ore (Baker et al. 2016). Methods of extraction in coastal regions include the creation of open pits that utilise dredgers, hydraulic pumps and excavators to extract and sort material (Biccard et al. 2018). Impacts generated by mining in coastal regions include erosion of sandy beaches, habitat degradation and weakening of dune systems, with potentially drastic impacts on coastal defences (Biccard et al. 2018 in Majiedt et al. 2019).

1.4 Shallow depths

The shallow zone is defined as being from the intertidal zone to the shelf break, with most mining taking place at depths of around 140m on average (Charlier 1992). The seabed of the continental shelf is covered by many types of substrata, each supporting diverse ecosystems. Regions of soft, sandy seabed are typically dominated by species with brief life spans, rapid reproduction and high productivity (Sink et al. 2010). Where continual exposure to natural disturbance is commonplace, from ocean currents, wave action and tides, fast-growing fauna are likely to demonstrate greater resilience than slow-growing species inhabiting hard-substrates (i.e. coral reefs, gravel/mud type substrate, Pulfrich and Branch 2014, Sink et al. 2010). Physical destruction of the immediate marine environment, whether intertidal or continental shelf seafloor, is an unavoidable consequence of marine mining that results in an immediate localised reduction in diversity (Niner et al. 2018). In nearshore environments (< 50 m), mineral mining is carried out by means of dredging (Baker et al. 2016) and suction pumps (diver-based mining, Majiedt et al. 2019). Further offshore, equipment such as seabed crawlers and drills are used to disrupt seafloor sediments (Lewele et al. 2019, Majiedt et al. 2019).

Hydrocarbons are also extracted using drilling methods, carried out from ships, semi-submersible platforms, and fixed platforms (Majiedt et al. 2019).

1.5 Deep-sea

With growing interest in the extraction of minerals from several diverse deep-seabed habitats, notably, abyssal plains (polymetallic nodules), seamounts (ferromanganese crusts) and hydrothermal vents (polymetallic sulphides, Levin et al. 2020), the requisite for sufficient baseline data regarding the often unique and diverse fauna inhabiting the seabed is crucial prior to commencing commercial activity (Cordes et al. 2016). The deep-sea is the most widespread habitat on our planet, unexpectedly supporting rich biodiversity (Paulus 2021). However, our understanding of the deep-sea and the biodiversity it supports has been limited by the financial and logistical constraints of conducting research at great depth, over large spatial scales (Thurber et al. 2014, Webb et al. 2010).

The deep-sea provides a wide range of ecosystem services that are essential to the functioning of ocean systems (Armstrong et al. 2012, Thurber et al. 2014). Deep-sea organisms contribute to supporting ecosystem services in various ways, for example, through the cycling of nutrients such as nitrogen and oxygen (Niner et al. 2018). They also perform regulating services through nutrient regeneration, climate regulation (through carbon sequestration) and waste detoxification, all of which contribute to the functioning of our oceans and terrestrial world (Armstrong et al. 2012, Thurber et al. 2014).

Deep-sea habitats are less affected by frequent natural disturbances such as storms, wave action and currents, unlike the mid to upper layers of the water column that are subject to continual movement (Drazen et al. 2020). Many deep-sea species can survive in the aphotic environments found at great depths because of life-history traits such as low metabolic rate, longevity, late maturation and lower reproductive rate linked to declining temperature as depth increases (Durkin et al. 2017, Montero-Serra et al. 2018). These traits, in conjunction with limited natural disturbance, are thought to make species less resilient to changes in their environment, making them particularly vulnerable to anthropogenic disturbance (Armstrong et al. 2012). Recovery rates for deep-sea communities are likely to be extremely slow on account of these traits, in addition to their depth and distance from a sedimentation source (e.g., river mouth, Biccard et al. 2018, Niner et al. 2018).

1.6 Impacts of Marine Mining

1.6.1 Negative impacts

Marine mining is known to produce a number of direct impacts that include the physical destruction of the benthos resulting in a loss of biodiversity, habitat, ecosystem services, changes in geochemical and biochemical processes and sediment composition (Le et al. 2017, Levin et al. 2020; Niner et al. 2018). The presumed rate of recovery for macrofauna following offshore diamond mining is between four and eight years (Pulfrich and Steffani 2007). However, previous studies have reported timeframes ranging from a minimum of two years for dredged habitats, up to a maximum of 15 years for rocky reefs impacted by diamond mining (Newell et al. 2004, Pulfrich and Branch 2014, Pulfrich and Penney 1999). The indirect impacts produced by mining are often widespread and may include the following examples: 1) Increased water turbidity and the release of potentially toxic substances (i.e., hydrogen sulphide H_2S) stored in seabed sediments. Disturbance of toxic compounds such as H_2S may cause eruptions that contribute to hypoxic conditions, resulting in the mass mortality of marine species (Currie et al. 2018; Kaikkonen et al. 2018). 2) Noise, light and electromagnetic pollution generated by mining activity spreads over large distances, modifying behaviour and communication in marine species (Drazen et al. 2020, Popper and Hawkins 2019). 3) Introduction of invasive species by way of a suitable vector, with marine infrastructure, a well-recognised stepping-stone for invasive species to disperse over wide distances (Capel et al. 2019, Degraer et al. 2020, De Mesel et al. 2015, Wanless et al. 2010). 4) Where deep-sea sediments are disturbed, vast amounts of carbon stored in deep-sea sediments may be released into the ocean, with wide-spread impacts on climate regulation (Collins et al. 2010; Levin et al. 2020). Excavation vehicles, drilling and discharged waste may also create localised warming, of up to 11.4°C (Bashir et al. 2012). If ocean temperatures rise further, temperature thresholds of some biota may be exceeded, affecting growth, metabolism, fitness and reproduction (Miller et al. 2018b). The release of significant amounts of marine carbon would cause an inevitable acceleration in climate change drivers such as acidification, warming and changes to circulation patterns that could increase the spread of contaminated particles and affect larval transport (Collins et al. 2010; Levin et al. 2020).

The rate at which a habitat recovers from mining is dependent on several factors, including, type of mining, location, substrate, equipment utilised, duration of disturbance and speed at which the mine is re-filled (Boyd et al. 2005, Pulfrich and Steffani 2007). Some researchers believe that marine infrastructure may positively impact biodiversity within a locality by functioning as an artificial reef and providing a novel habitat for hard substrate species (Adams et al. 2014, Degarer et al. 2020). However, whether the provision of offshore habitats for species typical in coastal ecosystems is beneficial, is a concept that warrants further investigation. Offshore structures may also provide refugia for commercially targeted species through the implementation of surrounding exclusion zones (Atkinson and Sink 2008).

1.7 Conflicts

The potential for conflict is substantial where multiple ocean-users with differing objectives display spatial overlap with one another, occurring most notably between hydrocarbon companies and other ocean-users. In the Southwest Indian Ocean, 28.1% of Ecologically or Biologically Sensitive Areas (EBSA) and 8% of Marine Protected Areas (MPA) are currently overlapping with hydrocarbon activity. Overlap between industries with different agendas is likely to generate conflicts, particularly where activities of mining companies cause detrimental harm to conservation areas, for instance, due to an oil spill (Tibaldelschi et al. 2018). Ecotourism stakeholders may be involved in conflict with mining companies where mining operations degrade the marine environment or disturb the presence of characteristic marine species (e.g., marine mammals and sharks), through sound, light and/or vibratory pollution associated with mining surveys (i.e., seismic surveys) and operations (Drazen et al. 2020, Folkersen et al. 2018).

Potential conflict between offshore hydrocarbon activity and fisheries may transpire for several reasons, including reduced access to fishing grounds due to exclusion zones placed around active drilling to protect seabed infrastructure and trawlers, and prevent collisions that would almost certainly cause a significant environmental disaster (Atkinson and Sink 2008). By contrast, Atkinson and Sink (2008) argue that conflict with fisheries may actually be limited where exclusion zones are implemented (Atkinson and Sink 2008). Where seafloor sediment is removed or damaged during the mining process, for instance during aggregate or diamond mining, benthic habitats are unavoidably altered. This in turn may impact on the commercially-important demersal species that predate on benthic macrofauna. Furthermore, if sediments containing potential toxins are re-suspended by mining disturbance, there is a risk that these toxins may contaminate harvested seafood (Atkinson and Sink 2008, Miller et al. 2018b).

Where mining occurs offshore, there is potential for conflict between mining and shipping companies as exclusion zones may cause shipping vessels to alter their routes (Miller et al. 2018b). It is also possible that seabed mining may cause damage to the numerous undersea telecommunication cables that connect countries, particularly along South Africa's west coast (Biccard et al. 2018).

1.8 South Africa

It has been well recognised that South Africa supports a wealth of biodiversity, in both the terrestrial and marine realm (Griffiths et al. 2010, Cordes et al. 2010). Within the marine realm, current estimates indicate that greater than 13,000 marine species inhabit the waters off South Africa, including a high number of endemics, most notably on the southwest coast (Griffiths et al. 2010, Scott et al. 2012, Sink et al. 2019). South Africa's climate is strongly influenced by the oceanographic conditions of the surrounding oceans (South Atlantic and Southern Indian) that play a crucial role in shaping the diverse marine bioregions (Sink et al. 2019, Griffiths et al. 2010). Along the west coast, the cool, slow Benguela current flows towards the equator along the narrow continental shelf edge. Further inshore, wind-driven upwelling of nutrient-rich water occurs seasonally, driving some of the highest productivity in the world (Griffiths et al. 2010). On the south coast, the western boundary Agulhas current transports fast but less nutrient-rich water from the tropics, following the narrow shelf edge, before moving further offshore and retroflecting back into the Indian Ocean at the southern tip of Africa (Griffiths et al. 2010, Gründlingh 1983). As is typical of the eastern boundary upwelling systems, high productivity on the west coast supports high biomass of few characteristic species, while the south coast is essential in species population dynamics as it represents a range limit for both cool and warm-temperate species (Sink et al. 2010).

A myriad of mineral resources including hydrocarbons, heavy minerals (i.e., diamonds, titanium, zircon) and rare earth minerals such as manganese are situated in the marine environment surrounding South Africa. Resources have been partly extracted from coastal and shallow waters for more than 50 years, most notably within the oil and gas industry (Lewele et al. 2019). On the west coast, licenses for phosphate exploration have been issued, but to date, no exploration has taken place (Currie 2018; Lewele et al. 2019). Since 1965, over 300 wells have been drilled within South Africa's Exclusive Economic Zone (EEZ, Sink et al. 2010). Currently, around 98% of the EEZ is subject to rights or leases for hydrocarbon extraction (Wepener and Degger 2019), and the cumulative impact of multiple industries exert significant pressure on marine biodiversity (Atkinson and Sink 2008, Tibaldelschi et al. 2018, Sink et al. 2019). The effect of such industries on South Africa's offshore biodiversity has attracted little consideration when compared with near-shore and shallow marine environments (Attwood et al. 2000), due to the absence of data for these environments (Sink et al. 2019). Although substantial improvements in knowledge of offshore biodiversity have been made in recent times, biodiversity data for much of the offshore marine environment remains scarce, particularly in regard to smaller invertebrates (Griffiths et al. 2010, Sink et al. 2019, Yemane et al. 2020).

'Operation Phakisa', introduced in 2014, is part of the national development plan unveiled to unlock South Africa's Ocean economy through aquaculture, greater extraction of natural resources, marine transport, construction and protection and ocean governance (Lawelle et al. 2019). Greater utilisation of natural resources requires sufficient assessment of potential impacts and mitigation measures to be put in place to protect the marine environment, which differs greatly from the terrestrial realm in terms of ecological degradation. Unfortunately, the only data are collected by prospective industries such as oil and gas companies that are exploring greater depths than ever before (Sink et al. 2010), with much of the data unavailable to the greater scientific community.

1.9 Open-Source Data

The so called ‘big data’ revolution has helped accelerate the movement towards greater accessibility to open-source data that has long been commonplace in other scientific disciplines such as oceanography, genomics and meteorology (Michener 2015; Sielemann et al. 2020). In the past 20 years, substantial strides have been made towards increasing transparency and resources within the life sciences, with a number of scientific journals now requiring data submitted to open-access repositories as part of the publication process (Sielemann et al. 2020). Pace of change in ecology, however, has been slower, with some researchers still resistant to collaborative data-sharing (Michener 2015). Open-access repositories facilitate greater data sharing both within and outside of the scientific community, benefitting both publishers and authors by improving understanding of ecosystems, increasing public transparency and study reproducibility and accelerating the pace of scientific discovery (Michener 2015, Sielemann et al. 2020). Dependent on the objectives of a study, open-source data have a wide array of applications, from the development of baselines to temporal and spatial scale monitoring and informing spatial management and creation of biodiversity maps (Caro et al. 2018, Yemane et al. 2020). The latter was demonstrated by Yemane et al (2020), where authors mapped biodiversity ‘hotspots’ within South Africa’s EEZ for three functional groups, by combining data from OBIS and Bio-ORACLE, open-source repositories for biological and environmental data, respectively. Despite progress within the data revolution, there are several challenges associated with the use of openly-available data that limit further understanding. For instance, data are often unevenly distributed, inclusive metadata are often missing or inaccessible (Caro et al. 2018), and information relating to methodology between studies is lacking, resulting in different outputs (Schulp et al. 2014). Another point to consider is the technological limitation of open-source data. For example, benthic habitats are often under-represented in databases due to the technological difficulties and increased cost of sampling at great depths, particularly as remote-sensing technology is unable to visualise benthic habitats (Griffiths et al. 2010, Lavorel et al. 2017, Tittensor et al. 2010, Yemane et al. 2020).

1.10 Knowledge gaps

Despite introduction of Operation Phakisa in 2014 to build an oceans economy, there exists an absence of relevant legislative regulation or controls within the exploration sector. Lack of relevant controls that consider South Africa's highly diverse marine habitats are of particular concern where much of the seafloor remains unstudied (Lewele et al. 2019, Sink et al. 2019). Many benthic ecosystems and invertebrate species are still poorly understood in comparison to ichthyofauna of the region, particularly at depths greater than 1000m, and are thus often under-represented in research (Griffiths et al. 2010, Pulfrich and Steffani 2007, Yemane et al. 2020). Environmental Impact Assessments (EIA) are often carried out as desktop studies utilising data available for the area of focus, with in-situ sampling not a legislative requirement (Houdet and Chikozho 2015). There is, however, a definite possibility that risk to the environment may be underestimated because of outdated research, lack of specific research within the area of focus and access to data in scientific literature. This is of particular significance for benthic ecosystems, where communities are patchily distributed across the seafloor and the ecosystem services provided are less understood (Karenyi 2014, Pulfrich 2014)

1.11 Study Aims

To investigate the utility of open-source data in exploring benthic biodiversity in two mining concessions on the west and south of South Africa, over a 40-year period (1983-2018). Lists of benthic taxa will be generated and biodiversity quantified within two sites subjected to different types of extractive pressure. From this research, this study hopes to assess the current feasibility of utilising openly-available data in biodiversity studies by evaluating their usefulness in time-series investigations. In addition, the study aims to assess whether different sources of data characterise the benthos differently and also whether biases in open-source data significantly constrain our understanding of biodiversity. By comparing the data available from specific in-situ sampling of the biota and openly-available data, this research aspires to provide evidence for greater physical sampling to enhance knowledge of seafloor ecosystems.

2 Methods

This study was carried out as a desktop study by utilising open-source datasets and other freely accessible information to explore biodiversity over the period 1983–2018. The Ocean Biodiversity Information System (OBIS) was utilised, as per Yemane et al. (2020), as the main source of marine biodiversity data between 1983 and 2006. Two mining licence blocks (Fig 1) were chosen as case studies as information was more abundant, including data related specifically to the impact of different types of mining on two dynamically dissimilar coasts. On the west coast, South African Sea Areas (SASA) 2c-5c (formerly ML3), is situated offshore from Namaqualand. On the south coast, Block 9 is situated on the mid Agulhas Bank, offshore from Mossel Bay. The 'Field guide to offshore marine invertebrates of South Africa' compiled by Dr Lara Atkinson and Dr Kerry Sink (2018) was utilised to produce a presence-only dataset for the two concession areas, based on long-term monitoring trawls carried out by the Department of Forestry, Fisheries and Environment (DFFE, formerly Marine and Coastal Management) between 2011 and 2018. The third dataset utilised mining impact data that were specific to each concession, to monitor changes in macrofauna following diamond mining in ML3/2003, and to monitor changes in the fouling community colonising the FA platform in Block 9 (Cook 1995-1999).

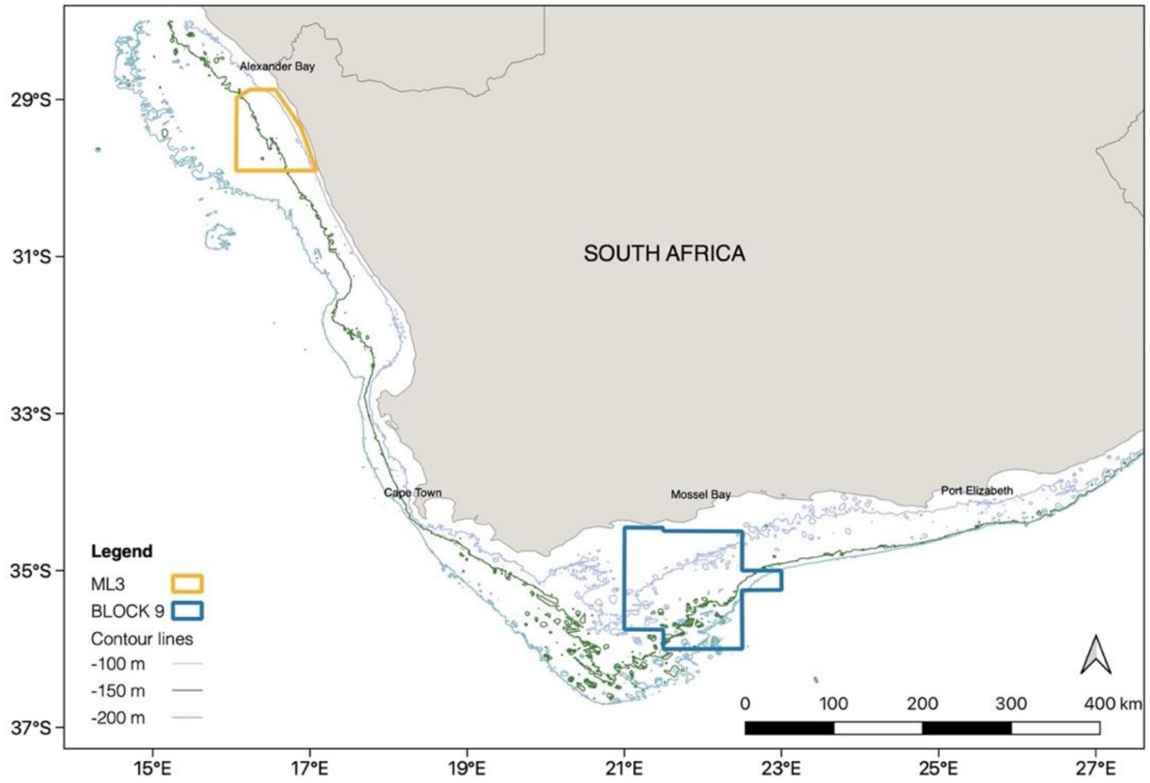


Figure 1. *Map of South Africa showing concession ML3 on the west coast and Block 9 on the south. Bathymetry lines for -100m, -150m and -200m depth shown. Map was created using QGIS (version 3.10.11) and bathymetry data was downloaded from GEBCO 2020 gridded bathymetry data download application (https://download.gebco.net).*

2.1 Case Studies

2.1.1 ML3/2003

The first study area, ML3, comprises concession areas 2c, 3c, 4c and 5c on the northern west coast of South Africa, 500km northward of Cape Town and approximately 5km offshore of Alexanders Bay to the North and Kleinsee to the South. The type of extractive activities that have taken place within ML3 include diamond mining and hydrocarbon exploration, with overlap observed between the two industries (Pulfrich 2014). The site spans an area of 8816.17km^2 offshore from the Namaqualand region, with depths ranging between 50m and 200m (Roos 2008). Seabed sediment is largely composed of muddy sand, sand and mud/sandy mud further inshore (Pulfrich 2014). Benthic habitats range in classification from 'Least Concern' to 'Vulnerable' (Sink et al. 2019). The target area is located within the cool temperate Namaqua bioregion (Sink et al. 2019), dominated by the influence of the Benguela current. The Namaqua bioregion has one of three upwelling cells within the Benguela ecoregion where prevailing winds displace coastal surface water offshore, causing intense seasonal upwelling of cold water rich in inorganic nutrients (i.e., nitrates, phosphates) from natural cyclic decomposition of phytoplankton that contribute to naturally high productivity within the region (Pulfrich 2014). The Benguela region is also known for frequent episodes of anoxia that trigger 'red tides' caused by vast decomposition of phytoplankton (Shannon et al. 2020). As is characteristic of other eastern boundary upwelling regions, the west coast supports a high biomass of few species (Griffiths et al. 2010, Awad et al. 2002). Within the locality of ML3 lie several areas of conservation importance, specifically the Orange Cone EBSA, an area spanning the mouth of the Orange River between Namibia and South Africa, 50km to the north and south of the Orange River. Currently, 10 of the 16 ecosystems situated within the Orange Cone range between critically endangered to vulnerable (Sink et al. 2019). Further southward, within concession 5c, the Namaqua Fossil Forest MPA spans 1200 km^2 of seabed protecting a now-extinct species of fossilised yellowwood trees, supporting habitat-forming cold water scleractinian corals (Biccard et al. 2018). Diamond mining has not taken place in 5c as there are no economically-viable diamond reserves located in this region (Majiedt et al. 2019). Within 2-3c, diamond mining took place between May 2007 and January 2010, during which time a total of 4.8 km^2 of seabed was disturbed by a seabed crawler (Pulfrich 2013).

2.1.2 Block 9

The second study area is located within the Bredasdorp basin on the Agulhas Bank on South Africa's south coast, the site of most historic offshore hydrocarbon exploration within the South African EEZ (Lewele et al. 2019). Situated within the Agulhas bioregion where the Indian Ocean and southern Atlantic Ocean converge, the Agulhas Bank is one of the most economically-viable regions within South Africa's EEZ with significant hydrocarbon exploration, commercial fisheries and shipping occurring within the region (Sink et al. 2010). Block 9 is located offshore from Mossel Bay, inward of the 150m isobath. The study area currently includes several oil and gas operations, including the FA gas field (105m depth) and Oribi/Oryx oil field that includes the unfixed ORCA infrastructure (117m depth) amongst others. Several areas of conservation are in close proximity to hydrocarbon operations within Block 9, for instance, the Agulhas Bank Nursey EBSA, that facilitates a wealth of warm and cool temperate species (Hutchings et al. 2009), including several endemics such as threatened reef-associated fish, red steenbras *Petrus rupestris*, octocorals and invertebrates (Sink et al. 2010). South Africa's south coast boasts the second-highest biodiversity across the South African coast, after the east coast (Yemane et al. 2020), presumably due to the oceanography, topography, and warm-temperate conditions on the wide continental shelf. The Agulhas Bank includes a variety of habitats, from threatened muddy, sandy and mixed seabed to deep reefs and kelp forests (Sink et al. 2019). Three nursery grounds and numerous ecological processes that include spawning, recruitment, connectivity and food provision occur on the Agulhas Bank (Hutchings et al. 2002). Much of the EBSA is now protected by the De Hoop, Stillbaai and Agulhas Bank Complex MPA that includes 4300km^2 of protected area.

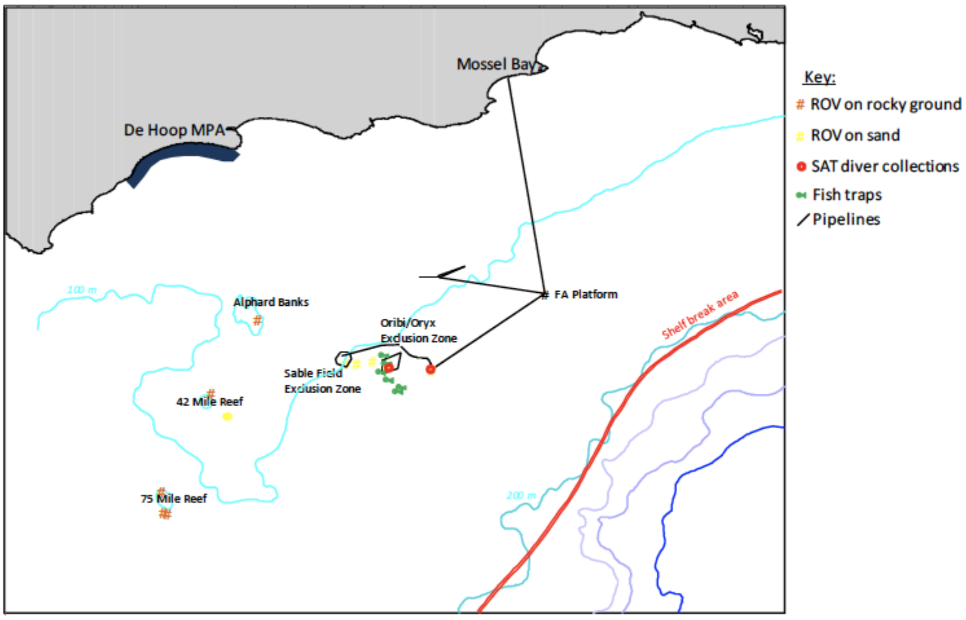


Figure 2*. From Sink et al (2010) showing the location of the FA platform, Oribi/Oryx exclusion zone, sable field exclusion zone and relative position of conservation areas (i.e. Alphard Banks and diverse rocky reefs) within the vicinity of Block 9.*

2.2 First Dataset: OBIS

OBIS has become one of the leading open-source repositories for occurrence data of all marine fauna and flora (Yemane et al. 2020). By integrating datasets from upward of 500 institutions across the globe data can be mapped and easily downloaded for regions of interest (https://obis.org). AfrObis is one of 20 regional nodes that gathers information from institutions within the African continent. Contributors to the dataset included in this study include the Iziko South African Museums fish, shark, mollusc and crustacean collection, Steffani and Pulfrich (2013), Natal Museum's mollusc collection and lastly, demersal and trawl surveys undertaken by the DFFE. Using the OBIS mapper method, all data at depth between 100m–1000m for the entirety of South African waters over a 40-year period (1980 – 2020) were selected. Reduced datasets were created by overlaying a shape file provided by the Department of Mineral Resources and Energy (DMRE) containing on- and offshore mining applications, both historical and current. By overlaying the shape file, the OBIS dataset was cut to the size of the two concessions using the geoprocessing tools on QGIS (version 3.10.11- A Coruña). De Beers monitoring data were excluded to allow for analyses as a separate dataset. Package 'rfishbase' (Boettiger et al. 2021) on R studio (Version 1.4.1103, RStudio Team 2021) was retained to identify and exclude non-benthic and non-demersal fish species for the purpose of this study. Species lists were captured for both concessions and subsequently scanned through the World Register of Marine Species (http://.marinespecies.org) database to check for compatibility of scientific names and correct assignment of class. Data downloaded from OBIS were transformed into a presence-only format for several reasons, including an absence of count data, as well as a lack of understanding surrounding different methodologies used across contributing organisations (Yemane et al. 2020). From the information available, the predominant method of sampling was trawling although some organisations opted for a dredge method, dependent on the biota being sampled. Sampling effort per year was estimated using trawling time stamps (i.e., T12, T20) within the 'event date' column of the data. Sampling effort for each mining block was highly variable both within and between years, with the number of samples ranging between 1-13 in ML3 (Fig 4) and 1-35 in Block 9 (Fig 4). Samples were also patchily distributed across each mining block, with little to no sampling in some parts of the block (Fig 3). For future studies it is recommended that sampling effort be accounted for, either by setting a lower threshold or perhaps utilising species accumulation curves (e.g., minimum of four samples). Pie charts were created using Excel to visualise which classes dominated each site, based on the number of species per class. Univariate analyses were carried out on raw data through PRIMER's 'diverse' function to produce diversity indices in terms of species richness (S). Neither dataset required transformation due to the binary format of the dataset. Jaccard similarity index was applied because Jaccard, alongside Sørensen, is the most suitable and widely used index for presence-only data. Jaccard was used to measure the similarity or dissimilarity between communities based on the number of shared species as well as species that were unique to a community (Hao et al. 2019). Multidimensional scaling (MDS) ordination and Cluster analyses were constructed to visualise similarities and dissimilarities between data points in each concession.

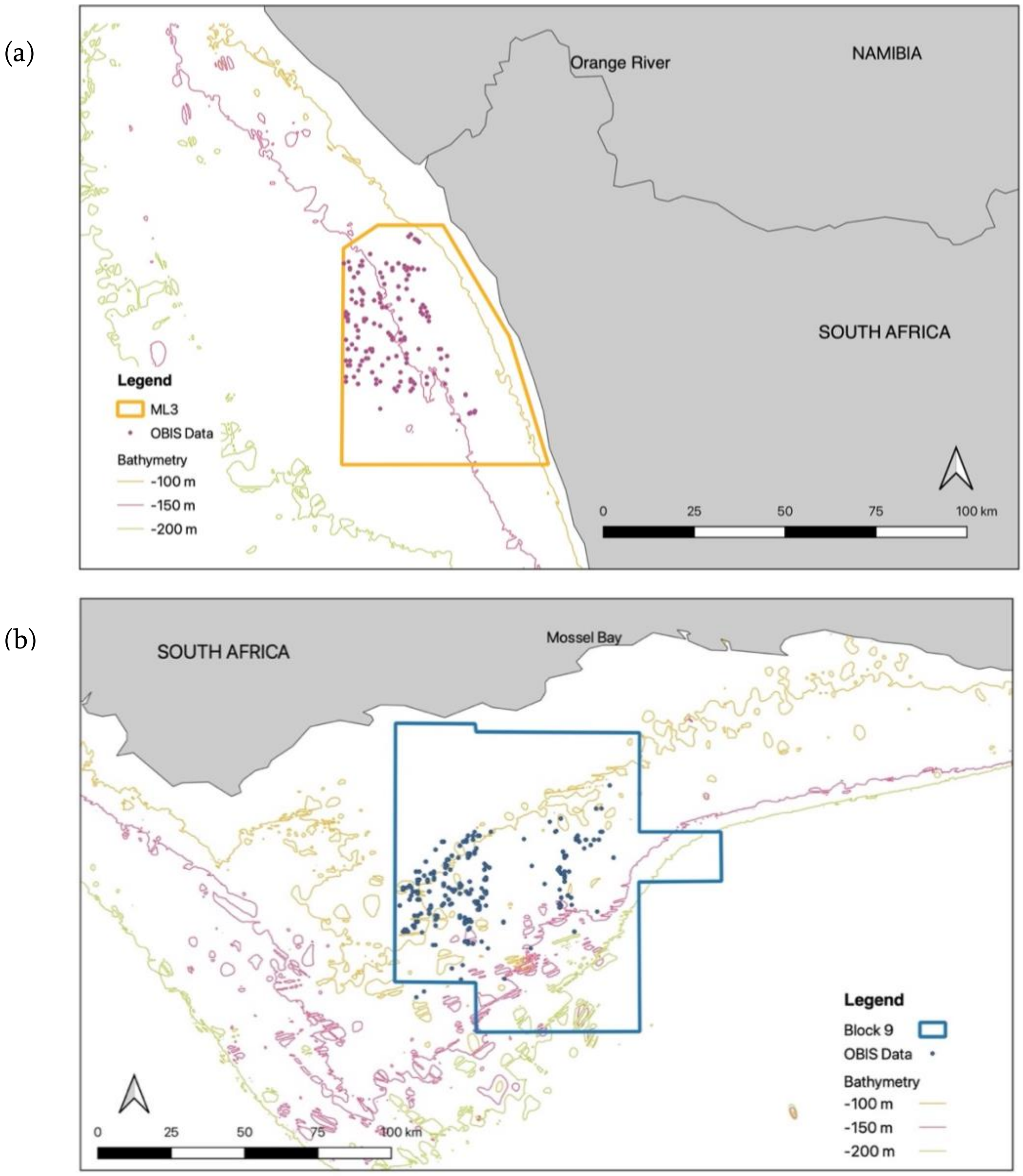


Figure 3. Concession sites with OBIS data points (a) ML3 on the west coast of South Africa, (b) Block 9 on the South coast of South Africa.

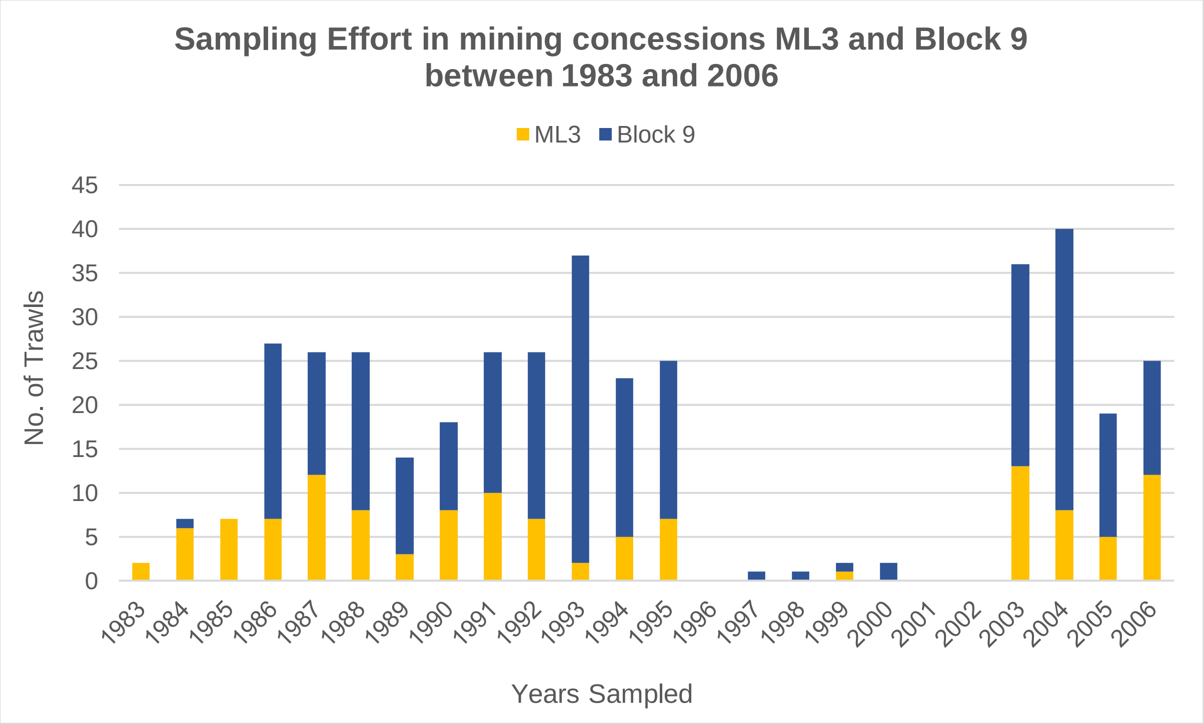


Figure 4. *Sampling effort from data obtained from OBIS for ML3 on the west coast (1983-2006) and Block 9 on the South coast of South Africa (1984-2006). Gaps represent years where data was not available.*

2.3 Second Dataset: Field Guide

The 'Field Guide to Offshore Invertebrates of South Africa' (Atkinson and Sink 2018), herein referred to as 'Field Guide' is openly-available online through the SAEON website (www.SAEON.ac.za). Long-term monitoring was established through collaboration between SAEON (Egagasini node), SANBI, DFFE, formerly known as (DAFF and DEA: Oceans and Coasts) between 2011-2018, to critically improve knowledge of offshore invertebrates, beyond invertebrates of commercial interest. Samples were collected during DFFE demersal trawls on the continental shelf from the Orange River on the west coast to Port Alfred on the east, at depths ranging between 30m and 1000m. For each species the guide includes a map of South Africa, including the EEZ and bathymetry lines, with data points to indicate the coordinates of where individuals were encountered during DFFE surveys. As the dataset is not openly accessible to the public, occurrence maps for each species were utilised to generate presence-only data for each concession. Also, for this reason it was not possible to obtain information regarding the number of samples collected, duration or equipment used during the DFFE demersal trawls. Using geographical features to define the parameters of each study site, it was possible to produce a rough estimate of invertebrate species present in each concession. Lists of species, taxonomic data and species richness (S) were extracted to characterise the epifauna within each region. Lastly, pie charts were produced to visualise the percentage of recorded species in each class and the contribution made by each class to the dataset, based on the number of species within a class.

2.4 Mining Impact Datasets

2.4.1 De Beers Benthic Monitoring Programme

De Beers Consolidated Mines (DBCM) Benthic Monitoring Programme is herein referred to as 'De Beers' or 'De Beers Monitoring'. Between 2003 and 2011, Pisces Environmental Services (Pty) Ltd carried out benthic macrofauna (>1mm, Pulfrich 2014) surveys on behalf of De Beers Consolidated Mines (DBCM) from authorisation of mining right in 2003 until 2011, with subsequent mine-closure occurring in 2013 (Pulfrich 2013). A baseline benthic survey was carried out in 2003, with successive monitoring surveys in 2008, 2010 and 2011, in both unmined (control) and mined sites to assess alterations in benthic community assemblages and monitor post-diamond mining recovery rates. A total of 40 samples were collected across the four-year monitoring period, with five grab samples collected per site using a 0.02m^2 Van Veen grab at depths of up to 142m (Steffani and Pulfrich 2013). To enable analysis of the effect of mining, 'Control' and 'Mined' sites were established by matching decimal longitude and latitude co-ordinates with established control and mined sites in data collected in Karenyi (2014). A list of all species present in the entire dataset were compiled and data were partitioned into two control sites (NI-C1, NI-C2) and two mined sites (NI-M1, NI-M2) to allow for direct comparison between all years sampled. PRIMER 6 (version 6.1.11) and PERMANOVA+ (1.0.1; Clarke and Gorley 2006, Anderson et al. 2008) software were employed for the purpose of statistical analyses. The 'diverse' function was performed to quantify the total number of species (S), number of individuals (N) and three diversity indices: Margalef's species richness (d), Pielou's evenness (J') and Shannon-Wiener index (H'). The Shannon-Wiener index of diversity was chosen for the final analyses as this index incorporates species richness and evenness into one calculation. Square root transformation was applied as abundance ranges were not orders of a magnitude different across taxa. Bray-Curtis similarity measure was applied to identify how different mined and unmined samples were, based on species found in either site.

Multidimensional scaling (MDS) and Cluster analyses were undertaken to visualise similarities and distinctive differences between data points, shown by factor 'treatment' (e.g. mined/ control). Several multivariate analyses were then applied to the data. Two-way crossed SIMPER analysis on factors 'year' and 'treatment' was performed to isolate species that contributed to 90% or more of similarity/dissimilarity. PERMDISP established whether data points were significantly dispersed from the centroid and PERMANOVA (permutational MANOVA) was applied to determine the significance of measured factors in explaining the variation within the dataset. Finally, a pairwise PERMANOVA was used to detect if years were significantly different, and if so, which pairs displayed the greatest difference.

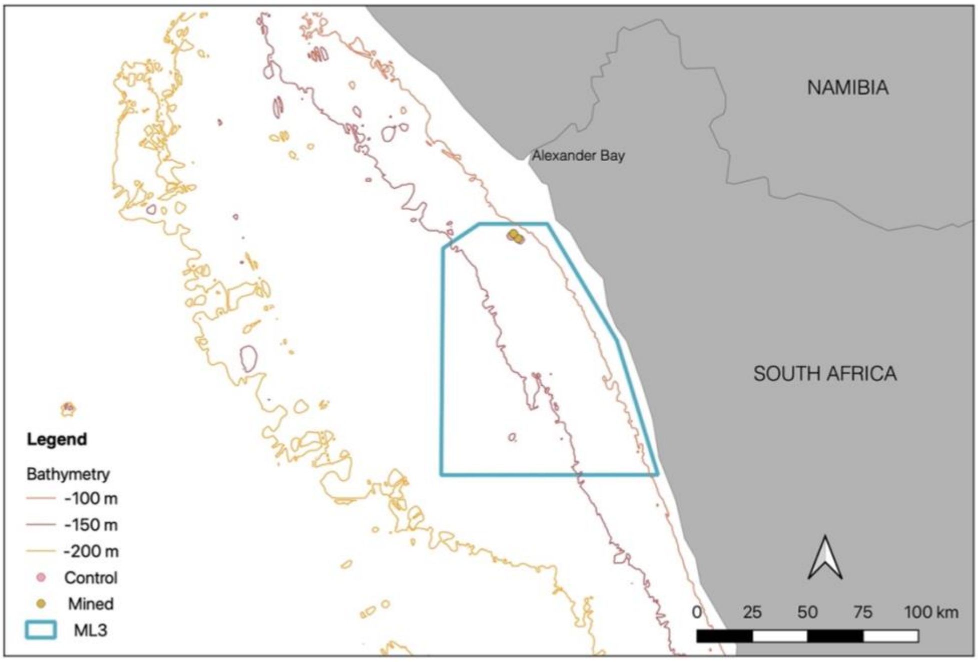


Figure 5. *ML3 on the west coast of South Africa, showing the four sites sampled (two mined, two control) by Pisces Environmental Pty in 2003 (baseline survey), 2008, 2010 and 2011 (monitoring).*

2.4.2 Cook (1995-1999)

Monitoring of fouling taxa inhabiting the FA platform on the Agulhas Bank at depths between 0-60m was undertaken by diver quadrats to identify species colonising the platform and assess changes in community assemblages in the years following construction in 1991 (Sink et al. 2010). For the purposes of this study, this dataset was utilised to enhance the overall assessment of biodiversity over time on the diverse Agulhas Bank. Although the data are unpublished, a table of findings from the dataset is available from Appendix 1 of Sink et al. (2010), which is freely available to download online. As the original dataset was not publicly available, it was not possible to determine the methodology used in the collection of the data, for example, how many times, the length of time or type of data that were collected by researchers sampling the FA platform. For each report year data were converted into presence-only in the absence of species abundance data and to enable greater comparability with the other datasets used in this study. For each of the four reports compiled by Cook (1995, 1996, 1997 and 1999), simple values of species richness were compiled to evaluate changes over time.

3 Results

3.1 ML3

3.1.1 Taxonomic composition (Phylum)

A total of 202 taxa belonging to 11 phyla, 23 classes and 128 families were extracted from all three datasets. Of the 202 taxa, 172 were identified to species or genus level, with 30 taxa identified to a higher taxonomic level. Phyla present in ML3 included Annelida, Arthropoda, Bryozoa, Chaetognatha, Chordata, Cnidaria, Echinodermata, Mollusca, Nemertea, Porifera and Sipuncula. Arthropoda (26%), Chordata (24%) and Mollusca (18%) were the best represented taxa, accounting for almost three quarters (68%) of species richness in ML3. Four phylum groups (Sipuncula, Nemertea, Bryozoa and Chaetognatha) were represented by one species each. A total of 25 invertebrate species were encountered in two of the datasets, with no species in all three. OBIS was the most diverse dataset with a total of 90, predominately demersal fish, taxa recorded. The Field Guide and De Beers datasets were the second and third most diverse datasets with 84 species of invertebrate and 55 species of macrofauna, respectively.

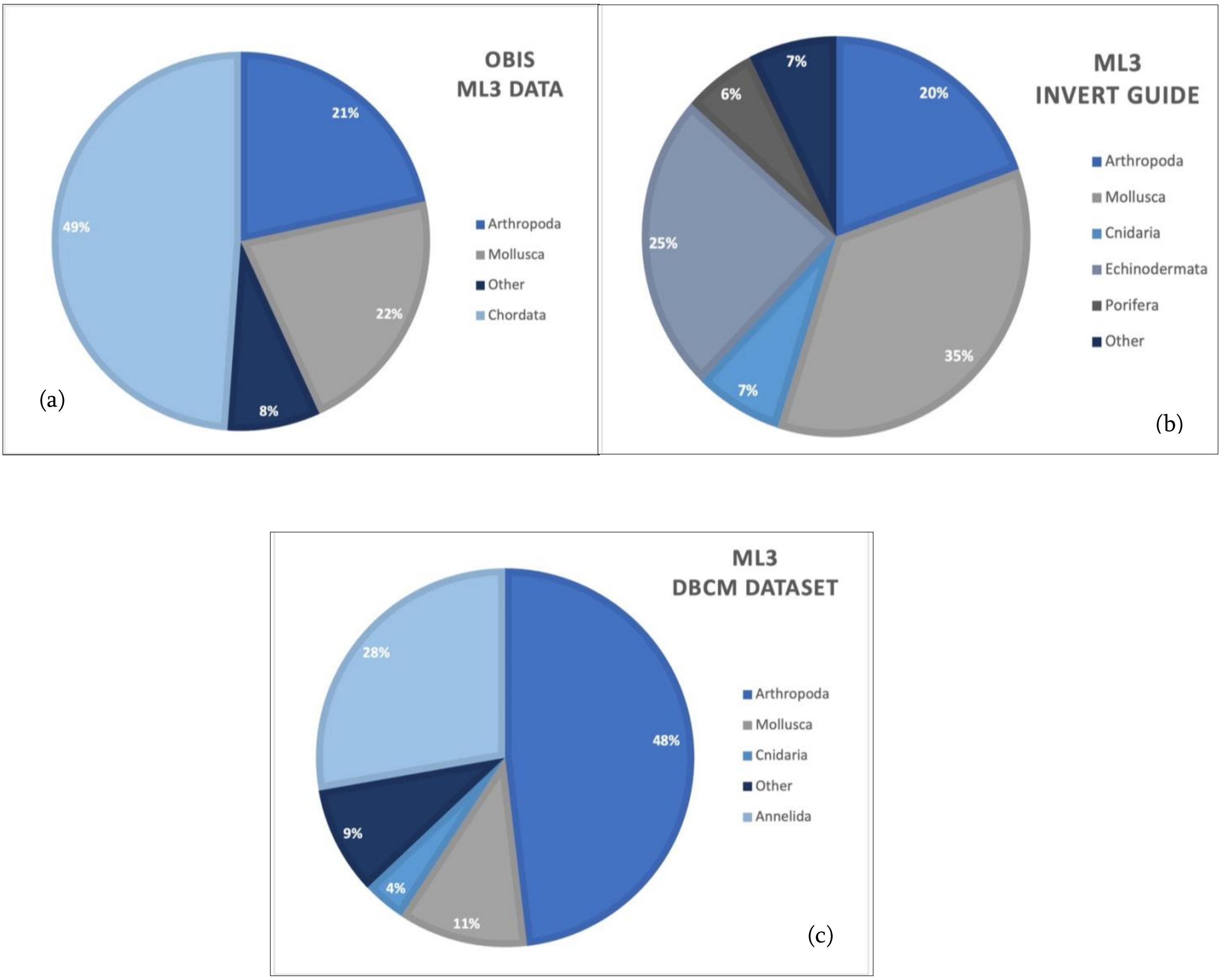


Figure 6. *Phylum percentage contribution for all three datasets in ML3, based on number of species. (a) OBIS data 1983–2006 (b) Field Guide data 2011-2018 (c) De Beers data 2003--2011. Group 'other' pools phyla with <4 species.*

3.1.2 Taxonomic composition (Class)

3.1.2.1 OBIS Dataset

Of the 90 taxa obtained from OBIS, a total of 11 classes and 56 families were identified in ML3. Benthic ecosystems were structured into four main classes; Actinopterygii, Elasmobranchii, Cephalopoda and Malacostraca. Classes represented by few (3 or less) species included

Demospongiae, Gastropoda, Myxini, Hexanauplia, Holocephali, Echinoidea, Scyphozoa. Community composition remained fairly similar across the decades studied, with minimal fluctuation in species richness within class groups. Demersal/ benthic fishes (Actinopterygii and Elasmobranchii) were consistently the most diverse taxa across all decades, contributing to more than half of the total diversity in ML3. Diversity at the community level increased over the decades studied, with greater contribution from single or few species recorded in the 1990s and 2000s.

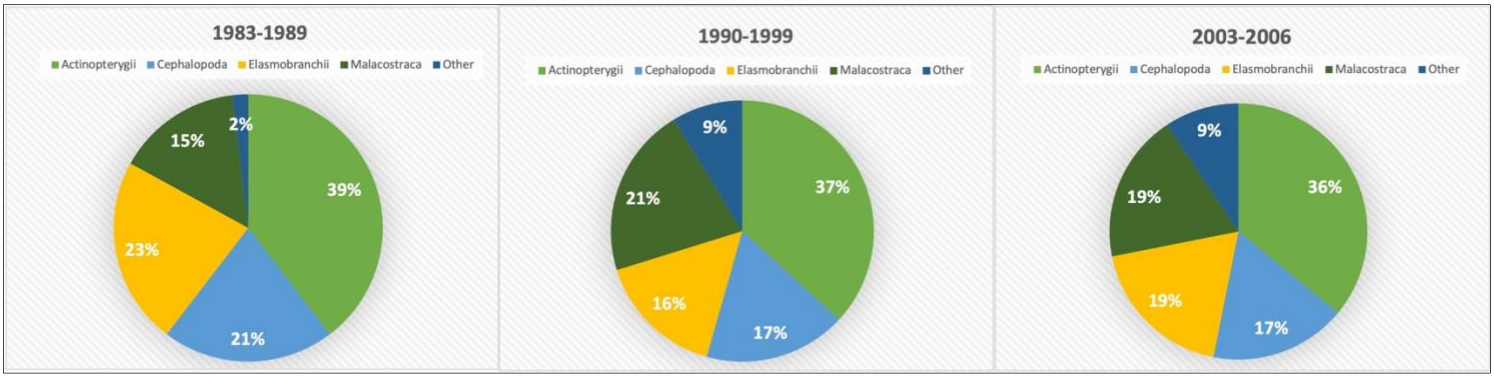


Figure 7*. Temporal changes in % contributed by each class recorded in data sourced from OBIS over three consecutive decades from 1983 – 2006 for ML3 (concessions 2c – 5c) on the Northwest coast of South Africa.*

3.1.2.2 Field Guide Dataset

ML3 hosted a diverse community of epifaunal invertebrates belonging to 17 class groups and 61 families. Community composition was composed primarily of six class groups (Anthozoa, Demospongiae, Echinoidea, Cephalopoda, Malacostraca, Asteroidea and Gastropoda). Of the six classes, Cephalopoda (19%), Malacostraca (18%), Asteroidea (17%) and Gastropoda (16%) were the four major contributors to community composition.

3.1.2.3 De Beers Monitoring Dataset

Of the 54 species of benthic macrofauna recorded in the northern inner shelf of ML3, a total of 8 classes and 34 families were identified: Malacostraca, Polychaeta, Gastropoda, Clitellata, Bivalvia, Holothuroidea, Hexanauplia, Anthozoa and five taxa whose class was unknown (Fig 7). Community composition was heavily dominated by Malacostraca in all four years sampled (2003, 2008, 2010 and 2011), most noticeably in 2003 where almost three quarters (74%) of the community consisted of crustacean species. Prior to the initiation of mining in 2003, no Polychaetes were identified in samples. After initiation of mining, Polychaete species played a substantial role in the composition of the infaunal community within the ML3 area, with the number of species displaying a slight increase over a 3-year period (2008 – 2011).

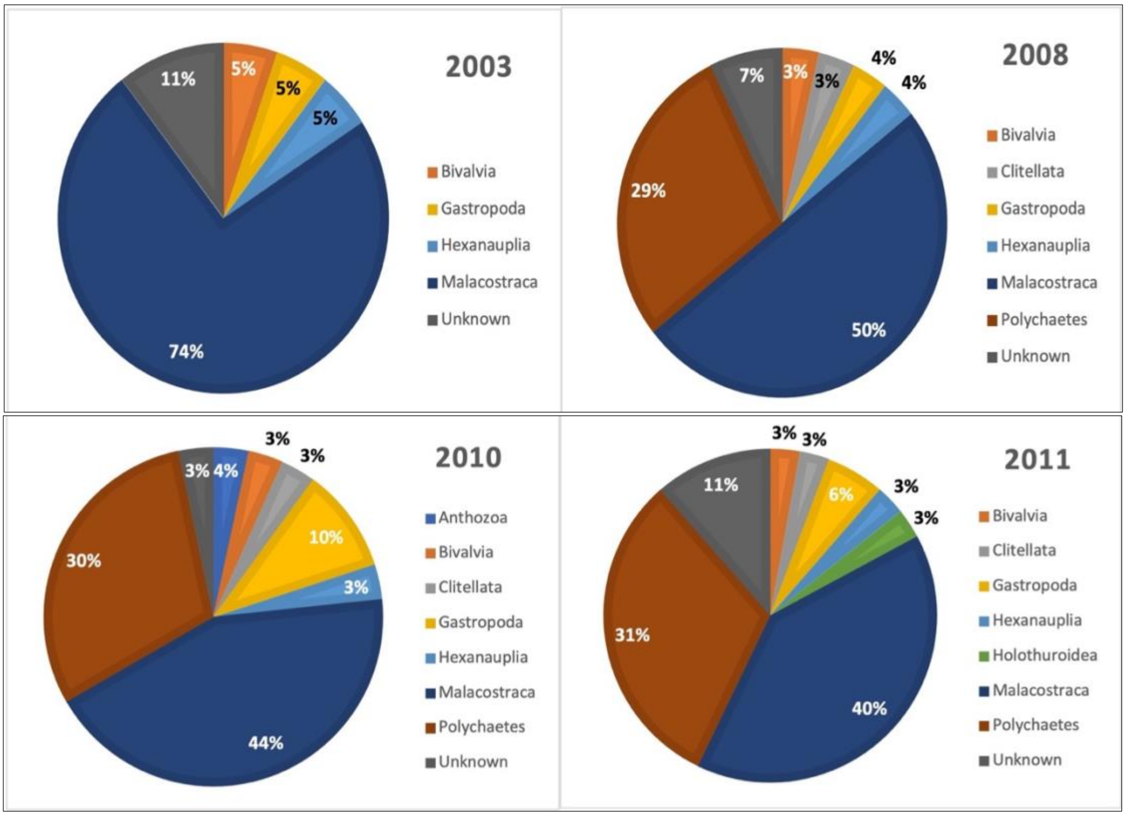


Figure 8. *Temporal changes in class contribution in northern ML3 sites over four benthic macrofaunal surveys carried out in 2003, 2008, 2010 and 2011. Shown as % class contribution for each year, combining mined and control sites.*

3.1.3 Diversity Indices

3.1.3.1 OBIS Dataset

Diversity increased considerably in ML3 between 1983 and 2006, from a maximum value of 36 species in the 1980s to 52 species in the 2000s, with the increase occurring almost entirely in the 2000s (Fig 8). Average diversity remained markedly low during the 1980s (25.1 species) and 1990s (26.4 species), rising by 16.4 species in the 2000s (42.8 species). Wide spread of annual diversity indicated high inter-decadal variability in the 1980s and 1990s.

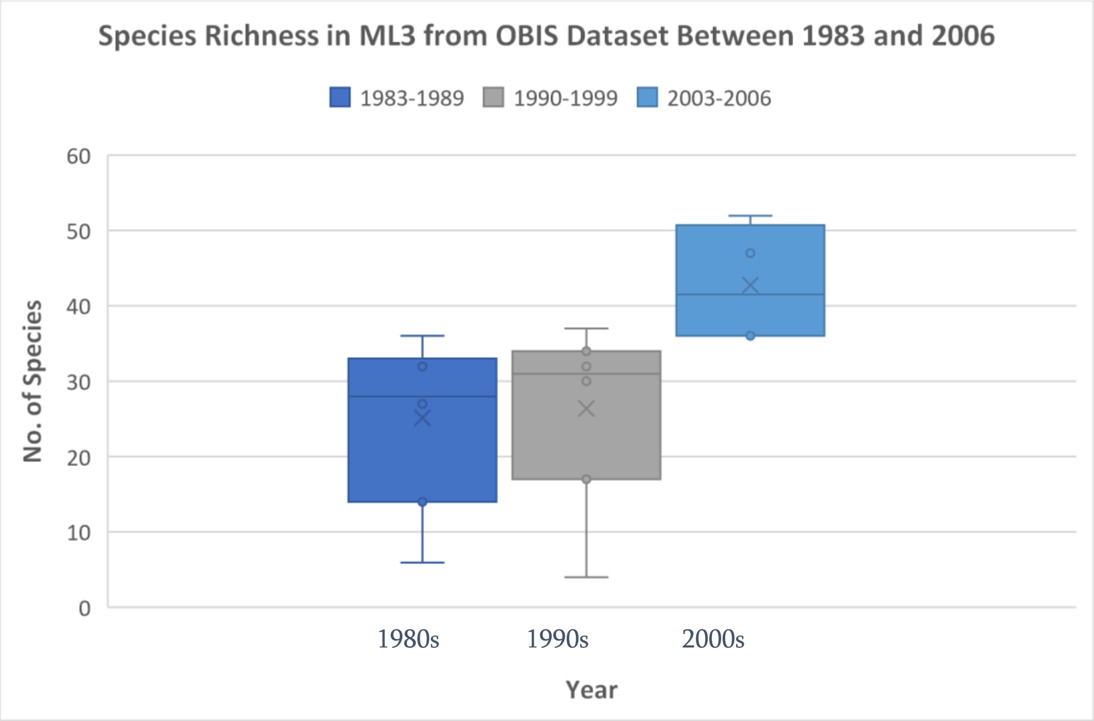


Figure 9. *Species richness in ML3 across three decades (1983 to 2006) using data obtained from OBIS.*

3.1.3.2 Field Guide Dataset

Total species richness in ML3 was relatively high with 84 epifaunal invertebrate taxa recorded. Cephalopoda and Malacostraca were the most species-rich taxa with 16 and 15 species respectively. Taxa from Ascidiacea, Bivalvia, Gymnolaemata, Hexanauplia, Hydrozoa, Polychaeta, Sipunculidea, Scyphozoa were each represented by a single species. Two potential Vulnerable Marine Ecosystem (VME) species were identified in ML3, namely, the sea sponge *Suberites dandelenae* and large sea pen *Anthoptilum grandiflorum*.

3.1.3.3 De Beers Dataset

Diversity of benthic infauna was remarkably low in 2003 Polychaetes were absent from all sites sampled in 2003. In the post-mining years (2008 – 2011), diversity increased gradually in both mined and control sites, from the first post-mining monitoring survey in 2008 (6 months after the cessation of mining) to the final survey in 2011. Diversity increased less over time in mined sites (H'= 0.52) compared with control sites (H'=1.19).

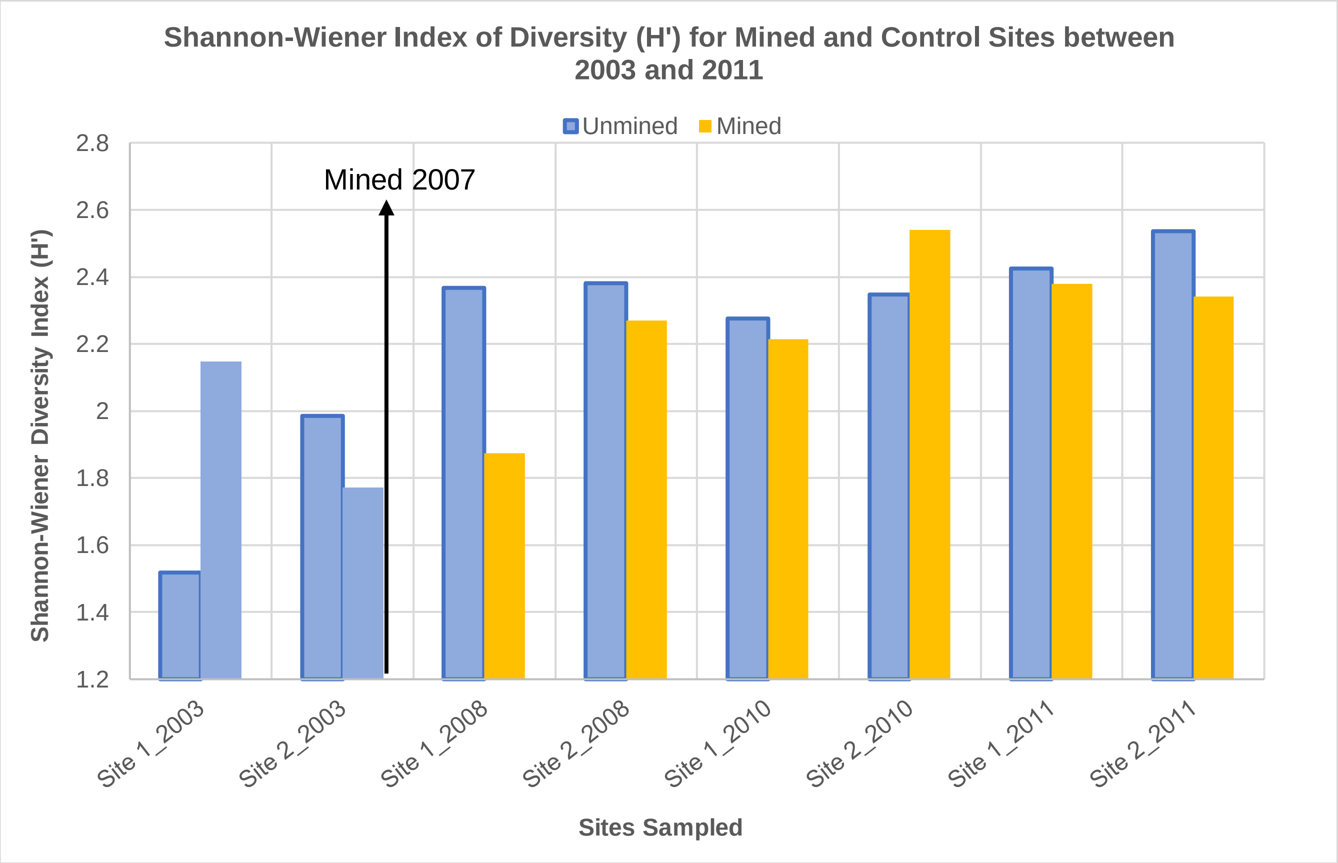


Figure 10. *Shannon-wiener (H') index of diversity for mined (NI-M1 and NI-M2) and control (NI-C1 and NI-C2) sites for each year of the four years surveyed (2003, 2008, 2010 and 2011). The inner shelf region was mined by De Beers for diamonds in 2007, depicted by an arrow on the figure.*

3.1.4 Multivariate Analysis

3.1.4.1 OBIS Dataset

MDS and Cluster analyses (Fig 5) show a fairly similar (~40% similarity) community assemblage between most years in ML3 from 1983 – 2006, excluding years 1983 and 1984 where lower species richness was reported. 1983 and 1984 were distinctly separated from the rest of the cluster owing to lower species contribution (6 and 14 species respectively). Sampling effort, in this case, annual number of trawls and indeed species richness was markedly varied between years, with no surveys carried out between 1997 and 2003. During this period only four species (*Aristias symbioticus, Laetmatophilus purus, Sphaeronella* sp., *Trischizostoma remipes)* were recorded (1999, not shown) in historical records held by the South African Museum (Iziko) at depths between 144m and 156m below sea level. 1999 was therefore excluded from Cluster and MDS analyses due to absolute dissimilarity with the other data points.

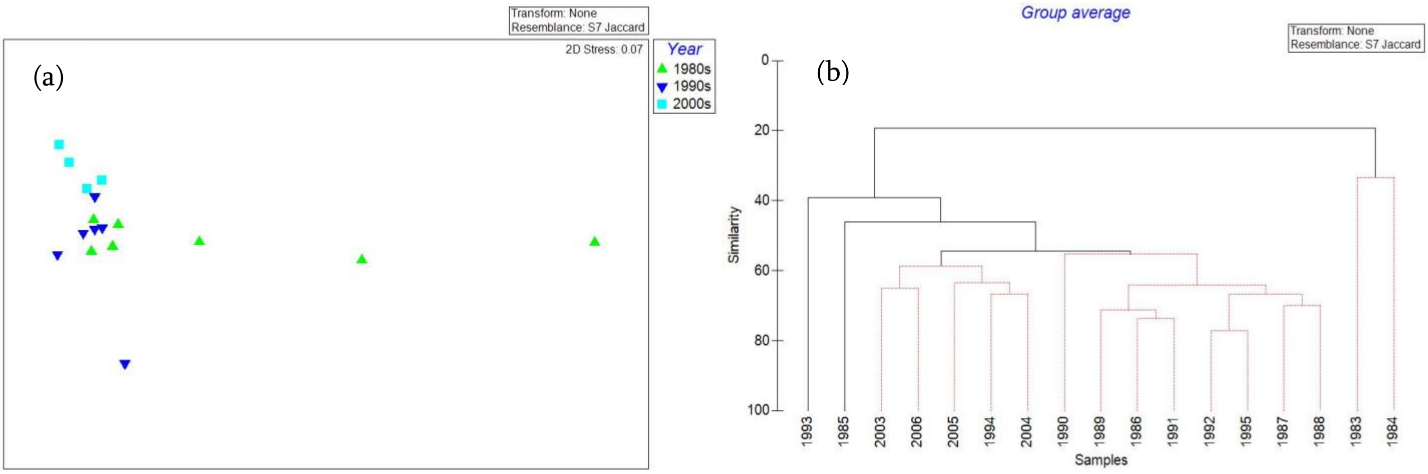


Figure 11. *(a) Subset Multidimensional Scaling (2d) ordination of ML3 data, based on Jaccard resemblance measure. Data points shown by decade. Outlier year (1999) not included in analyses. (b) Cluster dendrogram, with group average sorting, after Jaccard measure was applied.*

3.1.4.2 De Beers Monitoring Dataset

Pre-mining samples (2003) were distinctly separated from post-mining samples (2008, 2010 and 2011, Fig 12 and 13). Within the pre-mining cluster, all four sites sampled in 2003 were significantly similar (approximately 62%) in species contribution. Excluding one mined site in 2008 (NI-M1), all other post-mining samples indicated around 50% similarity in species contribution, with significant clustering between both mined sites (NI-M1 and NI-M2) in 2008 and 2010. There was approximately 70% similarity in communities in both mined sites and one of the control sites (NI-C2) in 2011. This was confirmed by the relative distance of the sites to the control sites in the MDS ordination (b, Fig 12). Site NI-M1 in 2008 is noticeably separated from other post-mining samples.

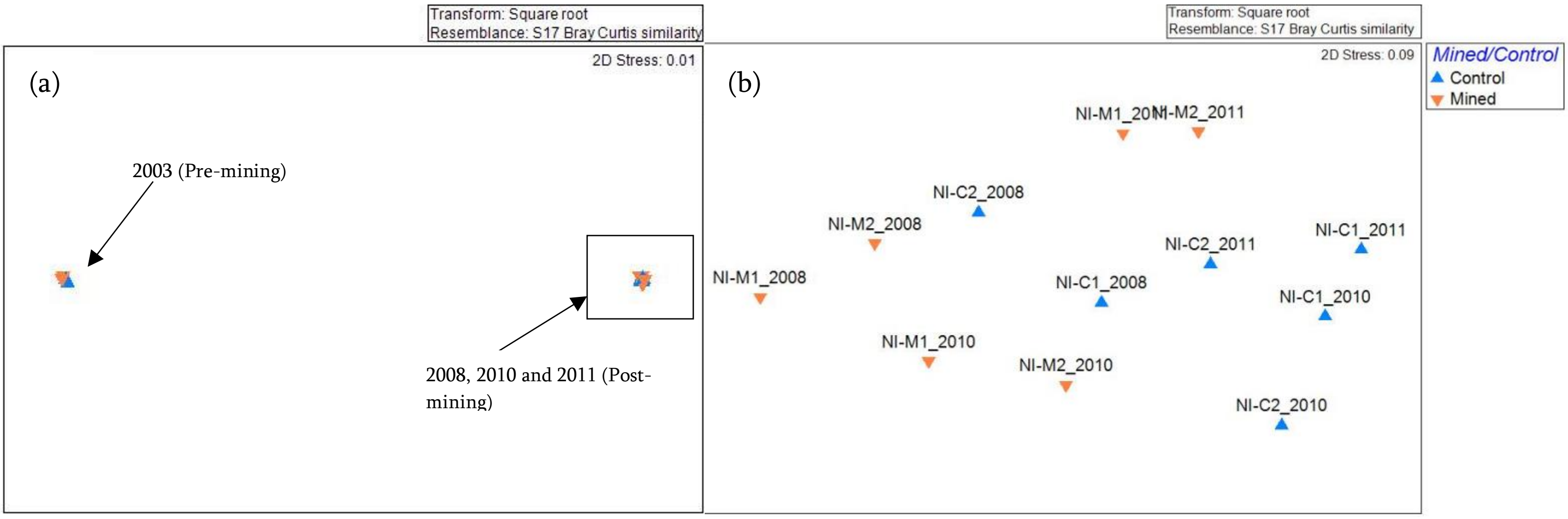


Figure 12. *Multidimensional Scaling ordination of benthic grab samples (n=16) collected for the De Beers Benthic Monitoring Programme. (A) On the left, all four samples from 2003 are clustered together. On the right, the 12 samples from 2008, 2010, 2011 are clustered together. (B) Subset of the cluster on the right-hand side of (a). Ordinations were based on Bray Curtis similarity measure after square-root transformation and are shown by treatment group (mined/ control).*

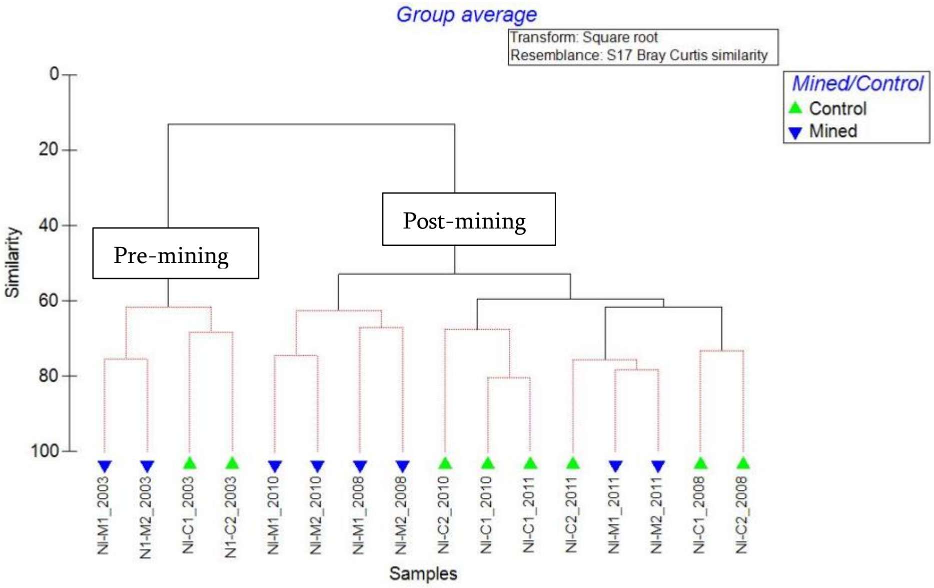


Figure 13. *Cluster dendrogram with Similarity Profile (SIMPROF) analysis of samples collected from mined (M1, M2) and control site (C1, C2) during surveys in 2003, 2008, 2010 and 2011. Based on Bray Curtis similarity, after square-root transformation and group average sorting.*

SIMPER analysis revealed species assemblage across the mined and control group to be highly similar (Table 1). *Calocaris barnardi*, Copepoda and *Paraphoxus oculatus* were the three most numerically dominant taxa in both groups, though average abundance was considerably lower in the mined group (Table 2). Seven taxa were more abundant in mined sites, with several occurring exclusively in mined sites, including polychaete *Cossura coasta (Av. Abundance =*

0.58) and crustacean Cumacea *(Av. Abundance = 0.28).* Copepods contributed most to dissimilarity between mined and unmined sites, with slightly greater average abundance observed in mined sites (4.71). *Callianassa australis* most consistently contributed to the dissimilarity (6.56%) between groups with an average abundance more than six times lower in mined sites. Bivalve *Moerella tulipa* (6.36%), peracarid crustacean *Paraphoxus oculatus* (5.98%), whelk *Nassarius vinctus* (5.17%) were also main contributors to dissimilarity with an average abundance more than two times lower in mined sites.

Table 1. *SIMPER analysis showing common infaunal species from mined and control sites. Similarity between species assemblage between mined sites was 73.73% and 71.76% between control sites. The 10 most abundant macrofauna taxa are shown in order of abundance for Mined and Control groups.*

Group Mined	**Group Control**
73.73% Similarity	71.76% Similarity
Calocaris barnardi	*Calocaris barnardi*
Copepoda	*Copepoda*
Paraphoxus oculatus	*Paraphoxus oculatus*
Nephtys hombergii	*Moerella tulipa*
Themisto gaudichaudii	*Themisto gaudichaudii*
Paraprionospio pinnata	*Callianassa australis*
Nassarius vinctus	*Nephtys hombergii*
Pterygosquilla capensis	*Leptochela*
Leptochela	*Euphausia lucens*
Mediomastus capensis	*Nassarius vinctus*

Table 2. *Two-way Similarity Profile (SIMPER) analysis comparing average square-rooted abundance of macrofauna taxa in mined and control groups across all years. Top 10 species contributing to 90% dissimilarity' are shown below.*

Species	*Control Av.Abund*	*Mined Av.Abund*	*Av.Diss*	*Diss/SD*	*Contrib%*	*Cum.%*
Copepoda	4.46	4.71	3.01	0.92	7.73	7.73
Callianassa australis	**3.76**	0.53	2.55	1.52	6.56	14.29
Moerella tulipa	**3.85**	0.82	2.47	1.14	6.36	20.65
Paraphoxus oculatus	**5.04**	2.72	2.33	1.58	5.98	26.63
Nassarius vinctus	**5.56**	2.31	2.01	0.61	5.17	31.80

Sigambra parva	**2.71**	0.35	1.81	1.54	4.65	36.45
Euphausia lucens	**3.48**	1.86	1.69	1.03	4.35	40.80
Pterygosquilla armata	**1.24**	0.53	1.66	0.47	4.26	45.06
Calocaris barnardi	**6.41**	4.31	1.44	0.85	3.71	48.76
Ampelisca anomala	**1.54**	1.19	1.37	1.18	3.53	52.29

Pre- and post-mining samples were extensively dispersed from one another (Figure 12a), while post-mining samples demonstrated greater clustering (Figure 12b). Within the post-mined group, mined and control samples tended to cluster in their respective groups, aside from 2011 that showed greater similarity with control samples (Figure 12b). PERMDISP confirmed that dispersion between treatment groups was significantly different ($F = 1.139$, $p \leq 0.462$), with much of this difference appearing to stem from the differences between pre-mining (2003) and post-mining (2008-2011) samples. Of the factors measured, PERMANOVA analysis revealed that all three factors measured were statistically significant (<0.05) in explaining variation within the data. Differences in macrofaunal abundance were driven primarily by variation between years ($p= 0.0001$), treatment, i.e., whether a site was mined or not, ($p = 0.019$) and to a lesser extent, the interaction between the two ($p = 0.0405$). Pairwise PERMANOVA revealed that the composition of benthic species was significantly different between mined and unmined sites ($p = 0.02$ $t = 2.196$) across all year groups (Table 4).

Table 3. *Two-way crossed PERMANOVA on fixed factors 'treatments' (Tr) and 'year' (Ye). Degrees of freedom (df), sum of squares (SS), mean sum of squares (MS), pseudo-F, p – value (P(perm) and unique permutations out of 9999 permutations performed.*

Source	***df***	***SS***	***MS***	***Pseudo-F***	***P(perm)***	***Unique perms***
Mi	1	1821	1821	4.8241	0.019	9950
Ye	3	23017	7672.4	20.326	0.0001	9937
MixYe	3	2933.6	977.88	2.5906	0.0405	9945
Res	8	3019.8	377.47			
Total	15	30792				

Table 4. *PERMANOVA pairwise analysis for term 'year' for pairs of year groups. Group pairing, pseudo t, P (perm) and number of unique permutations shown.*

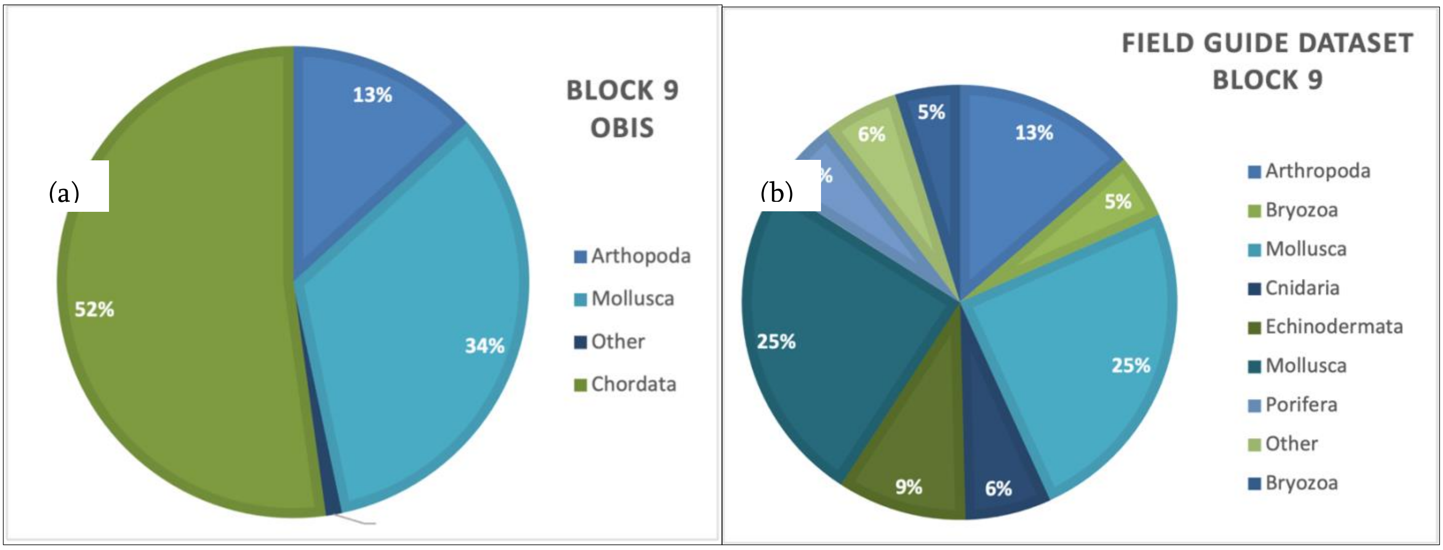
Groups	***t***	***P(perm)***	***perms***

2003, 2008	5.2796	0.0037	315
2003, 2010	6.0056	0.0031	315
2003, 2011	6.3358	0.0027	314
2008, 2010	2.2714	0.0069	315
2008, 2011	2.7364	0.0023	315
2010, 2011	2.0951	0.0122	315

3.2 Block 9

3.2.1 Taxonomic Composition (Phylum)

A total of 302 taxa belonging to 12 phyla, 24 classes and 192 families were extracted for Block 9. Most taxa (299) were identified to species or genus level, with 19 identified to a higher taxonomic level and 7 unidentified taxa. Phyla recorded in Block 9 included Chordata, Mollusca, Arthropoda, Echinodermata, Cnidaria, Annelida, Porifera, Bryozoa, Rhodophyta, Sipuncula, Hemichordata and Brachiopoda. The latter four phyla were represented by a singular species. Chordata (32%) Mollusca (20%) and Arthropoda (8%) were the top represented taxa, accounting for 60% of species richness in Block 9. A total of 20 species were encountered in at least two datasets, with no species in all three. OBIS was the most diverse dataset, with 184 taxa extracted, more than half of which were Ichthyofauna (52%) and Mollusca (34%). Ninety-two invertebrate taxa were extracted from the Field Guide and 54 biofouling taxa (including six unidentified taxa) were extracted from data collated in Cook (1995-1999).



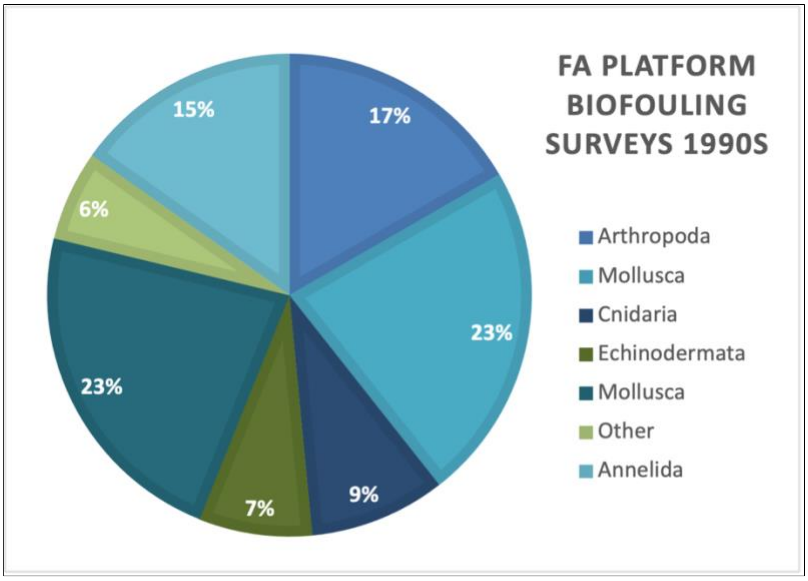


Fig 14 *Phylum percentage contribution for all three datasets in Block 9, based on number of species. (a) OBIS data 1984 – 2006 (b) Field Guide data 2011 - 2018 (c) Cook Data 1995-1999. Group 'other' pools phyla with < 4 species.*

3.2.2 Taxonomic Composition (Class)

3.2.2.1 OBIS Dataset

From the 184 taxa extracted from OBIS, a total of 10 classes and 112 families were identified in Block 9, including, Actinopterygii, Malacostraca, Elasmobranc (c) nd Cephalopoda Hexanauplia, Gastropoda, Bivalvia, Echinoidea, Pycnogonida and Scyphozoa. Ecosystems on, or near to, the seabed were constructed of four main classes, namely, Actinopterygii, Malacostraca, Elasmobranchii and Cephalopoda (Fig 15). Fishes (Elasmobranchii, Actinopterygii and Holocephali) were the most diverse group in Block 9 throughout the 1980s, 1990s and 2000s, increasing by 25 species over the duration studied. The critically endangered soupfin shark (*Galeorhinus galeus*) and several endangered demersal fish species, including the puffadder shy shark *Haploblepharus edwardsii*, white skate *Rostroraja alba* and the shortspine spurdog *Squalus mitsukurii* were encountered across all three decades.

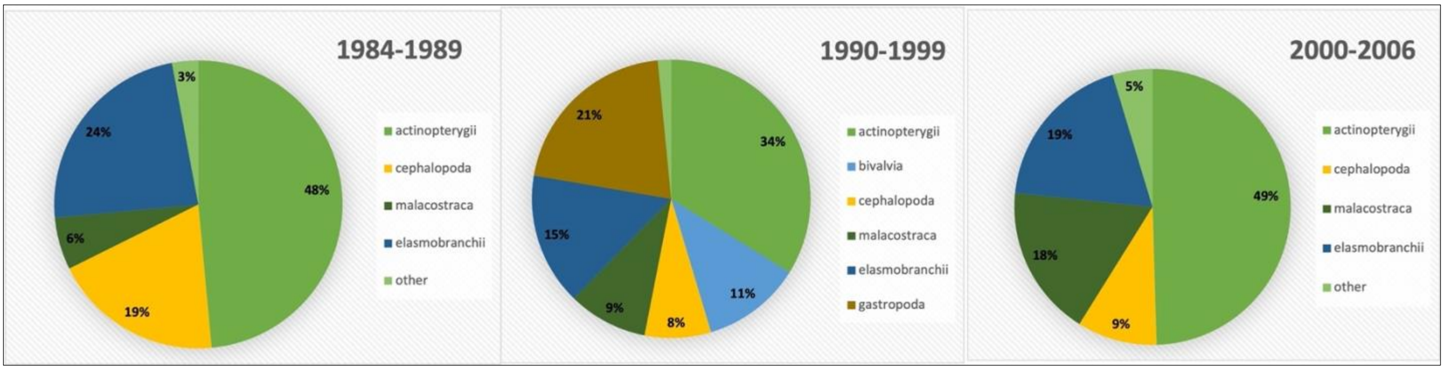


Fig 15. *Temporal changes to % contributed by each class recorded in data sourced from OBIS over three consecutive decades from 1983 – 2006 for Block 9 on the Agulhas Bank, South Africa.*

3.2.2.2 Field Guide Dataset

Community composition in Block 9 was highly diverse, with epifaunal invertebrates from a total of 19 classes and 62 families. Malacostraca, Gastropoda, Cephalopoda, Demonspongiae and Asteroidea were the six most diverse classes encountered within Block 9. From this, Malacostraca (17%), Gastropoda (%) and Cephalopoda (16%) were the greatest contributors to community composition. Almost half of the classes identified in Block 9 were represented by a single species, namely, Graptolithoidea, Bivalvia, Hexanauplia, Hydrozoa, Polychaeta, Rhynchonellata and Scaphopoda.

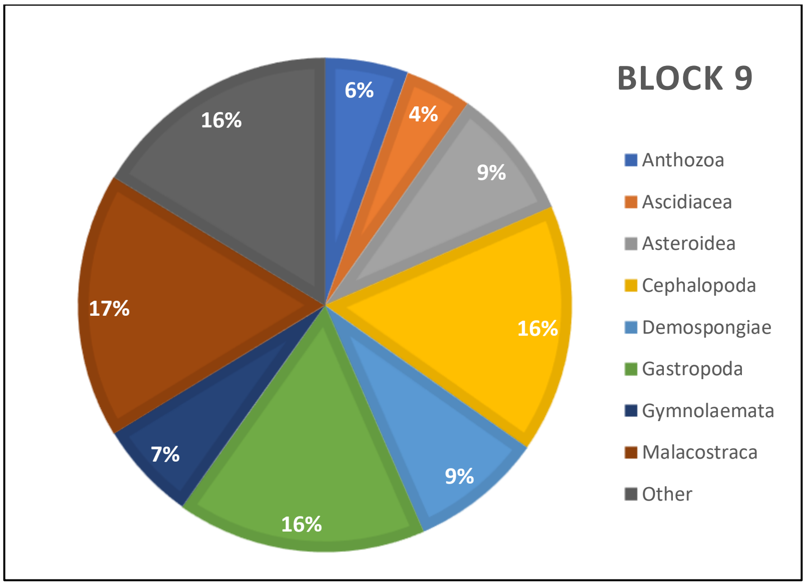


Figure 16. *Class contribution for invertebrate species captured in DAFF research trawl surveys and historic records for Block 9 on the South coast of South Africa. Group 'Other' pools species from classes with species richness of < 3.*

3.2.2.3 Cook Dataset

Invertebrates belonging to 11 classes and 33 families were identified as part of the biofouling community colonising the FA platform. These classes included Anthozoa, Bivalvia, Echinoidea, Florideophyceae, Gastropoda, Hexanauplia, Hydrozoa, Malacostraca, Ophiuroidea, Polychaeta and Sipunculidea. Of these 11 classes, Florideaophyceae, Polychaeta and Sipunculidea were characterised by a singular species. Bivalves were the most predominant species encountered on the platform across the four surveys carried out in the 1990s, followed by polychaetes and crustaceans. Interestingly, the invasive Mediterranean mussel *Mytilus galloprovincialis* was identified on the platform in just one survey (1996), after which time no further individuals were identified . In addition *to* the Mediterranean mussel, a number of species occurring predominantly in the intertidal zone, including the brown mussel *Perna perna*, native black mussel *Choromytilus meridionalis*, oysters and urchins were also identified across the depth range studied.

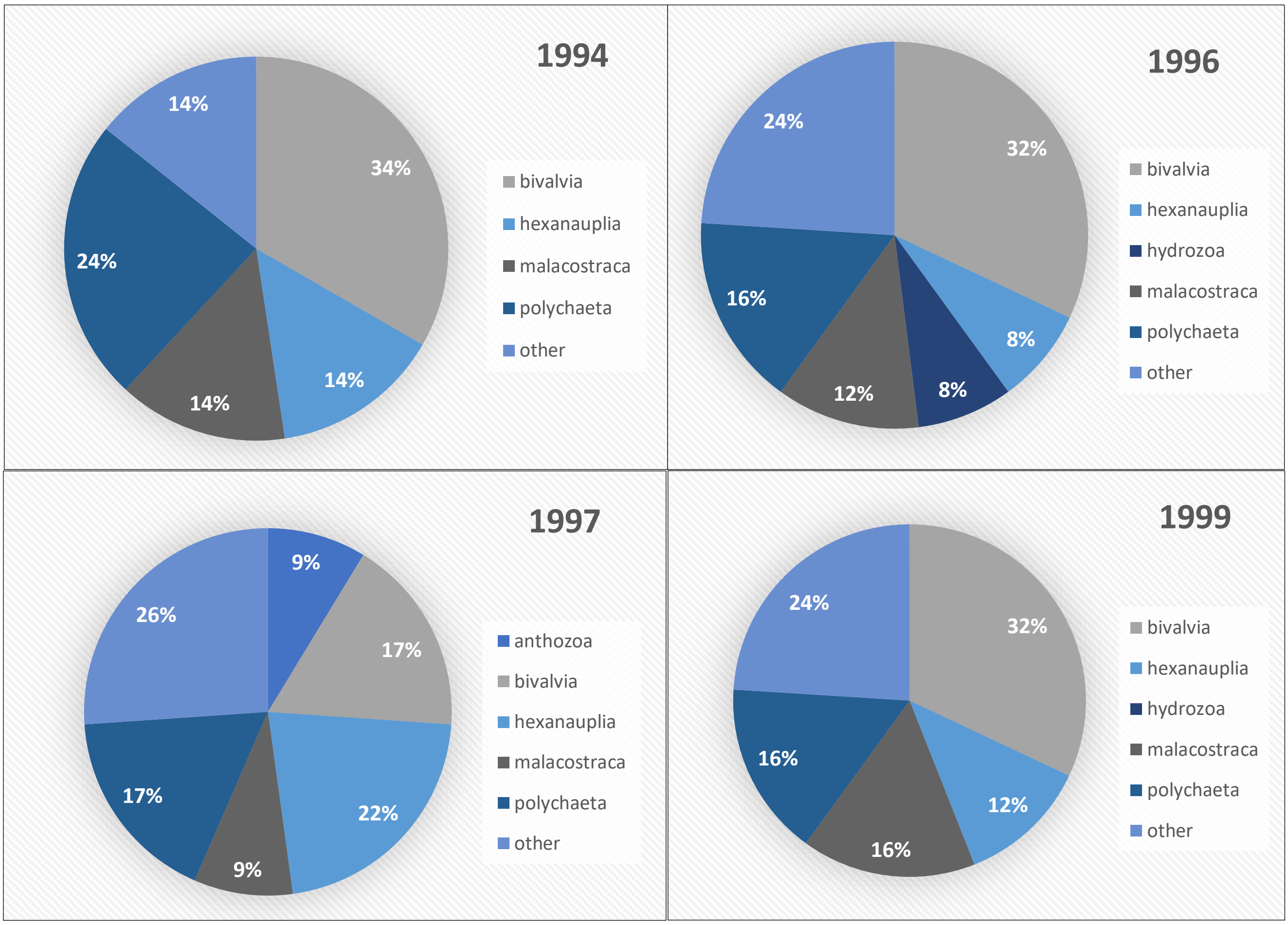


Fig 17. Time-series showing community composition of the FA platform biofouling community between 1994-1999. Data was collected by researchers at the University of Cape Town (Cook 1995-1999).

3.2.3 Diversity Indices

3.2.3.1 OBIS Dataset

A temporal increase in diversity was observed in Block 9, with a wide range of species encountered per decade. The average number of species rose from 29.2 to 51.2 species, with the majority of the increase occurring between 1980 and 1999 (Fig 18). Species diversity was highest in Block 9 during 1994 (80 taxa) and 2006 (86 taxa) respectively.

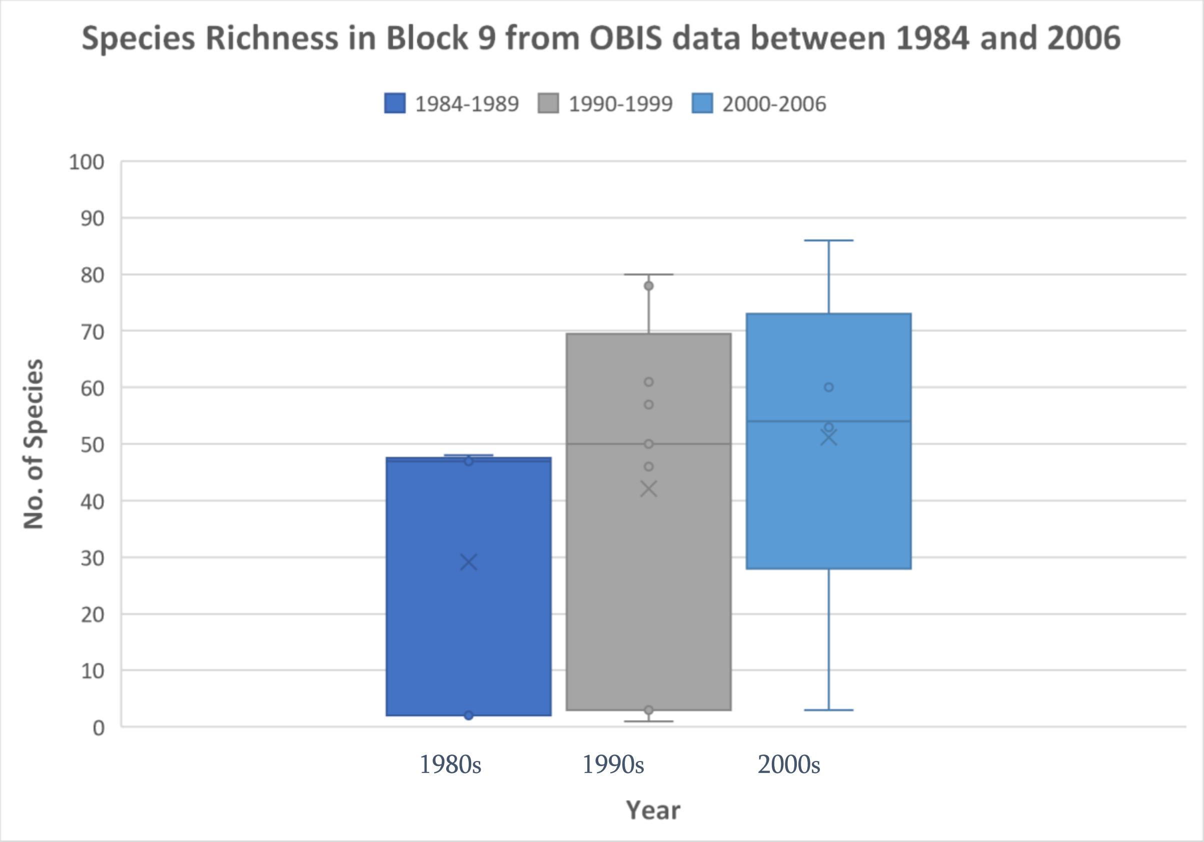


***Figure 18. Average** species richness in Block 9 between 1984 and 2006. Shown with maximum and minimum range using data obtained from OBIS.*

3.2.3.2 Field Guide Dataset

A total of 92 epifauna invertebrates belonging to 19 classes were identified in Block 9. Malacostraca and Cephalopoda were the most speciose class groups with 17 and 16 species respectively. Several potential Vulnerable Marine Ecosystem (VME) species, as identified by Atkinson and Sink (2018), were sampled in Block 9. These taxa included several fragile corals: bamboo coral, Honeycomb false lace coral and right-angled corals, endemic sabre bryozoan and relatively unknown agar animal (Atkinson 2018 in Atkinson and Sink 2018)

3.2.3.3 Cook Dataset

A total of 38 taxa were identified to species level, 12 to genus level and 7 taxa were unidentified. Diversity on the FA platform remained relatively low, increasing by just four species over the years sampled. Species richness did not increase linearly (Fig 19) however, with the increase occurring predominantly between 1994 and 1996 (>4 species) with a dip in diversity in 1997 (<2 species).

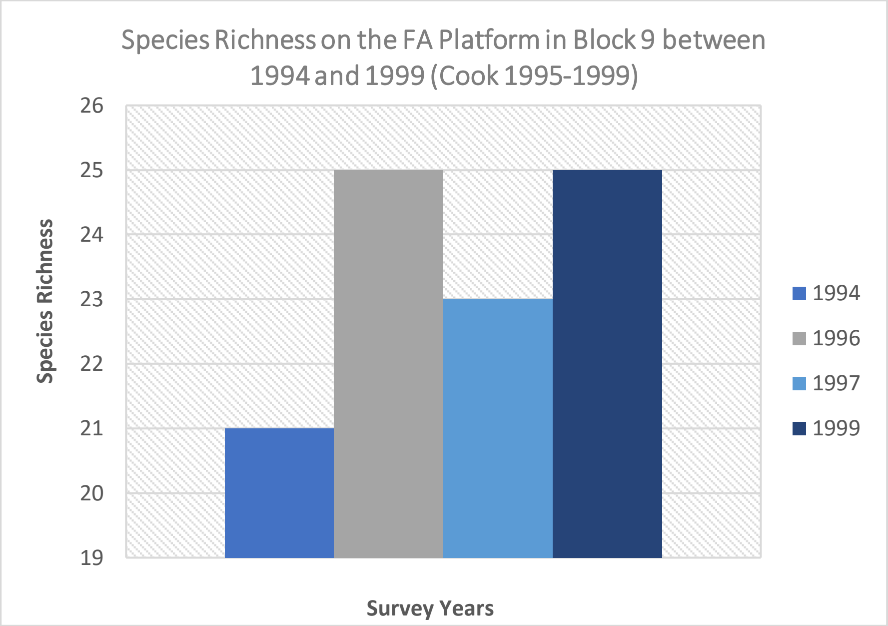


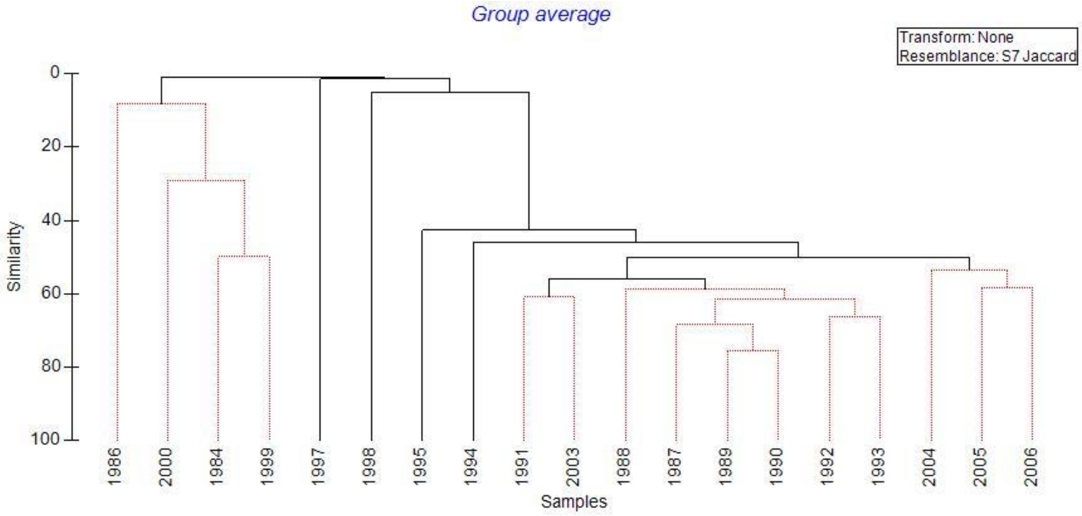
Figure 19. *Species richness for communities inhabiting the FA platform located in concession Block 9. Historical data was collected during fouling surveys carried out by University of Cape Town staff from 1992 – 1999 (Cook 1995, 1996, 1997, 1999) and sourced from Appendix 1 of Sink et al (2010).*

3.2.4 Multivariate Analysis

3.2.4.1 OBIS Dataset

Cluster analysis found approximately 42% similarity in species composition between 13 of the 19 years for which data is available from OBIS, with eight further significant clusters within the group. MDS ordination (b and c, Fig 20) showed decadal structuring of year groups along the z axis, from least to most diverse, indicating increasing diversity and similarity in species assemblage from the 1980s to the 2000s. Significant clustering was identified between 1987, 1988, 1989, 1990, and 1992 and 1993 with around 60% similarity in species composition. Within the cluster, communities in 1989 (47 species) and 1990 (46 species) showed the greatest similarity (around 75%) in species composition. Significant clustering was also identified between years 2004, 2005 and 2006 with around 55% similarity in community assemblage. Years 1986, 2000, 1984 and 1989 were clustered at approximately 10% similarity.

(a)



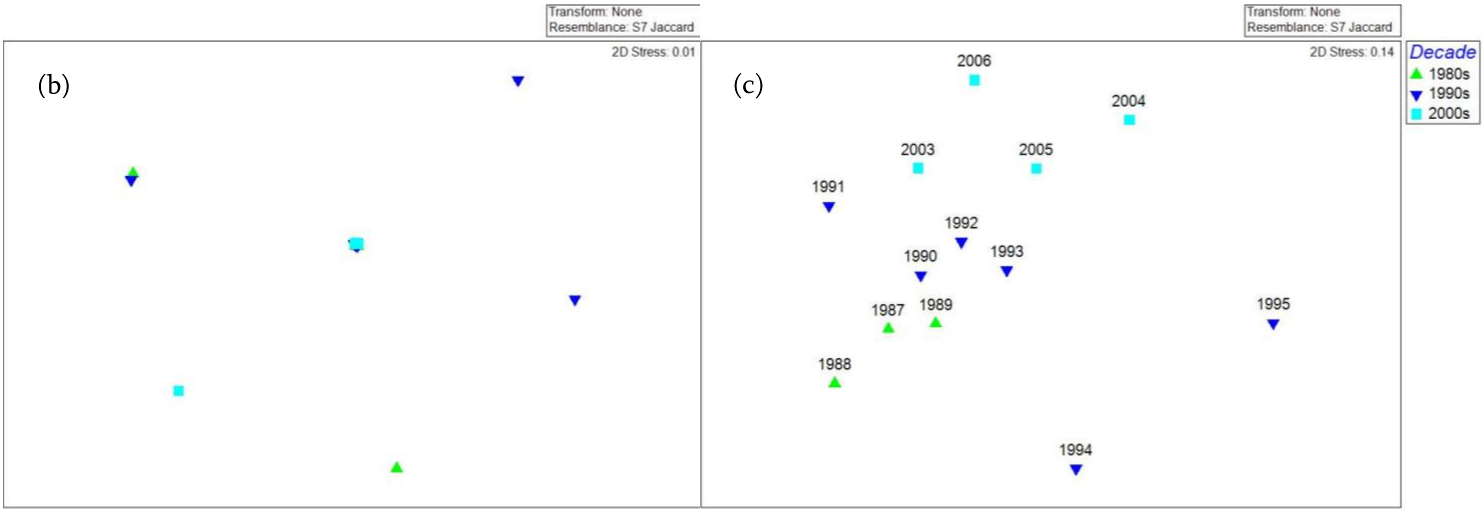


Figure 20. *(a) Cluster dendrogram with similarity profile (SIMPROF) analysis for each sample year (1984-2006) obtained from OBIS, after group sorting, based on Jaccard similarity measure, (b) MDS ordination of OBIS data points based on Jaccard similarity measure, (c) Subset MDS of central cluster seen in (b).*

3.2.4.2 Cook Dataset

Biofouling communities in all years showed around 20% similarity in species composition, with year groups delineated into two significant clusters (2 years per cluster). Interestingly, years 1994 and 1999 were the most similar in species composition (approximately 50% similarity) despite their chronological distance, whilst 1996 and 1997 were marginally less similar in species composition (~45%) despite surveys being taken approximately one year apart.

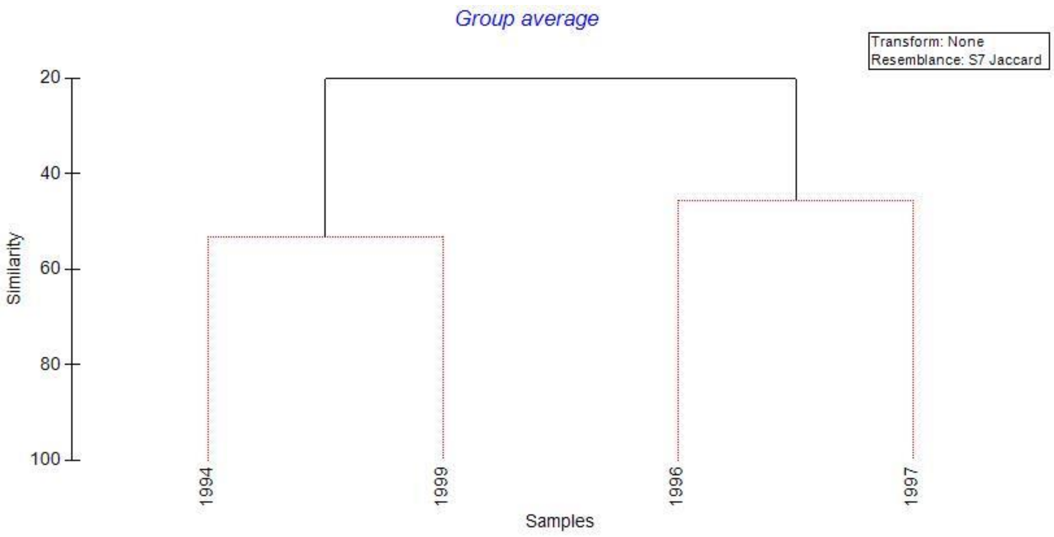


Fig 21. *Cluster dendrogram, based on the number of species observed on the FA platform between 1992 and 1999. Dendrogram was generated by group average sorting, based on Jaccard similarity measure.*

4 Discussion

Access to ecological data is increasingly imperative in the current climate, as our oceans and the diverse ecosystems they support are subjected to increasing anthropogenic stressors (Caro et al. 2018, Kirkman et al. 2013, Singh et al. 2017). With growing interest in extracting materials at greater depths than ever before (Miller et al. 2018b), it is crucial that the biota (biodiversity and relevant environmental variables) be characterised prior to the introduction of disturbance. Baseline data facilitate greater understanding of the scale at which human-mediated stressors impact marine ecosystems, particularly in understudied ecosystems (i.e., deep-sea benthos, Miller et al. 2018b). Utilising open-source data, this study was able to characterise and quantify the biodiversity of benthic and demersal habitats, using two concessions subject to significant diamond (west coast) and hydrocarbon (south coast) extraction as case studies. As a first step, open-source data were found to be a decidedly useful and cost-effective method for characterising the biota of a particular area. Access to long-term data, as evidenced here with DFFE trawl data from OBIS, was useful for tracking historical changes in biodiversity. However, when evaluating the impacts imposed by mining, we found the utility of open-source data to be limited. Studies designed to consider the particular impacts of different types of mining (i.e., BACI design employed in De Beers) utilised here were the only datasets that allowed for known biodiversity impacts (e.g., reduced diversity, invasive species) of mining to be evaluated. The challenges and limitations of publicly available data have been acknowledged and will be discussed at length in subsequent sections.

4.1 Characterising the biota of concession areas

Based on openly-available information, it was possible to characterise the biota qualitatively (i.e., at multiple taxonomic levels) and quantitatively (i.e., total species richness, time-series). Here, the benthic biota on both coasts was strongly characterised by invertebrates and ichthyofauna because most of the data were contributed from demersal trawl stock assessments carried out by DFFE. Block 9 (south coast) contained considerably more species than ML3 (west coast), compatible with higher diversity on the south coast, as reported in previous studies (Awad et al. 2002, Shah 2018). Chordates (fishes), arthropods (crustaceans) and molluscs (cephalopods, gastropods and bivalves) were the most speciose phyla, accounting for the majority of total species richness in ML3 and Block 9, similar to findings in past biodiversity reviews (Costello et al. 2010, Griffiths et al. 2010). Similarly, in Griffiths et al. (2010), molluscs, arthropods and pisces contributed to 68% of the total number of species, consistent with findings in ML3 in particular (68.3%), although group contribution was notably different. The most diverse taxa encountered in each concession differed, between crustaceans (Malacostraca) on the west coast and fishes (predominately Actinopterygii and Elasmobranchii) on the south coast, with each coast being substantially important to breeding, spawning and as a nursey ground for juvenile species (Sink et al. 2010).

South Africa's marine environments are strongly influenced by the oceanography, topography and sediment type of the surrounding oceans (Sink et al. 2019). Both concessions are situated within distinctive ecoregions, as defined by the latest National Biodiversity Assessment (Sink et al. 2019), demarcated by differences in several variables, including species distribution, nutrients, productivity and temperature (Sink et al. 2019). On the west coast, diversity was markedly lower within the cool temperate Southern Benguela ecoregion, the most productive of South Africa's ecoregions (Sink et al. 2019). ML3 is further defined as the Namaqua region where upwelling is consistent and powerful, with less variable temperature fluctuations than further southward (Sink et al. 2019). As expected from previous studies (Awad et al. 2002, Shah 2018), the south coast supported many more species. Block 9 is situated within the warm temperate Agulhas ecoregion, an area of ecological and economical significance where the wide continental shelf supports areas of conservation importance that include reef complexes, banks, and nursery and spawning grounds (Hutchings et al. 2009, Sink et al. 2010, Sink et al. 2019). Fine-scale differences that exist between areas targeted for mining highlight the need for in-situ sampling, tailored mitigation measures and long-term monitoring to evaluate the true scale of mining impacts on the diverse marine ecosystems found off South Africa.

'Hotspots' typically refer to concentrated areas of biological importance as a result of exceptional biodiversity, productivity or importance in essential processes, e.g., for recruitment or feeding (Kirkman et al. 2013, Palacios et al. 2006, Reese and Brodeur 2006). Of the seafloor fauna considered in this study, no benthic or demersal hotspots were found to directly overlap with the concessions considered here. Benthic diversity was greatest on South Africa's east coast inshore region (Yemane et al. 2020). Whilst this may reflect limited sampling of the benthos within the offshore region, the greatest number of benthic samples have been collected on the west coast (Griffiths et al. 2010), suggesting diversity is genuinely low on the northwest coast, in line with current knowledge of upwelling regions such as the Benguela. Utilising OBIS data to generate biodiversity maps, Yemane et al. (2020) identified several hotspots for fish diversity within the South African EEZ. Hotspots were greatest at depths greater than 200m on the west coast and up to 200m depth on the mid-shelf region of the Agulhas Bank (Yemane et al. 2020). Within the southern Benguela region, a hotspot for demersal fishes and cephalopods was identified off the Northern Cape coast at depths of around 200m (Kirkman et al. 2013), exceeding the maximum depth within ML3. Although in theory, mobile species such as fish are able to move away from disturbance, connectivity of the ocean means that impacts such as noise are likely to persist over much larger distances (Popper and Hawkins 2019). If communities migrate away from disturbance, there may be greater ramifications for demersal fisheries, in particularly the highly-profitable hake fishery (Atkinson and Sink 2008). Of particular concern are the impacts on sedentary or relatively immobile species, particularly endemic species with limited range (Scott et al. 2012), some of which have been shown to overlap with hydrocarbon activity (Venegas-Li et al. 2019). Lack of geographical barriers in addition to the particular oceanography of a region, are crucial factors to consider when assessing the scale at which anthropogenic activity may impact the marine environment, particularly where vulnerable ecosystems are negatively affected.

4.2 Long-term studies to determine changes in diversity

A key advantage of large-scale data repositories such as OBIS is the access to long-term datasets that enable temporal patterns in biodiversity to be analysed, although interpretation is limited by the absence of abundance or biomass data, as well as differences in sampling size, methods, and study duration. Utilising long-term data available from OBIS, it was possible to identify the following patterns over time (1983 to 2006). On both coasts, benthic and demersal communities were composed primarily of the same four taxa throughout the decades considered, for example, ray-finned fishes (Actinopterygii), cartilaginous fishes, including rays and skates (Elasmobranchii), molluscs (Cephalopoda) and crustaceans (Malacostraca).

On the west coast, there were an increase in the number of species encountered across the decades, although this may be due to improvements in methods, reporting and sampling tools available in latter decades. Despite some of the limitations encountered with open-source data, for example, lack of metadata or understanding of objectives, the utility of such data types is highlighted in the ability to compare data with other scientific studies. For example, the demersal species were markedly similar to the characteristic species that form a distinct demersal assemblage within the southern Benguela, as identified in a study looking at community structure of demersal fishes along the west coast of South Africa (Mafwila 2017). Both species of cape hake*,* jacopever, monkfish and grenadier *Caelorinchus simorhynchus* were most abundant across stations on the west coast, with only one species (*Centroscymnus crepidater*) not encountered during DFFE surveys, possibly as the dogfish species typically inhabits deeper waters than are found in ML3. Similarity between assemblages revealed in Mafwila (2017) and the current study highlight the utility of open-source data in characterising the seafloor biota. However, it is also recognised that there is a preference towards the sampling of commercial species because of their monetary value within global economies.

On the south coast, however, increasing species richness was driven predominately by fishes, namely, ray-finned fishes (Actinopterygii) and to a lesser extent rays and skates (Elasmobranchii). The reason for this increase is not clear, though it is possible that a greater number of species were encountered over time, either by chance, due to increased abundance of less abundant species (Yemane et al. 2004), perhaps improved sampling equipment or increased sampling effort. Higher species richness in the 1990s and 2000s may be indicative of partial population recovery in a number of previously overexploited species known to have suffered intense fishing pressure in previous decades (e.g., linefish), resulting in depletion and even collapse of stocks (Blamey et al. 2015). It was reported in Blamey et al (2015) that the main data contributor, DFFE (formerly known as MCM), changed the type of trawl gear used during their interannual surveys at the turn of the 21st century. Use of alternate gear with a smaller mesh size or wider nets, for example, may be able to sample more fauna, producing a more accurate representation of biota living on and above the seafloor.

During the 1990s on the south coast, a substantial number of mollusc species (15 bivalves and 27 gastropods) were identified from sediment samples taken using dredging equipment on the Agulhas Bank. Gear type used by researchers to sample the benthic molluscs was entirely different from the (trawling) gear used by DFFE during demersal trawl surveys, from which the majority of available data originated. The use of different sampling equipment between the two datasets clearly highlights how different gear can be used to sample different components of the seafloor biota, explaining why the use of multiple gear types is preferable in enabling more of the biota to be sampled (Flannery and Przeslawski 2015). This was demonstrated in Sink et al. (2010) with the use of trap, line and diver sampling and ROV image analysis, to sample offshore benthic biodiversity on the Agulhas Bank.

Shallow-water hake *Merluccius capensis* were encountered consistently across all DFFE trawl surveys on both the west and south coast, which may suggest that *M. capensis* remained relatively abundant across the decades studied despite pressure from a profitable hake-targeted fishery. However, the frequency at which the species were encountered may also be an artefact of the trawling equipment used to sample the biota as hake are known to inhabit the soft-sand seabed where commercial trawlers target hake. On the west coast, deep-water hake *Merluccius paradoxus* were encountered just as frequently, in contrast to the south coast, where *M. paradoxus* encounters were sparse until the latter decade. The upper limit of *M. paradoxus* range is known to occur around 150m, beyond the extent of Block 9 on the Agulhas Bank, which may in part explain why they were seldom encountered, though this fails to explain why they were encountered frequently during the 2000s (Hutching et al. 2009). Furthermore, after spawning on the Agulhas Bank, larvae of Cape hake *spp* are typically transported onto the west coast by north-westerly flowing currents, which may also explain why hake *spp* were encountered more frequently on the west coast (Grote et al. 2015). Encounters with Cape hope squid *Loligo reynaudii,* known locally as 'Chokka' were frequent on the Agulhas Bank, where most of the biomass is concentrated. On the west coast however, encounters occurred intermittently between the mid 1980s and early 1990s (Roberts 2005). It is possible that lack of encounters with *L. reynaudii* on the west coast after the 1990s may have been due to suboptimal environmental conditions within an area on the eastern Agulhas bank used for spawning and juvenile stages of *L. reynaudii*, resulting in less recruitment (Roberts 2005). It may also be the result of increased fishing pressure from the targeted jig fishery introduced in the 1980s, as prior to this Chokka was sourced solely as bycatch by the demersal trawl fishery (Jereb and Roper 2010).

Prior to the 2000s, individuals of the genus Scyphozoa or 'true' jellyfish were absent from trawl data on the west and south coast, primarily as these species are pelagic in nature. In the 2000s, jellyfish were encountered more frequently, particularly on the west coast where they were encountered in three out of four years. It is conceivable that increased encounters with jellyfish species, at depths exceeding 100m, may indicate destabilising of marine ecosystems, possibly due to top-down pressure from overfishing, creating a niche that other species can exploit (Lynam et al. 2006) Proliferation of jellyfish may, in part, be explained by heavy fishing pressure on species targeted by the small pelagic fishery, coinciding with lower landings recorded in 2000s (Blamey et al. 2015, Lynam et al. 2006). Furthermore, frequent ecosystem shifts have been noted for species targeted within this fishery (Blamey et al. 2015, Coetzee et al. 2008, Shannon et al. 2020).

Endemism was more pronounced in the south coast mining area, compared with the west coast, conforming with a well-established pattern of endemicity that peaks on the south coast (Awad et al. 2002, Griffiths et al. 2010, Sink et al. 2019). During the 2000s, several endemic fish species, including *Chrysoblephus gibbiceps* and *Rhabdosargus globiceps,* known to have suffered from heavy exploitation in previous decades, were observed at greater frequency than previous decades. Due to the longevity of these sparid species, particularly of red stumpnose (Attwood et al. 2019, Griffiths et al. 2002), it seems unlikely that their increased presence on the Agulhas Bank in the 2000s is the result of population growth. Instead, it is possible that exclusion zones surrounding marine infrastructure may have provided refugia for fishing pressure for these overexploited species. On the other hand, it is also possible that increased presence of previously less-encountered species may be due to differences in trawl gear used by DFFE, as discussed previously (Blamey et al. 2015).

Data sourced from OBIS were the only dataset where time-series analyses were feasible, considering patterns between the three decades studied. The ways in which open-source data can be applied are wide-ranging, as discussed previously. Nonetheless, several functional challenges were encountered with publicly-available data in this study. Firstly, how the biota are characterised in the data is constrained by the type and focus of contributing parties. For example, fisheries are likely to focus on species of commercial importance, as encountered here with DFFE trawl data. Research bias towards particular taxa has led to distinct under-representation within some taxa (i.e., smaller benthic invertebrates, Griffiths et al. 2010, Yemane et al. 2020), limiting knowledge of the distribution of these taxa and their contribution to biodiversity in various ecosystems. Information on methodology used in contributing datasets and metadata is often lacking or misunderstood. Similar findings were identified in a study mapping coastal and marine ecosystem services in Europe (Caro et al. 2018). Though the application of open-source data used in Caro et al. (2018) differed from this study, the constraints of publicly-available data appear to be similar across various repositories. Patchily-distributed sampling may lead to species richness being underestimated within less-studied regions. When considering the utility of open-source data in relation to the impact of mining, openly-accessible data were found to be generally unsuitable on account of being generic to the whole concession, meaning it was not possible to distinguish at the finer-scale. To be able to decipher whether mining had caused a distinguishable impact in either concession would require suitable BACI (Before-After-Control-Impact) data and relevant controls, all of which were lacking in the data utilised for this study. Such finer-scale data would include, information relating to whether data were collected from a mined area, the time elapsed since a mining event and suitable baseline data. Furthermore, lack of abundance data constrained temporal analyses at lower taxonomic levels (i.e., species or genus) as it was not possible to determine how mining pressure impacted the abundance of a particular species when using presence-only data, for instance, whether increased richness reflected an increase of previously less-abundant species (Yemane et al. 2004), range expansion or simply differences in sampling gear.

4.3 Mining Impacts

Dependent on the type of mining being carried out and the particular impacts expected, data may need to be collected in different ways to accurately consider how anthropogenic disturbance directly impacts local biodiversity. This was considered here by means of datasets investigating how biodiversity is impacted by diamond mining (Steffani and Pulfrich 2008) and the installation of mining platform infrastructure (Cook 1995-1999).

There has been an upsurge in research relating to marine infrastructure, which has documented the role of infrastructure in providing a stepping-stone for non-native species to extend beyond their natural distribution (Adams et al. 2014, De Mesel et al. 2015). If introduced species become invasive, the threat to the preservation of local biodiversity may be substantial where native species are outcompeted for nutrients or space (De Mesel et al. 2015). On the Agulhas Bank, the pervasive Mediterranean mussel *Mytilus galloprovincialis* was the only known invasive species identified as part of the biofouling community on the FA platform in the mid 1990s. Of the possible colonisation routes, passive dispersal of larvae and use of a vector are discussed here. Passive dispersal is driven by oceanography and wind stress exerted within a particular region, transporting larvae away from the parent population, as demonstrated in McQuaid and Phillips (2000). Comparing wind patterns with the dispersal of larvae from native brown mussel *Perna perna* and *M. galloprovincialis*, McQuaid and Phillips (2000) discovered that larvae are displaced like 'passive particles', for up to 200km from the parent population. *M. galloprovincialis* larvae demonstrated much greater dispersal capacity than native *P. perna*, also part of the biofouling community identified here (McQuaid and Phillips 2000). This was further demonstrated in Miller et al. (2018b), where authors describe how 300 coastal species, including *M. galloprovincialis,* were transported more than 6,000km via marine debris generated by the 2011 earthquake and tsunami in Japan. Up to 50% of the debris sampled by Miller et al. (2018b) were colonised by *M. galloprovincialis,* demonstrating sheer resilience within the species. For larvae displaced offshore, permanent infrastructure provides a novel vertical habitat for hard-substrate species to colonise amongst the largely unconsolidated sediment (Degraer et al. 2020). Given the offshore location of the platform, it is possible that colonisation by *M. galloprovincialis* and indeed other coastal species, could have occurred by way of ship-mediated vectors (Chan et al. 2015, Gollasch 2002), infrastructure (Capel et al. 2019) or possibly even construction equipment. Oil platforms were recognised as the most probable vector for the introduction of two invasive cup coral species, *Tubastraea coccinea* and *Tubastraea tagusensis*, transported to Brazil from the Indo-Pacific region where they became invasive, dominating around 3,500km of Brazil's coastline. Capel et al. (2019) clearly demonstrates the role of infrastructure as a stepping-stone in the introduction of species to habitats outside their distribution. In their study of petroleum infrastructure on the Agulhas bank, Sink et al. (2010) identified a further seven introduced and three cryptogenic species, three of which are potentially invasive due to extensive cover of petroleum infrastructure (Sink et al. 2010). Discovery of a further nine non-native species over a 10-year period highlights the unmistakable role of marine structures as stepping-stones for non-native fauna, with potentially severe impacts on local biodiversity. The risk of introducing species is likely to be considerably lower when mining soft sediment habitats due to a lack of suitable substate for hard-substate invertebrates like mussels, barnacles and oysters to attach (De Mesel et al. 2015), assuming equipment has been properly cleaned prior to its introduction into the marine environment. Validation of this was shown in ML3 as no invasive macrofauna were identified from samples collected either before or after diamond mining.

Several differences in species abundance were revealed during the first monitoring survey in ML3 (approximately 6 months after mining, Pulfrich and Steffani 2007), with some taxa displaying greater abundance in sites subjected to mining pressure. The snapper shrimp *Calocaris barnardi*, belonging to the Axiidea (formerly Thalassinidean) family, was the most abundant species in both mined sites in 2008. It is possible that the abundance of *C. barnardi* is a result of the considerable depths (of up to 3m) burrowed by Axiidea species, where they feed on deposits deep within the sediment (Pillay and Branch 2011). Burrowing to such depths may adhere individuals with some protection from removal of shallow sediment from the seafloor during the diamond mining process. Based on their extensive review, Pillay and Branch (2011) communicated that Thalissindeans possess a number of adaptations that allow them to survive when exposed to a wide range of extreme environmental conditions, including hypoxia/anoxia and even sulphide exposure. Although mining is likely to have had some impact on abundance of individuals burrowing at shallower depths, the ability of *C. barnardi* to burrow at considerable depth, coupled with high tolerance to environmental variability, is likely to have protected the species from the extreme disturbance experienced by macrofauna found within top sediment layers. Predatory amphipods *Thermisto gaudichaudi* (Hyperiidae; family) and *Paraphoxus oculatus (*Phoxocephalid; family) also contributed substantially to total abundance, though both species were considerably more abundant in unmined sites. Unlike the snapper shrimp, *T. gaudichaudi* and *P. oculatus* are actively predatory, residing upon the sediment surface. Phoxocephalid species are known to feed opportunistically on benthic invertebrates, favouring small soft-bodied invertebrates such as nematodes and polychaetes (Oliver et al. 1982). Using gut analysis, *T. gaudichaudi* was also found to feed non-selectively, consuming copepods, worms, diatoms and phytoplankton (Watts and Tarling 2012). It is likely that nearby communities of *T. gaudichaudi* and *P. oculatus* were able to exploit benthic destruction created by mining activity, possibly allowing them to feed on the soft-bodied insides of hard-shelled species that are usually unobtainable. The polychaete *Paraprionospio pinnata* was also among the most abundant species present in 2008, though numbers varied between the two mined sites. High abundance of an anoxia-tolerant species such as *P. pinnata* may suggest that damage caused to seafloor sediment has altered oxygen levels, presumably due to a reduction in ecosystem engineer species (Steffani et al. 2015). Although these findings contrast with similar studies in Namibia mining areas (Savage et al. 2001, Van Der Merwe 1997), the differences may be partially explained by environmental variability between Namibia and the west coast of South Africa.

Large scale seafloor disturbance can significantly alter seafloor habitats and benthic fauna inhabiting them, reducing diversity and abundance and altering species assemblage (Lundquist et al. 2018). On the west coast, results revealed a reduction in species abundance and diversity following diamond mining within mined sites, with some species exhibiting greater sensitivity to mining disturbance than others. For instance, exceptionally low abundance was observed for the relatively long-lived burrowing mud prawn *Callianassa australis* (Steffani et al. 2015), up to four years after cessation of mining. It is possible that sensitivity observed in *C. australis* may be linked with changes in sediment type generated by mining, as they require a sediment that can support the extensive burrow networks they create (Pinn et al. 2010). Sensitivity of benthic species such as *C. australis* highlights the scale of impacts imposed on macrofauna with little exposure to natural disturbance. Apart from the expected reduction in biodiversity within the community, significant changes in species composition within both undisturbed (control) and disturbed (mined) sites were also observed. Multivariate analysis revealed a significant difference in species composition between all years surveyed, consistent with previous studies (Boyd et al. 2005, Newell et al. 2004, Pulfrich et al. 2013). Of the factors measured, differences in communities were best explained by variation between years, with mining playing a significant, but lesser role. In regions of high environmental variability, such as the Benguela, the true impact of anthropogenic disturbance is likely to be masked by natural fluctuations within the environment, as shown in similar studies (Pulfrich et al. 2004, Pulfrich and Branch 2014, Seiderer and Newells 1999, Sciberras et al 2013). Larger sample sizes, access to metadata and the corresponding environmental data would have substantially improved analyses and subsequently the interpretation.

Distinct changes in macrofauna were observed in ML3 in the years following mining, from a copepod and crustacean-dominated community to predominantly crustaceans (decapod, peracarid) and polychaetes. Following mining, Tellinidae *spp* remained absent from mined areas for around four years, whereas thereafter they were encountered in low numbers. Immediate impact and slow recovery of Tellindae *spp* may be due to a combination of factors, for example, mortality or damage sustained from mining, burial within sediment that may limit filter-feeding ability, or possibly changes in sediment composition (e.g., grain size, and sediment type) generated by the removal and subsequent filling in of sediments (Pulfrich and Steffani 2007). Species inhabiting disturbed and undisturbed sites displayed a high degree of similarity to each other. Due to the proximity of sites, similarity in community assemblage may indicate that these spatially-distinct communities originate from the same metacommunity. Aside from a select few, most taxa were expectedly less abundant following removal of seafloor sediments and benthic organisms. However, the natural hydrology of a region facilitates the displacement of (unconsolidated) sediments that 'fill in' the damaged seabed (Pulfrich and Steffani 2007), allowing some species to recolonise disturbed areas. For some, the impacts of disturbance may take much longer to recover, particularly in the deep-sea where natural disturbance is limited (Drazen et al. 2020).

The benthic community in ML3 underwent significant modification between 2003 (pre-mining) and 2008 (post-mining). One hypothesis for this is that the baseline community sampled in 2003 was not characteristic of the region, particularly as macrofauna were sampled approximately four years before mining was initiated. It is conceivable that there was a significant environmental disturbance, for example, flooding of the nearby Orange River, or exceptional hypoxia in bottom waters (Jarre et al. 2015), both of which are known to occur periodically. In some cases, variable environments such as the Benguela, may actually confer benefits on disturbed environments, for instance, where river sediments moved by flooding speed up the pace of recovery by 'filling in' disturbed sediments (Pulfrich and Steffani 2007).

Artificial reefs generated by the introduction of marine structures are widely believed to increase biodiversity through the provision of novel habitat. However, ecological costs associated with these structures are likely to outweigh the perceived benefits to local biodiversity, including alteration of local biota, hydrodynamics and larval dispersal (Bishop et al. 2017, Degraer et al. 2020, Dunn et al. 2019), in addition to providing habitat for invasive species (Adams et al. 2014, De Mesel et al. 2015, Degraer et al. 2020). Artificial reefs are often inherently different to natural reef systems, due to differences in substrate type and complexity, oceanographic conditions and infrastructure age, amongst other factors (Bishop et al. 2017). The fouling community supported by the FA platform consisted of numerous hard-substrate species, such as mussels, barnacles, anthozoans and anemones, resembling intertidal and shallow coastal habitats in South Africa (Sink et al. 2010). Evidence of vertical range expansion was also observed on the platform, with intertidal species such as mussels and barnacles observed at depths of up to 50m, far exceeding their usual distribution (Sink et al. 2010). The fact that estuarine molluscs *Arcuatula capensis* and *Dosinia hepatica* were found at depths of 30m and 15m respectively, highlights the significance of marine structures in creating dispersal pathways, enabling species from widely different habitats (i.e., coastal, brackish) to exploit open ocean environments. Where hard structures are introduced into a fluid environment, there is likely to be impact on ocean interconnectivity, water flow (e.g., currents, tidal movement), population dynamics (i.e., recruitment, gene flow) and the movement of resources. Vertical placement of structures such as rigs and wind turbines within the water column may also affect sediment dynamics, currents and tidal movement that disperse larvae and nutrients, limiting the dispersal of gametes, with impacts on gene flow and resilience (Bishop et al. 2017, Dunn et al. 2019). If infrastructure is no longer required or becomes obsolete (i.e., well closure), several measures can be taken to mitigate the biodiversity impacts associated with the introduction of infrastructure. An example of this may include the removal of infrastructure onto land, where it should be extensively cleaned to prevent further impacts on biodiversity, although removal of infrastructure is often costly for mining companies. If the decision is made to leave structures in-situ, it is essential that the ramifications for biodiversity, fisheries and other ocean using industries are fully considered.

5 Conclusion

Extractive industries are a well-recognised pressure on the preservation of marine biodiversity (Sink et al. 2019), particularly in offshore seafloor habitats that remain less well-studied than coastal systems (Griffiths et al. 2010, Yemane et al. 2020). This study was able to highlight the utility of open-source data as a first step in quantifying benthic biodiversity within an area where mining is proposed. The utility of openly-accessible data in evaluating the impacts imposed by mining was found to be constrained by a lack of fine-scale information, for example, disturbed versus undisturbed. Such fine-scale information is often only available in studies designed to consider the type of mining proposed and the expected impacts thereof, as demonstrated with the De Beers (Steffani and Pulfrich 2008) and Cook (1995-1999) datasets evaluated here. Due to the variety of impacts generated by different extractive industries, it is recommended that mitigation measures be tailored to the type of mining proposed and the impacts expected, as opposed to the most cost-effective method. Despite the increasing application of open-source data in marine ecology, constraints encountered here (e.g., presence-only data, missing metadata, gaps) further highlight the need for greater sampling and access to data relating to the offshore seafloor. Further recommendations are that in-situ sampling be a requirement of impact assessments, utilising at least two gear types to sample a greater proportion of benthic biota (Flannery and Przeslawski 2015). Additionally, datasets should be made available to the greater scientific community, to improve understanding of offshore ecosystems and how they are impacted by anthropogenic stressors. It is also recommended that benthic datasets should exclude species that are accidentally sampled, as was encountered with copepods in the datasets utilised in this research, as this further complicates the analysis of these large datasets. Implementation of long-term monitoring programmes (timescale dependent on impacted habitat) is also highly recommended for the same reasons mentioned above. Future studies could improve on this research by utilising an open-source repository with relevant environmental data, demonstrated in Yemane et al. (2020) to map mining pressure with open-source biodiversity data.

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