



Seabed Mining in the Northern Territory

Review Paper

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Glossary of Terms

Term	Definition
Accretion	Filling of the sea floor by deposition of sediment
ADCP	Acoustic Doppler Current Profiler
ADZ	Active Dredge Zone
Anoxia	The absence of oxygen
ANZECC	Australian and New Zealand Environment Conservation Council
ARMCANZ	Agriculture and Resources Management Council of Australia and New Zealand
BACI	Before-After-Control-Impact
BAT	Best available technology
Bathymetry	The depth and topography of the seabed
Benthic Fauna	Animals that live on the seabed surface (epifauna) or within the seabed (infauna)
Biomass	The mass (weight) of organisms in an ecological community
BMP	Best Management Practice
Borrow Site	An area of seabed in which sediments are removed to replace those lost elsewhere
CHIRP	A Chirp system emits an acoustic pulse (called a "ping") from the transducer (located in a towfish) that travels down through the water column which is used to image sub-surface features in an aquatic environment
Coastal Waters	The body of water 3 nautical miles seaward of the Territorial Sea Baseline
Community	An assemblage of animals and plants that live together
CSD	Cutter Suction Dredger
CSIRO	Commonwealth Scientific and Industrial Research Organisation
DoEE	Department of Environment and Energy (Commonwealth)
DENR	Department of Environment and Natural Resources

Term	Definition
DLRM	Department of Land Resources Management (NT)
DNREA	Department of Natural Resources, Environment and the Arts (NT)
DPIF	Department of Primary Industry and Fisheries (NT)
DRDPIFR	Department of Regional Development, Primary Industry, Fisheries and Resources (NT)
DSM	Deep Sea Mining
EIA	Environmental Impact Assessment
EIS	Environmental Impact Statement
ESHIA	Environmental, Social and Health Impact Assessment
EW	Environmental Window
JTU	Jackson Turbidity Unit
LHC	Life History Characteristics
MP	Management Practice
NT	Northern Territory
NT EPA	Northern Territory Environment Protection Authority
NTU	Nephelometric Turbidity Unit
<i>EPBC Act</i>	<i>Environment Protection and Biodiversity Conservation Act 1999</i>
Eutrophication	A process in which nutrients (usually phosphorous and nitrogen) become highly concentrated leading to increased growth of organisms such as algae or cyanobacteria
Footprint	The area of seabed that is directly affected by mining
IBA	Important Bird Area
MNES	Matters of National Environmental Significance
Nautical Mile (nm)	The equivalent of 1852 metres
NT	Northern Territory
NT EPA	Northern Territory Environment Protection Authority

Term	Definition
NPA	Northern Planning Area
PAR	Photosynthetically Active Radiation
Placer Deposit	An accumulation of valuable minerals formed by gravity separation during sedimentary processes
PTS	Permanent Threshold Shift
ROV	Remotely Operated Vehicle
Sediment Plume	An area of elevated suspended solids in the water column
SEL	Sound Exposure Level
SSC	Suspended Sediment Concentration
SPL	Sound Pressure Level
TSB	Territorial Sea Baseline
TSS	Total Suspended Solids
TTS	Temporary (Auditory) Threshold Shift
TSHD	Trailing Suction Hopper Dredger



Executive Summary

A moratorium on seabed exploration and mining in coastal waters of the Northern Territory was implemented by the Northern Territory (NT) Government on March 6 2012, for a period of three years. This was extended by a further three years on 5 March 2015 to allow for a sufficient review of actual and potential impacts of seabed mining and the methods for management of the impacts of mineral extraction. This review, commissioned by the Northern Territory Environment Protection Authority (NT EPA), provides an overview of the potential impacts of seabed mining on the NT coastal environment and also outlines methods for managing potential impacts.

Seabed mining can be subdivided into two components: shallow marine mining and deep sea mining. Although the distinction between shallow marine mining and deep sea mining is not formally demarcated, an emerging consensus says that deep sea mining is the removal of minerals from seabeds deeper than 500 m. In NT coastal waters, the continental shelf depth is generally less than 50 m, which is the focus of this review.

Shallow water mining techniques have historically developed from, and are largely based upon, dredging technology. Large-scale dredging has occurred internationally for centuries and modern industrial techniques are well-established. Economic recovery of seabed mineral resources has also been undertaken on a commercial scale for over a century. Various dredging technologies have been utilised to recover a broad range of mineral resources including tin, gold, iron sands and diamonds but by far the largest seabed mining activity occurs in the recovery of sands and gravels.

From an environmental perspective, exploratory mining techniques are relatively benign as they generally rely on airborne geomagnetic studies or marine-based geophysical and geotechnical surveys for collection of data. Where direct sampling is required, such as geotechnical coring or vibracoring, the environmental effects are usually very localised. In contrast, the process of mining or excavation of minerals from the seabed is a very direct and destructive process. The extent of impact on species, communities and the broader environment is largely dependent on a range of direct and indirect physical impacts that can result in a range of direct and indirect ecological effects. Key aspects that would require detailed consideration in any mining proposal would be the assessment of seabed disturbance and the direct loss of benthic habitat, assessment of turbidity and sedimentation and the assessment of tailing disposal. A large number of potential effects have been considered in the review and all would need to be assessed as part of the impact assessment process. The indirect effects of mining on coastal processes should also not be underestimated, in particular where mining is likely to occur relatively close to shore.

Many of the physical, chemical and biological effects of seabed mining (in shallow waters) are reasonably well-understood, as the process of dredging relies on similar extractive methods and processes. As a result, many of the monitoring and mitigation strategies proposed have been taken from large-scale dredging projects. Similarly, the tolerance limits discussed in the review are based on knowledge obtained from other dredging projects elsewhere in Australia, using species that are most vulnerable to impact, such as corals and seagrass. While application of such limits to the NT marine environment may not be directly applicable, they can be used in a precautionary manner in the absence of NT specific guidelines.



Overall, the greatest uncertainty associated with understanding potential impacts associated with seabed mining remains the relatively poor current state of knowledge about the NT marine environment.



1 Introduction

1.1 Background to Study

A moratorium on seabed exploration and mining in coastal waters of the Northern Territory was implemented by the Northern Territory (NT) Government on March 6 2012, for a period of three years. This was extended by a further three years on 5 March 2015 to allow for a sufficient review of actual and potential impacts of seabed mining and the methods for management of the impacts of mineral extraction. The moratorium prevents any mining activities within NT coastal waters. There is currently limited information on the impacts and management of seabed mining, which limits the capacity of the Minister to assess industry management and apply appropriate conditions relating to the authorisation of this type of mining.

This report has been commissioned by the Northern Territory Environment Protection Authority (NT EPA) to assist with providing an overview of the actual and potential impacts of seabed mining on the environment and proposed methods for managing those identified impacts.

1.2 Objectives/Scope of Review

The specific objectives of this review are to identify and describe:

- Shallow-ocean mining techniques as they may relate to the coastal waters of the Northern Territory;
- The actual or potential impacts of seabed mining on the Northern Territory environment;
- Tolerance limits for the identified impacts for key marine receptors in the Northern Territory;
- Mitigation and monitoring strategies for managing the impacts of seabed mining; and
- Best-practice measures and technologies in seabed mining.

The review considers only the Coastal Waters of the Northern Territory as defined by the Commonwealth *Coastal Waters (Northern Territory Powers) Act 1980*; the waters between the seaward 3 nautical miles (nm) from the Territorial Sea Baseline. It also only considers the negative environmental impacts resulting from sea-based mining and management (i.e. it excludes onshore mining and marine disposal of onshore produced tailings), though where specific environmental impacts favour a species or community, this is mentioned.

For the purpose of this review:

- Seabed mining is defined as the commercial recovery of minerals at the surface of or below the seabed. This includes the exploration and mining of a 'mineral' or 'extractive mineral' as defined in the NT Mineral Titles Act but does not include oil and gas recovery.
- Northern Territory waters are all coastal waters of the NT, as defined under the Commonwealth Coastal Waters (Northern Territory Powers) Act 1980. 'Coastal waters' refers to the belt of water between the limits of the Northern Territory and a line 3 nautical miles seaward of the Territorial Sea Baseline (TSB). The TSB normally corresponds with the low water line (i.e. the level of the Lowest Astronomical Tide).



2 Shallow-Ocean Mining Techniques

2.1 Definition

Seabed mining can be subdivided into two components: shallow marine mining and deep sea mining. Shallow marine mining largely refers to the extraction of mud, sand and gravel for construction purposes and in some cases can also refer to the mining of valuable minerals in the nearshore shallow waters (SOPAC 2004). Although the distinction between shallow-water mining and Deep Sea Mining (DSM) is not formally demarcated, an emerging consensus says that DSM is the removal of minerals from seabeds deeper than 500 metres as referenced at <https://www.oceanfdn.org/resources/seabed-mining>.

In NT coastal waters, the continental shelf depth is generally less than 50 m (Harris *et al.* 2005) and hence the focus of the review is on shallow marine mining.

2.2 Background

Shallow water mining techniques have historically developed from, and are largely based upon, dredging technology. Large-scale dredging has occurred internationally for centuries and modern industrial techniques are well-established and understood. Economic recovery of seabed mineral resources has also been undertaken on a commercial scale for over a century. For example, exploitation of coastal diamond deposits in the southern African region started in 1908 with the discovery of surface deposits near Lüderitz, Namibia, and discovery of large diamond-gravel deposits north and south of the Orange River Mouth from 1927 - 1945. Actual marine diamond mining started in 1960 after discovery of offshore diamond deposits near Chameis Bay, Namibia (Penney *et al.* 2008). Various dredging technologies have been utilised to recover a broad range of mineral resources including tin, gold, iron sands and diamonds but by far the largest seabed mining activity occurs in the extraction of sands and gravels.

In recent decades the opportunity to exploit seabed resources has become possible due to the development of more specific and advanced technology that allows exploration in deeper waters (SPC 2011). In the case of potential economic mineral resources, in the Gulf of Carpentaria and coastal waters of the Northern Territory, established commercial mining and dredging techniques are applicable due to the relatively shallow nature of the continental shelf.

Coastal waters of the Northern Territory are generally shallow and relatively calm compared to the open ocean and have limited access from shore. This lends well to mining techniques developed during the past century for the diamond industry in West Africa. These techniques however, are yet to be used for bulk commodities on a commercial scale, thus, their economic viability is currently unknown.

There are four key stages of work involved in mining for minerals (Environment Foundation, NZ 2016). These include:

- **Prospecting (usually non-invasive)** – The purpose of this stage is to identify areas that are likely to contain mineral deposits. The work includes geological, geochemical and geophysical surveys (e.g. seismic surveying), aerial surveys, including the taking of samples by low-impact mechanical methods.



For example, in the marine environment, multibeam swath bathymetry can be used to map the seafloor, and Remotely Operated Vehicles (ROV) or submersibles can be used to collect images of the sea floor, take core samples of the seabed at low concentrations, and collect targeted samples.

- **Exploration (invasive)** – The purpose of this stage is to evaluate the feasibility of mining identified deposits. This stage can include any drilling, dredging or excavations needed to assess the nature and size of a mineral deposit.

The exploration phase may involve core sampling at higher concentrations (than undertaken in the prospecting phase) in small areas, test pit excavation, test drilling, trials of extraction methods and bulk sampling. This exploration could involve the removal of larger volumes of deposits.

- **Production** – The purpose of this stage is to actively extract and process the mineral resource. Minerals in the terrestrial environment may be mined using underground or open cut methods. Open cut mining is used in situation where the minerals are relatively shallow or less concentrated. Underground mining is used for deposits at greater depth, higher grade deposits, or vein ores. Underground mining is more expensive, but less environmentally destructive.

Techniques for mining minerals in the marine environment may involve seafloor suction dredging, seafloor slurry pipes, use of tracked vessels on the seafloor and seafloor cutting/fragmentation. Following extraction and processing, tailings (unwanted material) may be deposited back onto the seafloor. Some production operations involve a processing vessel that moors to "permanent" anchor blocks or anchor moorings.

- **Rehabilitation** – The purpose of this stage is to restore the area following the conclusion of mineral production. Techniques will vary depending on the mining that has occurred the location of the mining and the desired future use of the area. (It should be noted that active rehabilitation of the seabed is not usually feasible and generally relies on natural recovery).

2.3 Economic Application

Of the many factors that influence the economic viability of mining a marine ore body, the most important include:

- The grade and size of the resource;
- The metocean conditions (e.g. wind, waves and tides) at the area of operation; and
- The depth of water at the resource.

Extensive exploration work is yet to be undertaken in NT waters by any company to determine whether seabed mineral prospects will be economically and technologically feasible to extract (NTEPA 2012), however it is understood that manganese is a potential target resource. Seabed mining exploration licenses highlighted in Figure 2-1 are south of Groote Eylandt.

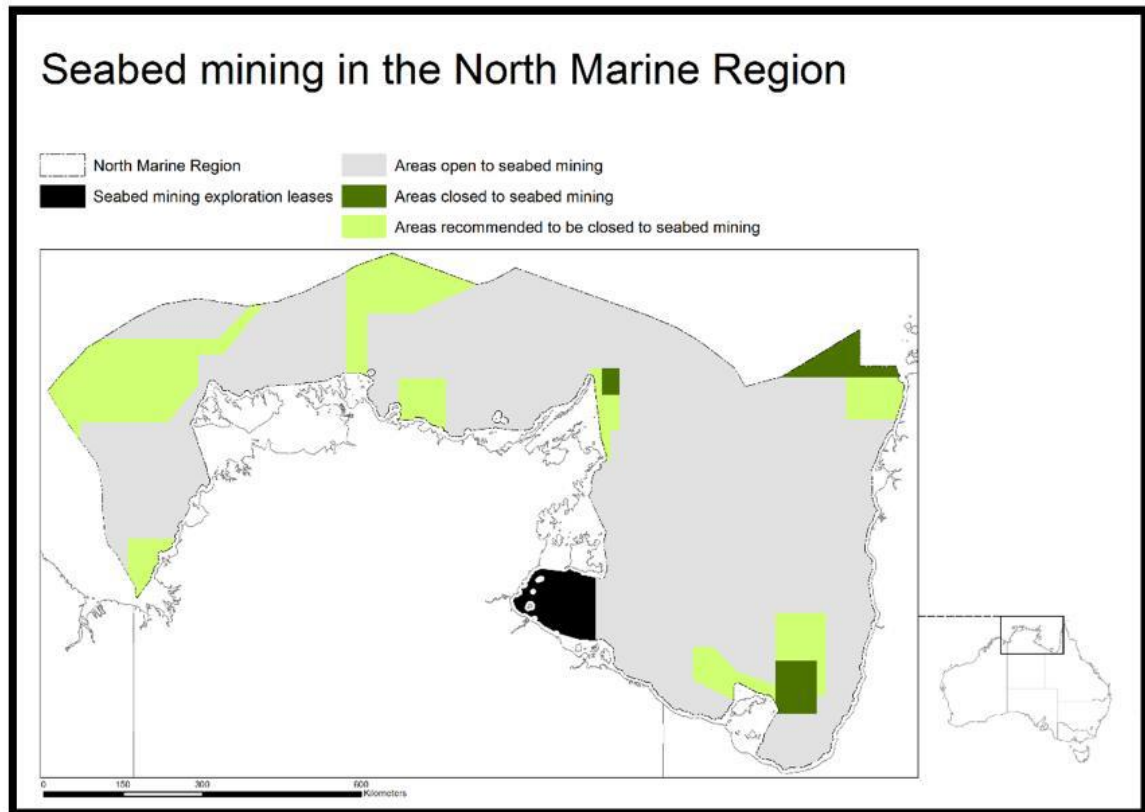


Figure 2-1 : Seabed Mining in the North Marine Region (from CCG 2015)

According to Geoscience Australia (2015) the most common form of offshore exploration and mining occurs as an extension to mining onshore deposits. Examples of where mineralisation is known to extend offshore from onshore deposits, or where exploration has been carried out offshore for such extensions are:

- Coal. At Newcastle in New South Wales, offshore seams have been mined by underground methods from onshore. The Gippsland Basin off the coast of Victoria has brown coal deposits.
- Scheelite (Tungsten). At King Island off Tasmania, mineralisation occurs in basement rocks of the continental shelf surrounding the island.
- Manganese. At Groote Eylandt off the eastern Northern Territory rich manganese beds in the Lower Cretaceous are known to extend off the island.
- Iron ore. At Cockatoo Island and Koolan Island off Western Australia iron ore extending under the sea floor is being mined using seawalls

Other examples of potentially viable mineral deposits in coastal waters around Australia include the presence of tin in Ringarooma Bay in north east Tasmania. Van Dieman Mines plc is investigating the viability of mining onshore and offshore deposits of tin bearing alluvial resources with associated gem and heavy mineral sands. Offshore the company has reported that around 200 million cubic metres of an alluvial tin resource has been identified.

Offshore accumulations of tin have also been identified off King Island and Cape Barren Island in Tasmania. Some offshore exploration for tin and tantalum also has been undertaken in Bynoe Harbour in the Northern Territory.



Offshore mineral deposits occur in a broad variety of geological forms including unconsolidated sands and gravels (as placer deposits), consolidated cobalt rich ferromanganese crusts, massive sulphide deposits and polymetallic nodules. Some deposits can be solidly composited with non-valuable materials while others such as diamonds and gold may lie on the bedrock covered with overburden of silts and sands. Other deposit types such as manganese nodules tend to lie on or close to the surface of the seabed.

2.4 Seabed Mining Techniques

Dredging techniques have been utilised to recover economic minerals for over a century, first utilised commercially with the recovery of gold on the Clutha River in the southern New Zealand gold fields in 1881. Common techniques including bucket, hopper and suction dredging, are utilised globally to recover sands and gravels. The resulting products are used for land reclamation, a source of cost-effective fill materials, construction aggregate and in infrastructure and urban construction. Although largely discontinued (for economic reasons), bucket and suction dredging has been used to recover gold in the Philippines, Korea, New Zealand and USA and to recover tin in Indonesia, Thailand and the UK. The techniques involved are well-established and have been used in water depths up to about 50m. There is plentiful information in the public domain regarding dredging techniques, which are summarised below. More detail is provided in the USC OTA (1987) assessment of marine mineral exploration in the USA.

2.4.1 Exploration Techniques

Exploration generally involves reconnaissance drilling to recover sediment samples for analysis onshore, airborne geomagnetic studies and seismic surveys, as summarised in Table 2-1 (from EFNZ 2016, after Thompson 2012).

Table 2-1: Methods of Subsea Mining Exploration (from Thompson 2012)

Method	Summary
Boat-towed magnetometer systems	<ul style="list-style-type: none"> Useful for mapping placer deposits (an accumulation of valuable minerals formed by gravity separation during sedimentary processes) or diffuse deposits where the magnetic minerals are spread over a wide area Recent research has demonstrated that titanomagnetite can be mapped well using a seabed sled developed by the University of Bremen that measures magnetic fields over a sediment thickness of a metre or so. This work is still in progress
Vibracore sampling	<ul style="list-style-type: none"> Undertaken in water depths of up to 200m and to a target depth of 10m below the sea floor A single 2 to 10m long core tube (deployed and operated by a cable) is driven into the seabed A vibration allows the tube to penetrate the surface When the core is removed, the hole created collapses in on itself, leaving a slight depression in the seabed Sampling sites are generally discrete and around 200 metres



Method	Summary
	<p>from each other</p> <ul style="list-style-type: none"> ▪ A vibracorer does not work well if the material is too coarse, well- consolidated or cohesive
Reverse circulation sampling	<ul style="list-style-type: none"> ▪ Used for unconsolidated sediment samples ▪ Tube is generally around 70 – 80mm in diameter ▪ A series of water jets drive a hole into the substrate ▪ Often uses a drilling “mud” which is mixed with water and pumped down into the hole ▪ Samples withdrawn using an air-lift system ▪ Hole that has been created, collapses in on itself, leaving a slight indentation in the seabed around 250 millimetres deep and 400 millimetres in diameter ▪ Used for sampling along the Taranaki coast, NZ where six discrete samples per hole are taken at a depth of approximately 6m
Sonic drilling	<ul style="list-style-type: none"> ▪ High frequency vibrations sent down a drill ‘string’ (or sampling rod) to the drill bit. ▪ Vibrations resonate and magnify the amplitude of the drill bit, which then fluidises the seafloor ▪ Has the ability to recover samples to 100m depth (below seabed) ▪ Typically uses a 100mm drill bit within a 150mm external diameter sleeve. ▪ No materials discharged at sea ▪ Sampling generally occurs at discrete distances between 100 -200m ▪ Drilling “muds” (typically consist of bentonite clay, polymers and other additives) are sometimes used
Side-scan sonar and multi-beam sonar	<ul style="list-style-type: none"> ▪ Maps topography and sedimentary environment of seafloor ▪ Provides information about the surface sediments in terms of texture and reflectivity rather than the minerals that are present ▪ Side-scan sonar uses a towed sonar device to emit a conical or fan-shaped pulse of sound through the water across a wide angle ▪ Device can be towed or mounted to the hull of the vessel ▪ Water depth data is collected using multi-beam sonar ▪ The frequency of the sound is generally between 100 – 500kHz, but closer to 100kHz



Method	Summary
Low frequency echo sounder	<ul style="list-style-type: none"> Used to map sand in harbours and on the continental shelf Systems such as a CHIRP Sub-bottom Profiler can be used These generally operate at between 3.5 and 15kHz, which gives a maximum range of between 20 and 50m depending on the type of sediment This corresponds to a peak sound output of 149-158 decibels (compared to 176 decibels for a standard 200 kilohertz echosounder)

2.4.2 Mining (Extraction) Techniques

Shallow water techniques for recovery of economic minerals have in the past predominantly utilised established and well-understood dredging technology. In shallow waters, there is no technological limitation to mining. There have been cases of economic mineral dredging dating back over 100 years.

Four basic methods of mineral extraction on land or at sea are described by the USC OTA (1987) and all are potentially relevant for seabed mining in NT waters.

These include:

- Scraping the surface;
- Excavating a pit;
- Tunnelling underground (or below the seabed); or
- Extracting the mineral through a borehole or other conduit as a slurry.

The mining system that is used to excavate the mineral deposits from the seabed will largely depend on the following factors:

- The ease with which the material can be excavated and removed from the surrounding environment
- The water depth; and
- The metocean conditions at the area of operations.

Conventional shallow water systems include mechanical dragline dredging that are commonly used for placer recovery and trailing suction hopper dredges (TSHD) that are used for recovery of sand and gravel. The dragline dredge typically involves use of large dredge buckets that retrieve material from the surface of the deposit (on the seabed) and place them into barges for transportation to shore. In comparison, the TSHD uses a pump to draw a slurry of water and sediment into a riser pipe from the seabed into a hopper at the sea surface. As the sediment accumulates in the hopper, the excess water spills overboard back into the sea (often referred to as overspill or overflow). The method is most commonly used in maintaining depth in navigation channels but is used extensively in sand and gravel mining.

The maximum depth to which dredging (and mining) is possible is limited by the vacuum head generated by the dredge pump. A typical medium sized trailer can dredge economically to a



depth of 35m although dredging to depths of 80m is possible at reduced production rates. The overflow level can be adjusted allowing some control of the volume and character of sediment that is retained in the hopper (Bray *et al.* 1997).

The TSHD mines the seabed by creating numerous shallow trenches in the seabed commonly about 1m wide and 0.3m deep by use of one of several drag heads. The trailer dredge is normally rated according to its maximum hopper capacity which is typically in the range of 750 to 10,000 m³ but may be significantly larger (Bray *et al.* 1997). An example of how quickly the industry can evolve and adapt to demand is the commissioning of the the TSHD Leiv Eiriksson, which was launched in 2010, has a hopper capacity of 46,000 m³ and can dredge sand and rocks to a maximum depth of 142m (<http://www.ship-technology.com/projects/eiriksson-dredger/>) This example provides an insight into how quickly the industry can evolve and adapt to demand.

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Examples of conventional dredges operating from floating platforms include the clamshell bucket, bucket ladder dredge, bucket wheel suction dredge, stationary suction dredge and cutter suction dredge which can all operate in stationary or anchored mode.

2.4.2.1 Diamond Mining Technique

In recent decades, three techniques specific to the mining of undersea minerals have been commercialised by DeBeers in West Africa. All three techniques are utilised in the recovery of diamonds off the West Africa coastline in ocean depths up to 500m. The two dominant technologies utilised by DeBeers are known as “horizontal” and “vertical”. The third method used by DeBeers uses compressed air to create a vacuum and transport loose sediment material to the surface, and is otherwise known as airlift dredging.

Vertical recovery uses a large diameter drill-like mechanism to bore into the seafloor which loosens sediments and enable them to be pumped to the surface for processing. This technique was developed specifically to recover diamonds from seafloor sedimentary material. Commonly utilised in areas where the seafloor terrain is not conducive to the use of horizontal Remotely Operated Vehicle (ROV), this method has the advantage of being relatively selective and can be utilised in a broad range of terrain and ocean depths, though commonly shallower than 300m.

Horizontal mining of the seafloor utilises a large ROV crawler to dig the ocean floor and pump material to the processing ship. This technology is favoured by DeBeers and is their most commonly-utilised mining method, as it allows for larger tonnages per hour than the vertical method. A key limit to ROV mining can be the seafloor terrain and geology, as it may be unable to traverse excessively rocky substrate and may struggle with steep inclines or declines. This method was commercialised by DeBeers to recover diamonds from seafloor sands and gravels and has been used in depths up to 200m at material rates of over 260t/hr.

Variations on this horizontal technique show the greatest promise for undersea bulk hard rock mining and are currently being trialed in deep ocean applications. In shallow waters it is commonly accepted that traditional dredging technologies are more likely to be utilised due to there being less risk involved with applying well understood and established technologies.



2.4.2.2 Phosphate Mining Technique

Mining for phosphate generally requires a mining vessel that deploys a trailing suction drag-head that moves along the seabed. Water jets and possibly cutting teeth are used to loosen the top centimetres of seabed sediment, and the resulting slurry of sediment and seawater is pumped to the mining vessel. The material will be mechanically processed on-board the ship to separate the coarse phosphatic material from the finer non-phosphatic sediments. The fine non-phosphatic sediment and seawater will be discharged to the seafloor or above the seafloor (NIWA 2012).

2.4.2.3 Iron Sand Mining Technique

Active mining of the seabed is generally undertaken using suction pipes to pump sand that has been mixed with seawater from the seabed. Iron ore is magnetically separated from the sand, whilst other minerals are extracted by sieving, before returning the residue back to the sea. The sand mining methods used will vary, depending on the depth of the seafloor where the sand is being extracted from, and the distance from shore.

Trans-Tasman Resources Limited recently applied to mine iron sands in the South Taranaki Bight (New Zealand) using a subsea sediment extraction device called a "Crawler". Each crawler is 8m high and moves at 0.04km per hour. The operating crawler is tethered to the Integrated Mining Vessel (IMV) and controlled remotely from it, dredging the sediment from the project area. The crawler has onboard sensors to assist with navigation. The Crawler creates a mining cut approximately 12 metres wide and up to 11 metres deep. The seabed material would then be pumped by a slurry delivery pipe to an IMV where it would be processed to separate the iron ore (approximately 10 per cent yield).

The IMV will be 335m long and 60m wide and designed to operate through all expected weather conditions in the area and to be surveyed and maintained entirely while at sea. The IMV will include a processing plant, desalination plant and power generation, as well as house two seabed crawlers which operate on a rotation basis to dredge sediment from the seabed. Onboard the IMV, magnetic separators will remove the titano-magnetite ore without using heat or chemicals. Once ore has been extracted, the remaining 90 per cent of the sands extracted will be returned to the previously mined area in a controlled process just 4m above the seabed, restoring the area from which sand has been removed. Ore will be transferred to a transshipment vessel to be dewatered and stored before being transferred to export vessels.

Different methods are used for inshore mining, depending on the location of the sand resource. For example, at the Waikato North Head mine site, the iron sand is processed at an on-site concentration plant, where a magnetic concentrate is extracted through a series of separation processes producing a slurry. The slurry is pumped to the Glenbrook Mill through an 18 kilometre long underground pipe. Fresh water used during this process is drawn from a lagoon adjacent to the Waikato River. The by-products from these processes are used for site rehabilitation after they have been dewatered.

At the Taharoa mining site, sand is extracted from a lagoon by a floating dredge and then, because the area has no natural harbour, pumped as a slurry to a bulk cargo ship moored three kilometres from the shore plant to be processed and then exported. Between 200 and 300 tonnes of magnetic concentrate is produced per hour.



The removal of sand from the Mangawhai-Pākiri embayment is generally completed using a suction dredge. This is typically a rotary cutter suction dredge, where a rotating head disturbs the sediment, allowing it to then be entrained into the inflow of the suction pipe. Sand is then screened using mesh sizes of 3 millimetres, which removes gravel, shell and debris.

2.4.3 Mineral Separation, Transportation and Processing Techniques

Mineral separation technologies are many and varied and involve both physical and chemical processes, ranging from simple screening to complex integrated metallurgical plants with many process stages. The selection of mineral processing technologies depends greatly on the mineralogy and grades of the minerals present but can also be influenced by location and commercial preferences.

Dredging operations generally remove the whole of dredged material (whole of ore) and the spoil is transferred via pipeline or barge to a location remote from the dredging operation. In the case of economic mineral dredging (i.e. where a significant resource or ore body is identified and the cost of extraction provides a positive return on investment), if processing is required it will be almost entirely undertaken onshore. This is due to the large tonnages of dredged material, which is impractical to process at sea.

At present, the only example of offshore mineral processing and only significant commercial ocean mining is conducted by DeBeers off the coast of West Africa. DeBeers conduct limited mineral processing on the mining vessel in the form of screening and dewatering to create a mineral concentrate of sorts, which is transported by helicopter onshore for further processing. Compared to most mining activities, the processing of this diamond ore is very simplistic and has proven to be economically-viable, even on relatively large scales.

When evaluating offshore mining operations, it is generally assumed that offshore mineral processing will be limited to screening and dewatering and in some situations, basic gravity separation and clay washing, where unwanted particle fractions are then disposed back to sea. Processing is limited to such basic screening and dewatering due to the space availability and material handling issues on the mining vessel.

The processing of minerals is highly project-specific and any mineral deposit needs to be evaluated on an individual basis. Proposals for offshore mineral processing have thus far been very limited in scope even for large projects and are usually not thought to be commercially viable to process minerals at sea, except where mineral deposits are of high grade and require minimal processing.

If in the future there is a need to develop complex mineral processing at sea, there will be significant technical hurdles to overcome which will add to project risk. Some issues that would become apparent if mineral beneficiation were required at sea would be a lack of understanding of how processes may respond in the constantly moving environment of a ship. The availability of fresh water is also likely to be a major problem. For example, sulphide flotation often may not respond well in salt water and therefore froth flotation is limited where fresh water is not available and it is unclear how recoveries in a hydrocyclone may be affected by vessel movements. Another difficulty with offshore processing is the risk of environmental contamination from chemicals commonly used in mineral processing. Mineral processing plants carry large inventories of toxic and harmful chemicals. Chemicals such as PAX, Frothers and Cyanide pose major risks to marine life if spilled and therefore it is not practical or safe to carry the large inventories of these chemicals required for mineral processing.

3 Northern Territory Marine and Coastal Environmental Values

Northern Territory (NT) waters consist of some of the biggest and most pristine catchments, tidal estuaries and coastal wetlands in Australia. Located within the North Marine Bioregion of Australia (NMBPS 2007, Figure 3-1), it is largely characterised by shallow, soft sediment habitats and expansive coral reefs, seagrass meadows, mangroves and sand/mudflats that are important hotspots for biodiversity. Northern Territory coastal waters do not have high levels of endemism by Australian standards (DEWHA 2008), however are home to many significant populations of national and internationally threatened species including sawfish, pipefish, sharks and rays and are increasingly recognised as an area of global conservation significance and as an aggregation area and staging point for migratory birds. The region also provides important breeding, nursery and feeding areas for many species, including colonies of shorebirds, seabirds and waterbirds..

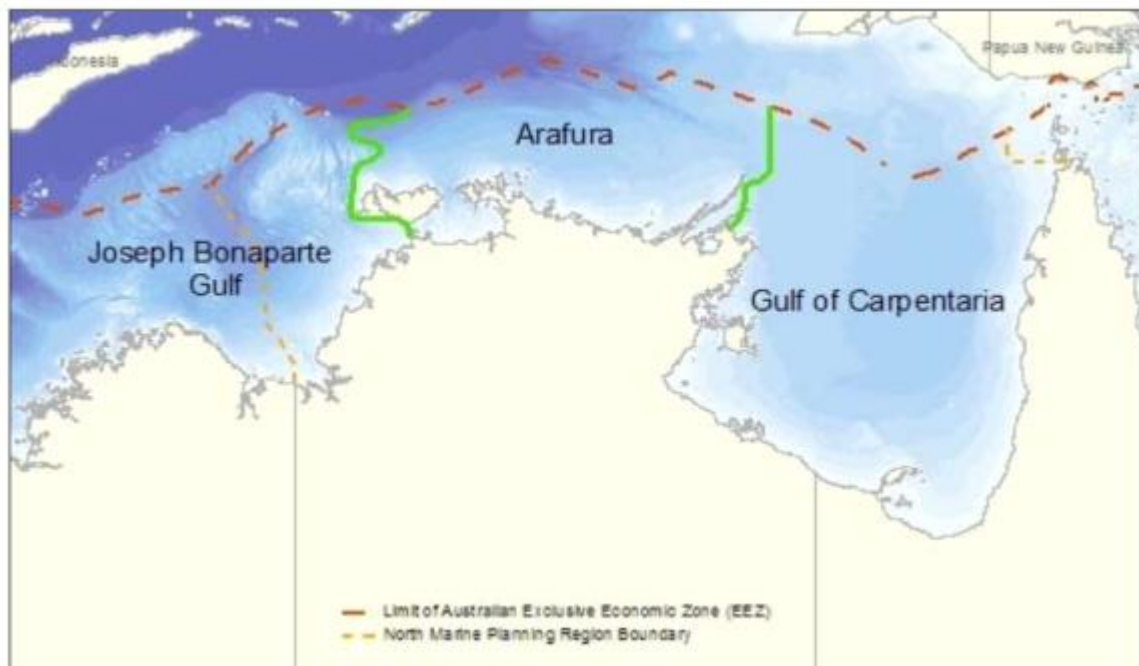


Figure 3-1 : The three ecological systems of the North Marine Region (NMBPS 2007)

Shallow coastal waters interact with the tropical monsoon climate of the region resulting in warm waters and a dynamic and unique marine environment (DEWHA 2007). There is a dramatic dry (April to November) and wet season (December to March) resulting in seasonal differences in water column mixing, waves and surface currents (DEWHA 2007; 2008). Monsoonal rainfall contributes significant quantities of freshwater runoff, sediments and nutrients to coastal waters in the wet season, creating highly productive near-shore environments (DEWHA 2007). Tidal currents are the major form of mixing, given the absence of major ocean currents in the region (DEWHA 2007). Tidal ranges are generally large, with mesotides (2-4m) in the Gulf of Carpentaria and Arafura systems, but exceeding four metres in the Joseph Bonaparte Gulf and subsequently this system is subject to the largest tidal range in northern Australia (DEWHA 2007). Cyclones are also characteristic of the tropical climate (DEWHA 2007; 2008).



Significant fisheries operate in the region, including the Northern Prawn Fishery. The gross value of the fisheries production in the North Marine Region (which includes some sections of Queensland) in 2014 was \$115.2 million (DSEWPac 2008). The Northern Prawn Fishery is a significant fishery utilising Northern Territory coastal waters and primarily targets banana and tiger prawns. The banana prawn fishery generally operates over a maximum of 10 weeks (April to mid-June) and the tiger prawn season operates from August to December (DSWEPac 2008; AFMA 2016). These prawns are usually found over muddy and sandy bottoms in coastal waters and estuaries. Banana prawns usually inhabit water depths between 16-25m and tiger prawns of depths up to 200m (AFMA 2016). Barramundi and mud crabs are also commercially important for the operation of NT fisheries.

3.1 Key Marine Features

3.1.1 Parks and Reserves

There are two marine parks and three coastal reserves in the Northern Territory's coastal waters:

- Cobourg Marine Park (also referred to as the Garig Garig Barlu National Park)
- Limmen Bight Marine Park
- Casuarina Coastal Reserve
- Channel Point Coastal Reserve; and
- Shoal Bay Coastal Reserve.

Furthermore, the following parks also include marine and coastal areas:

- Charles Darwin National Park
- Berry Springs Nature Park
- Tree Point Conservation Area
- Kakadu National Park
- Indian Island Conservation Area
- Keep River National Park Extension (proposed)
- Djukbini National Park
- Mary River National Park
- Limmen National Park
- Barranyi (North Island) National Park; and the
- Vernon Islands Conservation Reserve.

There are also two Aquatic Life Reserves; Doctors Gully and East Point, both of which are near Darwin. The Cobourg Marine Park and Kakadu National Park are also declared RAMSAR wetlands.

Soft sediment benthic habitats, seagrass meadows and coral communities, are specifically identified as regionally important communities and habitats for the marine bioregion (DEWHA 2008).

3.1.2 Soft Sediments

Soft sediments (muds and sands) are the most extensive and characteristic habitat of the Northern Territory coastal waters (DEWHA 2008) and the shallow nature of the coastal waters in the Territory make the extensive soft-sediment substrate attractive to seabed mining. This is due to shallow marine mining largely consisting of dredging of mud, sand and gravel for construction material or extraction of minerals (SOPAC 2004).

Few studies have focussed on the ecological characteristics of soft sediment habitats, and those in the Northern Territory are no exception, with studies largely limited to nearshore areas. Surveys completed by Russell & Smit (2007) and Smit et al. (2000) cover the majority of the inshore coastline between Cape Ford and Castlereagh Bay. The most recent surveys of Darwin Harbour (and immediate areas offshore from the harbour) were sampled as part of the INPEX Subtidal Benthos Monitoring Program (Cardno 2015e). More recent studies are lacking and of particular paucity is data relating to community composition in sediments at depths greater than 15-20m.

A diverse range of in- and epi-fauna rely on soft sediment habitats, including, for example, polychaetes, crustaceans, sea stars, sponges, solitary corals and ascidians. Microbial communities are essential to the continual functioning of the trophic web, as they form the basis of the food web, particularly for epi-benthic species (Russell & Smit 2007; DEWHA 2008; Cardno 2014).

A biodiversity survey focussing on the inshore coastal environments (sampling largely comprised of sediments between the 5-15m depth range) of North Western (NW) Arnhem Land was completed by Russell & Smit (2007). Here, sediments were found to consist of sand and/or mud. Similarly, the Beagle Gulf Benthic Survey (Smit et al. 2000) found that the majority of samples consisted of sand-sized particles (with mixed with either gravel or mud). Sediment types in Darwin Harbour are generally classified as sand or sand mixed with gravel (Cardno 2014).

The NW Arnhem Land survey revealed macroinvertebrate soft-sediment species richness is high (n=619 species) in the area, and comparable to coastal inshore areas in tropical Australia. Samples comprised primarily of invertebrate species (83%; polychaetes, molluscs and crustaceans) (Russell & Smit 2007). Community composition is very similar to that within Darwin Harbour, with INPEX surveys revealing 87% of taxa recorded were crustaceans, polychaetes and molluscs. Families contributing most to the total abundance in Darwin Harbour are deposit-feeders (ingesting nutrients from particles within the sediment) which are likely due to high organic load resulting from nearby mangroves (Cardno 2015e). The most abundant taxonomic groups in the Beagle Gulf Benthic Survey were crustaceans, molluscs and echinoderms (Smit et al. 2000).

3.1.3 Coral Reefs

While there are no large reef-building coral assemblages in the Northern Territory coastal waters, approximately 250 species of coral inhabit and form unique and extensive coral communities (Veron 2004) which support populations of economically important fish species, such as mackerel and snapper (DEWHA 2007; DNREA 2007). The distribution of corals in the Northern Territory is largely known only from targeted surveys, with known colonies on the Coburg Peninsula, Darwin Harbour, at the Wessel Islands, in the Gulf of Carpentaria and along the Arnhem coast (DNREA 2007).

Largely relic, submerged patch, platform and barrier reefs are known in water depths up to 50m in the Gulf of Carpentaria (DEWHA 2007). They support abundant hard corals (*Leptoseris* species),



plate corals (*Turbinaria* species) and many soft corals (DEWHA 2007). The closed circulation of the Gulf is thought to limit recolonisation and dispersal of corals from the Indo-Pacific (DEWHA 2007).

A study in 1927 of coral reef species documented corals in the Sir Edward Pellew Group, though it is unknown if the diversity of corals in the area still persists as no further studies have been conducted (DEWHA 2007).

A targeted study conducted in Arnhem Land, between Port Bradshaw and the Goulburn Islands, found that all sites were of relatively pristine condition and that the richest coral sites were at least partly protected by wave action (Veron 2004). Given the lack of clear waters, it was rare for corals to form large reef systems (Veron 2004; DNREA 2007). Faviids, in particular *Favia*, *Favites*, *Platygyra*, and *Goniastrea* were the most dominant species and the area contained several species that are common, albeit rare elsewhere in Australia (Veron 2004).

Approximately 50 species of hard corals are known in Darwin Harbour, with four well-developed reef systems in protected areas where high water quality allows systems to form; Channel Island, Wickham Point, Weed Reef and South Shell Island (Fortune and Drewry 2011; Cardno 2015d). Dredge-monitoring of these four areas in Darwin Harbour was conducted by INPEX and subsequently provides extensive baseline documentation of the coral species, colonies, extent and percentage cover of coral in the Harbour in addition to water quality data (Cardno 2015d). The coral monitoring program recorded a total of 48 species of hard corals, from 34 general and 13 families within the Harbour. Coral communities at Channel Island are listed on the Northern Territory Heritage Register, under the *Heritage Act 2011* as a result of the significant diverse coral community that exists at the Island, despite the unsuitable conditions for coral growth. Coral communities in Darwin Harbour are thought to have adapted to the harsh conditions, including high turbidity and low light, high sedimentation rates, elevated water temperatures and extreme low tides (Cardno 2015d).

Threats to coral reefs in the Northern Territory include commercial harvest, oil spills, coastal development leading to increased sedimentation and pollution, and groundings and scouring due to proximity to shipping routes (NOO 2004; DNREA 2007). It is thought however that human impacts currently pose limited threats to the nearshore coastal environment in the Northern Territory (DNREA 2007).

3.1.4 Mangroves

The Northern Territory has the largest extent of mangrove forest in Australia, with 9700 km² of mangrove ecosystems occurring along 4120km² of the 10953 kilometres of coastline (DLRM 2013a; Lee 2003). Thirty six species of mangroves occur naturally in Darwin Harbour alone, making it a significant natural resource and hotspot for mangrove diversity (Lee 2003; Cardno 2015c). The mangrove species *Avicennia integra* is endemic to the Northern Territory (Wightman 2006). There are also five species of mangroves listed as near-threatened; *Avicennia integra*, *Bruguiera sexangula*, *Cerbera manghas*, *Rhizophora lamarckii* and *Xylocarpus granatum* (note that this list is not exhaustive for the entire Northern Territory, rather just the eastern section that falls within the Northern Planning Area (NPA), Figure 3-2) (NOO 2004). There are also seven species of mangroves, with less than 10 known Northern Territory populations; *Amyema thalassia*, *Bruguiera sexangula*, *Cerbera manghas*, *Cynometra iripa*, *Dalbergia candenatensis*, *Lysiana maritime*, *Rhizophora lamarckii* (again note that this list may not be exhaustive, and contain species that only also fall within the NPA) (NOO 2004).

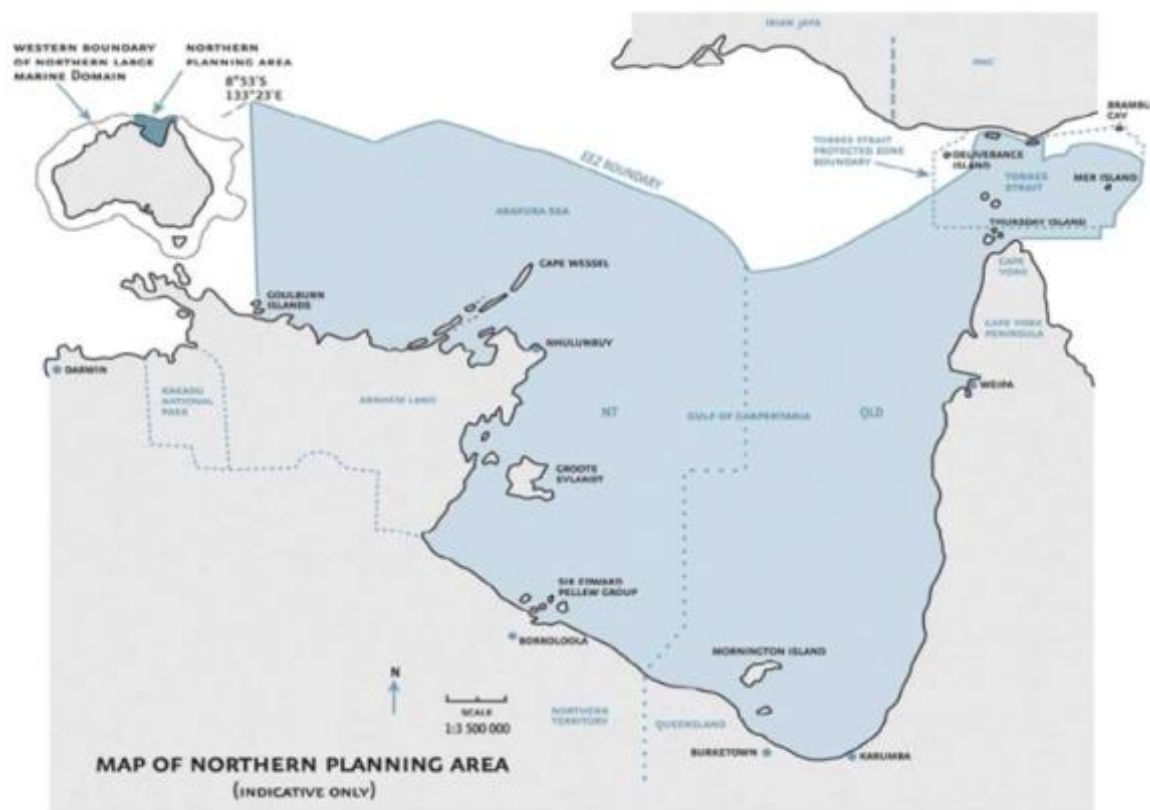


Figure 3-2 : The Northern Planning Area

A detailed list of mangrove species in the Northern Territory including their distribution, description, habitat and features can be found in the Northern Territory Botanical Bulletin No. 31 (Wightman 2006).

Seaward mangrove assemblages live at the interface between land and sea, and subsequently are well adapted to living with natural stressors such as changes in temperature, salinity and anoxia. However this therefore means they may survive close to their tolerance limits and be particularly sensitive to disturbances such as those arising from human disturbance. Mangroves in the Northern Territory are subject to natural tropical climatic seasonal changes, and severe weather events, including cyclones. Large changes in rainfall patterns between the wet and dry season result in expansion and contraction of mangrove extent dependent on rainfall fluctuations in particular regions. For example, a study in Moreton Bay, Queensland, found a significant positive relationship between rainfall and landward mangrove expansion (Eslami-Andargoli et al. 2009).

A more recent example of the variability in spatial extent due to extremes in climate occurred between late 2015 and early 2016 when extensive areas of mangrove tidal wetland vegetation died back along 1000km of the shoreline of Australia's remote Gulf of Carpentaria. The cause is not fully explained, but the timing was coincident with an extreme weather event; notably one of high temperatures and low precipitation lacking storm winds. The dieback was severe and widespread, affecting more than 7400ha or 6% of mangrove vegetation in the affected area from Roper River estuary in the Northern Territory, east to Karumba in Queensland. At the time, there was an unusually lengthy period of severe drought conditions, unprecedented high temperatures and a temporary drop in sea level. Although consequential moisture stress appears to have contributed

to the cause, this occurrence was further coincidental with heat-stressed coral bleaching (Duke et al. 2017).

Due to a relatively low human population, mangroves in the Northern Territory have minimal anthropogenic threats compared with some other parts of Australia. However, increasing use of the coastline means that mangrove condition may deteriorate. Primary threats to mangroves in the Northern Territory include pollution, agricultural runoff leading to increased nutrients and sedimentation, and coastal residential development (and broad scale land-clearing) (Moritz-Zimmermann et al. 2002; Lee 2003).

3.1.5 Seagrass

Seagrasses are a key habitat in the Northern Planning Area (Figure 3-2) which covers the NT coastline between the Goulbourn Islands eastward to the State boundary in the Gulf of Carpentaria for both dugong and marine turtles. Territory-wide distributions of seagrasses is largely unknown, however targeted surveys have found the following:

- Dominant species in Darwin Harbour are *Halodule uninervis*, *H. pinifolia* and *Halophila decipiens* (Fortune and Drewry 2011; Cardno 2015a);
- Six species have been identified at Goulbourn Islands, Castlereagh Bay: *Enhalus acoroides*, *H. uninervis*, *Halophila decipiens*, *H. ovalis*, *Thalassia hemprichii* and *Thalassodendron ciliatum* (McKenzie 2008);
- Melville Bay largely contains *Halophila* spp, though others are reported (McKenzie 2008);
- Eight seagrasses identified in the western Gulf of Carpentaria, of which *H. ovalis* and *H. uninervis* dominate intertidally and *Cymodocea serrulata* and *Syringodium isoetifolium* dominate subtidally (McKenzie 2008);
- 15 species of seagrass have been recorded in the Northern Planning Area (which includes Queensland's east coast of the Gulf of Carpentaria): *C. rotundata*, *C. serrulata*, *H. uninervis*, *H. pinifolia*, *S. isoetifolium*, *T. ciliatum*, *Enhalus acoroides*, *H. decipiens*, *H. minor*, *H. ovalis*, *H. spinulosa*, *H. tricostrata*, *T. hemprichii* and *Zostera capricorni* (NOO 2004);
- Extensive meadows occur in areas of Blue Mud Bay, around Groote Eylandt and its offshore islands, to the north and south of the Roper River delta and inshore of the Sir Edward Pellew Group (DEWHA 2007);
- Important seagrass beds exist in the Limmen Bight Marine Park, particularly east of the Limmen Bight River and between the Roper and McArthur Rivers (Delaney 2012); and
- Seagrasses are reportedly on the western side of Dorchester Island and Yelcher Beach, though the extent and species are not clear (Guinea 2004).

Two broad studies have documented the distribution of seagrasses over large scales:

- *Halophila* is the dominant genus in surveys conducted in the Van Diemen Gulf and Arnhem Land (see Russell and Smit 2007 for specific details of seagrass distribution in the areas); and
- A broad survey of the intertidal regions from the eastern Van Diemen Gulf round to the east coast of the Gulf of Carpentaria (see Roelofs et al. 2005).

Given the water column in coastal waters is quite turbid, the depth to which seagrass can grow is somewhat restricted (DEWHA 2007) and also difficult to map. Key threats to seagrasses in the Northern Territory are those relating to climate events or climate change as seagrasses are sensitive to nutrients, light (and sedimentation) and physical disturbance (Roelofs et al. 2005; Erftemeijer and Robin Lewis 2006).

3.2 Protected Species and Areas

Coastal waters of the Northern Territory are home to a range of protected species, including whales, sharks, saltwater crocodiles, turtles, dugongs, dolphins, fishes and marine and migratory birds. The location of important sites in coastal waters (and land) in the North Marine Region for these species is shown in Figure 3-3).

An *Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act)* Protected Matters Search was conducted. The report listed 17 threatened marine species, in addition to many additional species that are listed as protected under the Act. Key groups of protected species are further discussed below.

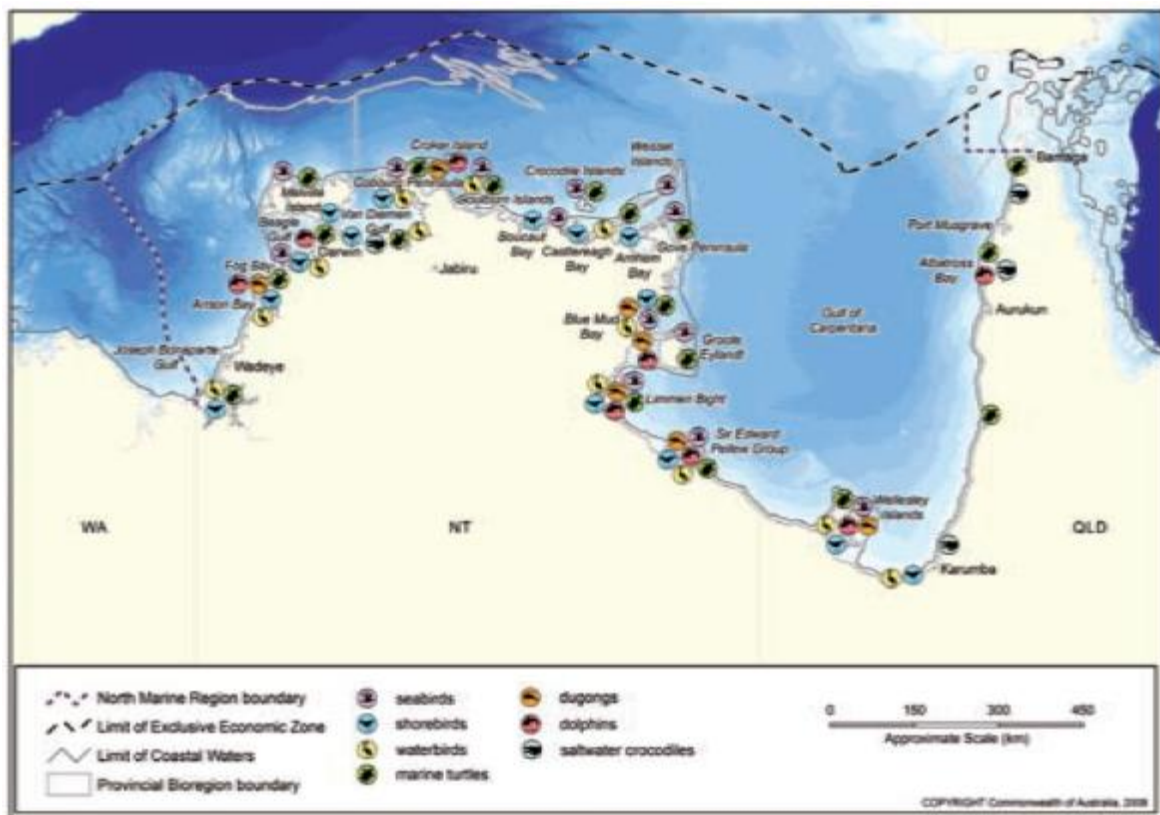


Figure 3-3 : Important coastal and land sites adjacent to the North Marine Region for birds, marine turtles, dugongs and dolphins (DSEWPac 2008)

3.2.1 Whales

Two whales are listed as threatened and migratory under the EPBC Act; the Blue Whale and Humpback Whale (Table 3-1). These species travel north on their migration along the west coast of Western Australia from May to mid-August and back south from October to November and December. As they are often with their calves on the return journey, Humpback whales tend to transit closer to shore during the southern migration. Northern Territory waters are not known to be part of the migration route, as the primary calving area is at Camden Sound in Western Australia. The migration patterns of blue whales are less well known due to their preference for deeper, offshore waters (DoE 2015a).

Table 3-1 : Conservation Status of Whale species in the Northern Territory

Species	Common name	EPBC Status	Type of presence	Territory Status
<i>Balaenoptera musculus</i>	Blue Whale	Endangered	Species or species habitat likely to occur within area	Data deficient
<i>Megaptera novaeangliae</i>	Humpback Whale	Vulnerable	Species or species habitat likely to occur within area	Not listed

3.2.2 Turtles

Six of the world's seven marine turtles can be found in the waters of the Northern Territory, and all six are listed as threatened under the *EPBC Act* (Table 3-2):

- The loggerhead turtle tends to occur in waters of coral and rocky reef, seagrass beds and muddy bays and nesting areas have been recorded from Shark Bay to the North West Cape on the west coast of Australia (DoEE 2017a). There are no known breeding areas in the Northern Territory (DNREA 2003). Foraging areas however, are known to be wider (DoEE 2017a).
- Green turtles forage and travel in relatively shallow waters, less than 25 m deep (Chevron Australia 2012). Seven regional populations have been identified, one of which is in the Gulf of Carpentaria (approximately 5000 individuals) (DoEE 2017a) where it is known to forage on submerged rocky reefs and nest on sandy coastal beaches and offshore islands on the north-central west coast (east of the Wessel Island chain, to the south of Groote Eylandt) (DEWHA 2007).
- The leatherback turtle spends most of its time in the open ocean, migrating to breeding grounds in neighbouring countries (Indonesia, Papua New Guinea, and Solomon Islands) (DoEE 2017a). Isolated nesting has been recorded in the Northern Territory (DoEE 2017) at the Sir Edward Pellew Islands, near Maningrinda, Danger Point on Cobourg Peninsula and Palm Bay on Crooker Island. The population in the Northern Territory is thought to be less than 50 mature individuals (DLRM 2013b).
- The hawksbill turtle generally forages and travels in shallow waters less than 10 m deep (Chevron Australia 2012). Key nesting sites in the NT include the Coburg Peninsula, East Arnhem Land, Groote Island, Sir Edward Pellew Islands and Wessel and English Islands (DoEE 2017). They are also known to forage on the submerged rocky reefs on the north central west coast of the Gulf of Carpentaria and nest on sandy coastal beaches and offshore islands (DEWHA 2007).
- The olive ridley turtle prefers soft-bottom habitats. The only tracking study in Australia (study conducted in NT) has shown individuals do not leave the continental shelf, with all tracked females (n=5) all migrating in different directions (DoEE 2017). Nesting has been recorded in the Northern Territory along the Arnhem Land Coast (including Crocodile, McCluer and Wessel Islands, Grant Island and Coburg Peninsula) (DoEE 2017). No major breeding areas have been recorded in Australia, though nesting is known to occur on sandy coastal beaches and offshore islands on the north central west coast of the Gulf of Carpentaria (DEWHA 2007). Olive Ridley's appear to undertake solitary or low-density nesting. It is expected that nesting females in Australia is in the order of a few thousand annually (DoE 2015b).
- Flatback turtles forage and travel in water depths less than 70 m deep (Chevron Australia 2012). Flatback turtles are the most widely-nesting turtle species in NT with the majority of

nesting occurring on islands (DoE 2015). Migration tends to be restricted to the continental shelf (DoE 2015) Hatchling emergence period - December to April.

Table 3-2 : Conservation Status of Turtle species found in the Northern Territory

Species	Common name	EPBC Status	Type of presence	Territory Status
<i>Caretta caretta</i>	Loggerhead Turtle	Endangered	Foraging, feeding or related behaviour known to occur within area	Vulnerable
<i>Chelonia mydas</i>	Green Turtle	Vulnerable	Breeding known to occur within area	Near threatened
<i>Dermochelys coriacea</i>	Leatherback Turtle	Endangered	Breeding known to occur within area	Critically Endangered
<i>Eretmochelys imbricata</i>	Hawksbill Turtle	Vulnerable	Breeding known to occur within area	Vulnerable
<i>Lepidochelys olivacea</i>	Olive Ridley Turtle	Endangered	Breeding known to occur within area	Vulnerable
<i>Natator depressus</i>	Flatback Turtle	Vulnerable	Breeding known to occur within area	Not listed

3.2.3 Sharks, Rays and Sea Snakes

The North Marine Region is an important area for sea snakes, with all nineteen species known to occur in the region listed under the *EPBC Act* (DSEWPaC 2012a). Due to their low fecundity and slow growth rates, dredging (physical habitat modification) pressures in the North Marine Region have been listed as of 'potential concern' for sea snakes (DSEWPaC 2012a).

There are 36 known species of rays in the Northern Territory, none of which are listed as protected under the *EPBC Act* or under the *Territory Parks and Wildlife Conservation Act 2000* (NOO 2004). Those ray species that are dependent on reefs and corals may be threatened by habitat loss (NOO 2004).

Four species of shark that may occur within Northern Territory coastal waters are protected under the *EPBC Act* (Table 3-3):

- Great white sharks can be found from close inshore around rocky reefs, surf beaches and shallow coastal bays to outer continental shelf and slope areas (DoEE 2017c). They also make open ocean excursions and can cross ocean basins (DoEE 2017c). While the EPBC Report lists Great White Sharks as likely to occur within the area, there are no records of Great White Sharks in the Northern Territory (DoEE 2017c).
- The Northern River Shark is found in brackish waters and there are only few records of them in the Northern Territory; the Adelaide River and East and South Alligator River systems of NT (DLRM 2012a). Little is known about the species.
- The Speartooth shark is recorded only in tidal rivers and estuaries in NT; Adelaide River, South, West and East Alligator Rivers, Murganella Creek and Marrakai Creek (DoEE 2017). They exhibit cyclic behaviour, moving downstream at ebb tide and upstream with the flood tide (DLRM 2012b, DoEE 2017d).



- Whale Sharks are found in all tropical seas though most commonly seen in waters off northern Western Australia, Northern Territory and Queensland (DoEE 2017b). Ningaloo Reef (Western Australia) is the main known aggregation point in Australian waters. Whale shark presence coincides with the coral mass spawning period, when there is an abundance of food (krill, planktonic larvae and schools of small fish) in the waters adjacent to reef.

Table 3-3 : Conservation Status of Shark species found in the Northern Territory

Species	Common name	EPBC Status	Type of presence	Territory Status
<i>Carcharodon carcharias</i>	Great White Shark	Vulnerable	Species or species habitat likely to occur within area	Not listed
<i>Glyphis garricki</i>	Northern River Shark	Endangered	Breeding known to occur within area	Endangered
<i>Glyphis glyphis</i>	Speartooth Shark	Critically Endangered	Species or species habitat known to occur within area	Vulnerable
<i>Rhincodon typus</i>	Whale Shark	Vulnerable	Species or species habitat known to occur within area	Not listed

3.2.4 Avifauna

The Northern Territory coastline supports significant colonies of shorebirds, seabirds and water birds. Extensive mudflats along the Gulf of Carpentaria coastline provide habitat for primary prey for thousands of resident and migratory shorebirds and waterbirds, with more than 270 species of seabirds and waders recorded from the area (DSEWPaC 2008), a number of which are protected as migratory marine species by the *EPBC Act* and the *Territory Parks and Wildlife Conservation Act 2000*. Some of the more prominent protected bird species include terns and noddies, gulls and jaegers, sandpipers, plovers, boobies, frigate birds and tropicbirds, raptors, shearwaters, and egrets, herons and ibis (DSEWPaC 2008).

The Northern Territory also contains 31 Important Bird Areas (IBAs), along the coastline, islands and inland (Figure 3-4) (Birdlife International 2009). The migratory lifestyle of some birds can lead to difficulties in conservation and subsequently identification of these IBAs allows managers to focus on the protection and use of the habitat.

Australia forms part of the East Asian-Australasian Flyway; a geographic region that supports significant populations of migratory waterbirds during their annual migration (Bamford et al. 2008). In particular, Chatto (2003) found that peak numbers of eastern curlew, marsh sandpiper and grey-tailed tattler shorebirds utilise Northern Territory sites as staging points. Seabirds, shorebirds and waterbirds are likely to be most affected by seabed mining where habitat degradation occurs. Habitat loss and direct disturbance are key threats for avifauna (Bamford et al. 2008), for example coastal raptors nest in the mangroves on a number of islands (Chatto 2001) and subsequently may affect nesting behavior of some raptor species. Furthermore, foraging is critical to migratory species, particularly after long-distance migration, where body weight is at a minimum.

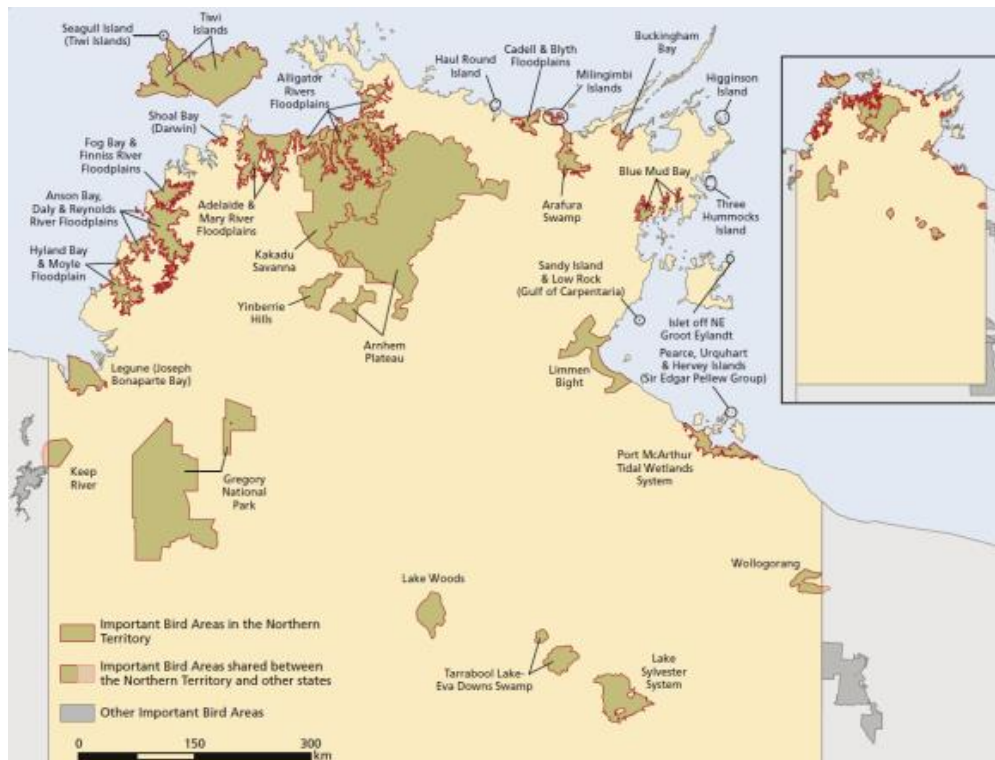


Figure 3-4 : Important Bird Areas (IBAs) of the Northern Territory (from Birdlife International 2009)

3.2.5 Sawfishes

Three species of sawfish are listed under the EPBC Act and the Territory Parks and Wildlife Conservation Act 2000 are known to occur in the coastal waters of the Northern Territory (Table 3-4):

- The green sawfish is the most common sawfish in Australian waters (DNREA 2006), yet its habitat requirements and abundance is largely unknown (DSEWPac 2008). Juvenile and sub-adult green sawfish utilise estuarine and marine coastal waters, and large adults have been recorded in coastal waters and offshore to the 70m depth contour (DSEWPac 2008).
- Dwarf sawfish occur in shallow waters (2-3m) in coastal and estuarine waters of tropical Australia (DLRM 2012). Little is known about the dwarf sawfish, however they are known to migrate to inshore waters. Offshore captures are rare so their distribution is poorly understood (DSEWPac 2012b; DoEE 2017e).
- The largetooth sawfish is both a marine and freshwater species, known to occur in freshwater as a juvenile and when 4-5 years old migrate to estuarine and offshore waters up to 25m deep (DNREA 2006; DoEE 2017e). Popping is thought to occur late in the wet season in the Gulf of Carpentaria (DSEWPac 2012b).

Table 3-4 : Conservation Status of Sawfish species found in the Northern Territory

Species	Common name	EPBC Status	Type of presence	Territory Status
<i>Pristis clavata</i>	Dwarf Sawfish	Vulnerable	Breeding known to occur within area	Vulnerable
<i>Pristis zijsron</i>	Green Sawfish	Vulnerable	Species or species habitat known to occur within area	Vulnerable
<i>Pristis pristis</i>	Large-tooth Sawfish	Vulnerable	Breeding known to occur within area	Vulnerable

3.2.6 Other Listed Species

Several marine species, while not listed as threatened under the *EPBC Act*, are still protected by the Act. These key groups are briefly discussed below.

3.2.6.1 Dolphins

The Australian snubfin dolphin, Indo-Pacific humpback dolphin and Indo-Pacific bottlenose dolphin are known to occur in the North Marine Region (DSEWPaC 2012a). All three species are listed as migratory and marine under the EPBC Act. These species rely on the waters of the North Marine Region and adjacent coastal areas for breeding and foraging (DSEWPaC 2012a). The Australian snubfin dolphin and Indo-Pacific humpback dolphin occur mostly in shallow waters up to 10 km from the coast and 20 km from the nearest river mouth. Indo-Pacific bottlenose dolphins tend to occur in deeper, more open coastal waters, primarily in continental shelf waters (up to 200 m deep), including coastal areas around oceanic islands (DSEWPaC 2012a). Key threats to dolphins include physical habitat modification, noise pollution and chemical pollution.

3.2.6.2 Dugong

Most of the world's dugong (*dugong dugon*) populations are between Shark Bay in Western Australia and Moreton Bay in Queensland (NOO 2004). Dugong are protected under both the *EPBC Act* and are listed as threatened under the *Territory Parks and Wildlife Conservation Act 2000*. They are long-lived, with slow reproduction rates and long gestation periods and rely heavily on shallow tidal and subtidal seagrass habitats for foraging, though have also been recorded at the edge of the continental shelf (NOO 2004; DEWHA 2008). The primary threat to dugong is habitat loss.

3.2.6.3 Saltwater Crocodiles

The saltwater crocodile is an EPBC Listed marine species. The salt-water crocodile is found in Australian coastal waters, estuaries, lakes, inland swamps and marshes (Webb et al. 1987). In the Northern Territory the salt-water crocodile has been found in the following rivers: Mary, Adelaide, Daly, Moyle, Victoria/Baines, Finnis, Wildman, West Alligator, East Alligator, South Alligator, Liverpool, Blyth, Glyde, Habgood, Baralminar/Gobalpa, Goromuru, Cato and Peter John Rivers (Fukuda et al. 2007). Studies from Arnhem Land (Northern Territory) indicated that the Salt-water Crocodile mostly occurs in tidal rivers, coastal floodplains and channels, billabongs and swamps (Webb et al. 1987) up to 150 km inland from the coast (Webb et al. 1983) although they have also

been recorded kilometres offshore on small coastal islands. The primary threat to crocodiles is habitat loss.

3.2.7 Fish Protection Areas

The Moyle/Port Keats Reef Fish Protection Area (RFPA) includes Emu Reef and Howland Shoals. It is designed to protect known aggregations of black jewfish (*Protonibea diacanthus*) and golden snapper (*Lutjanus johnii*) and to provide proactive protection to more pristine stocks west of Darwin that may be supplying recruits to Anson Bay, Peron Islands and the Dundee/Fog Bay area. The establishment of the RFPA in 2013 was in response to the outcomes of an ecological risk assessment on a number of common coastal reef fish stocks in the Darwin marine / coastal area undertaken in 2009, and the subsequent NT DPIF consultation papers in 2012 and 2013 (DPIF 2012, 2013).

Other RFPAs are located on Bathurst Island, Melville Island, Charles Point Wide, Lorna Shoal and Moyle and Port Keats.

3.2.8 Commercial Fishery and Aquaculture

There are 11 main wild catch fisheries operating in the Northern Territory, each requiring separate licences and each operating under their own gear restrictions, defined area and management regulations: Aquarium, Barramundi, Coastal Line, Coastal Net, Demersal, Mud Crab, Offshore Net and Line, Spanish Mackerel, Timor Reef and Trepang.

The two major aquaculture activities include Pearl Oyster (*Pinctada maxima*) culture and Barramundi farming (*Lates calcarifer*). Other products include sea cucumber (trepang), giant clams and freshwater plants. Sea cucumber 'ranching' occurs on Goulburn Island and Groote Eylandt, with hatchery-produced juveniles used to restocked suitable areas at sea.

Red-legged banana prawns, tiger prawns (*Penaeus esculentus*) and blue endeavour prawns (*Metapenaeus endeavouri*) are the key prawn species targeted in the Northern Prawn Fishery (NPF).

The majority of these fisheries operate in coastal offshore waters. However, a number of fisheries rely on freshwater and estuarine environments of northern Australia, including the aquarium fishery.

3.2.9 Indigenous Protected Areas

Indigenous Protected Areas (IPAs) are part of Australia's National Reserve System - a nation-wide network of reserves especially set up to protect examples of Australia's unique landscapes, plants and animals for current and future generations. IPAs are areas of indigenous-owned land and/or sea where traditional owners have entered into a legally binding agreement with the Commonwealth Government to promote biodiversity and cultural resource conservation. The majority of IPAs are declared as mainly International Union for Conservation of Nature (IUCN) Category V protected areas, to protect land and seascape values and/or Category VI protected areas, managed mainly for the sustainable use of natural ecosystems (DoE 2015b).

IPAs in the Northern Territory with a significant marine and coastal component include:

- Anindilyakwa Indigenous Protected Area (Groote Eylandt Archipelago)



- Dhimurru Indigenous Protected Area (western edge of the Gulf of Carpentaria)
- Djelk Indigenous Protected Area (central Arnhem Land to the Arafura Sea)
- Laynhapuy Indigenous Protected Area (north east Arnhem Land)
- Yanyuwa Indigenous Protected Area (central Gulf covering Sir Edward Pellew Islands)



4 Review of Environmental Impacts

Comprehensive studies and reviews have been conducted assessing the physical, biological and ecological impacts of various seabed mining techniques (e.g. Charlier 2002; Byrnes *et al.* 2004; Birkland and Wijsman 2005; Penney *et al.* 2008; Krause *et al.* 2010; Newell and Woodcock 2013). Exploration, mining, processing and the transport of mined material all have the potential to impact the marine environment as summarised in Table 4-1.

Table 4-1 : Overview of Potential Effects on the NT Marine Environments

Effect	Corals/Seagrass	Mangroves	Benthic Fauna	Megafauna
Direct Effects				
Removal of Habitat	High	High	High	High
Change in Bathymetry	High	High	High	Low
Change in Sediment Characteristics	Variable	Variable	High	Low
Noise	Low	Low	Low	High
Entrainment	Low	Low	Low	High
Indirect Effects				
Change to Hydrodynamics	High	High	High	High
Change to Coastal Processes	High	High	High	Variable
Turbidity	High	Low	Variable	Variable
Sedimentation	High	Low	Variable	Variable
Contaminants	High	High	High	High
Light Emissions	Low	Low	Low	High
Spills	High	High	High	Variable
Marine Pests	Variable	Variable	Variable	Variable



4.1 Direct Physical Impacts

The physical impact of coastal seabed mining will be dependent on the method of removal, quantity and grade of deposit desired or rejected and any overspill or discharge of unwanted material (Newell *et al.* 1998). Direct physical impacts resulting from seabed mining include changes in seabed topography and changes in sediment characteristics. The effects on the water column are considered in greater detail in Section 4.4.3.

4.1.1 Changes in Seabed Topography

Mining of the seabed requires mechanical excavation and removal of the benthos with the desired material to be processed. Extraction of any part of the benthos will alter the existing topography and/or bathymetry of the seabed at various scales. Stationary extraction of coastal deposits by bucket or suction dredges occurs in a series of specific locations, leaving pits or depressions on the seabed (Figure 4-1) (Newell *et al.* 1998; Kim and Lim 2009). These artefacts may have diameters exceeding 100m, and burrow depths of greater than 10m (Kim and Lim 2009; Uscinowicz *et al.* 2014). In an extreme case of alterations of seabed topography, field surveys in Kyunggi Bay, Korea, show dredge pits were in the order of 2-3 km wide and 10 to 15m deep (Kim and Lim 2009). This is considerably larger than those reported elsewhere.

Dredged pits are likely to become persistent features, particularly in low-energy systems (as typically occur in NT waters), as in higher energy systems sands are mobile and able to infill depressions (Newell *et al.* 1998; Byrnes *et al.* 2004; Manso *et al.* 2010). Recovery and regeneration of pits will be contingent on the type of material that has been extracted, the magnitude of features created, water depth and the hydrodynamics of the affected system (Steele *et al.* 2010). Ecological recovery and aspects relating to recolonisation of habitat are discussed in detail in Section 5.5.1.

Uscinowicz *et al.* (2014) studied the impact of stationary dredging in water depths between the 15 and 20m contours, before, directly after and 11 months following sand dredging of Polish waters in the Baltic Sea. Distinct pits were evident following dredging, with diameters ranging between 80-120 m and between 3-4.5m deep. These pits, 11 months after the dredging event, were shallower by 2-2.5m, however the diameter had increased by 40-50m with the overall volume only approximately 3.5% smaller than directly after dredging. It was evident that the filling of the pits was primarily due to the slipping of the pit slopes, rather than infill from mobile sediments in the area. To the contrary, sand mining along the Pakiri-Mangawhai coast, New Zealand, occurs in water depths of 3-8m using mechanical excavators (Hesp and Hilton 1996). Due to the shallow depth of extraction, high swell and wave activity reportedly led to replenishment over a period of hours to days (Hesp and Hilton 1996).

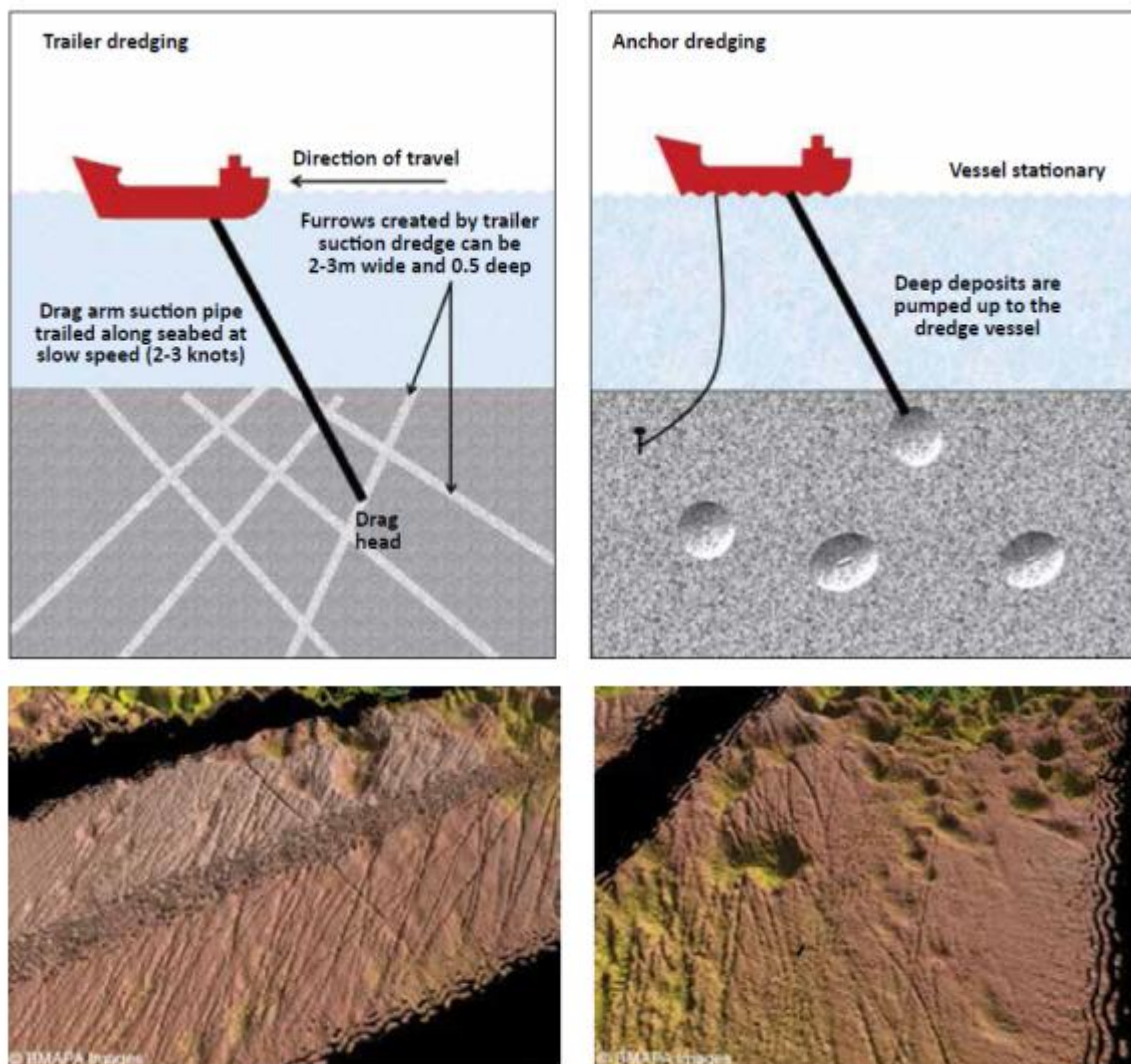


Figure 4-1 : Schematic of (top) and changes to the topography (bottom) resulting from trailer dredging (left) and stationary dredging (right) (from Newell and Woodcock 2013)

Similar observations of infill rates have also been reported in the south Baltic Sea at the Tromper Wiek, where hopper dredge pits in sandy sites refilled quicker than the gravel sites (Kubicki et al. 2007). As in Uscinowicz et al (2014), refilling rates at Tromper Wiek were related to wall steepness, and the diameter of the pits increased regardless of the infill rates. Manso *et al.* (2010) report that pits in Tromper Wiek regenerate more rapidly in the years immediately following extraction, however the marks were still detectable some 10 years later.

Trailer dredging (Figure 4-1) creates furrows and undulations in the seabed that are significantly shallower and more narrow than stationary dredges, but may occur over a larger area of the seabed (Newell *et al.* 1998; Uscinowicz *et al.* 2014). Furrows are typically 2-3m wide, and 0.2-0.5m deep (Uscinowicz *et al.* 2014). A study assessing the impact of marine aggregate extraction in Dieppe, France found that topography was significantly altered by dredging (Desprez 2000). Changes included introduction of 'megaripples' in the substrate, in addition to large furrows separated by crests of shingles. Furrows as deep as 5m (where dragheads have repeatedly followed the same path) were found to have persisted in areas after several years, though some recovery was evident due to the strong tidal current in the area (Desprez 2000). Dredge tracks generally

recover more quickly in dynamic mobile sand areas than in calm, deep more static environments (Birkland and Wijsman 2005).

The complete recovery of the benthos following seabed mining is varied. A summary of the recovery times at a range of marine aggregate extraction sites within the UK following dredging is reported by Hill *et al.* (2011) (as cited in Newell and Woodcock 2013). Times range between months and decades depending on the hydrodynamics of the area, dredging intensity and severity of impact. Seabed monitoring in Darwin Harbour during the Ichthys project found that abundance and diversity of subtidal benthos at the spoil ground had recovered within 13 months of completion of dredging (Cardno 2015e). However, it is important to note that monitoring at the excavation areas was not required or undertaken, so no commentary can be provided regarding benthic recovery inside the navigation channels. Although aspects of the Inpex monitoring program may not have been adequate due to the lack of baseline data and absence of reference sites, the conclusion provided (including the timing to recovery) has been substantiated by numerous other studies that assessed benthic recovery following seabed disturbance.

Natural sand formations such as dunes, ripples and ridges could also be removed or altered by seabed mining. Seabed dunes in Gyeonggi Bay, Korea, were surveyed from 2006 to 2008 to assess changes in dune morphology as a result of mining (Kum *et al.* 2010). These surveys demonstrated that an interaction exists between tidal currents and a decrease in sediment availability as a result of sand mining, which together led to a decrease in dune height. Furthermore, the seabed became irregular in shape and was both narrower and shorter.

Modified bathymetry has been shown to change nearshore wave and current patterns which have the potential to affect coastal shore stability due to erosion of the seabed, though this is likely to be limited to shallow waters only (i.e. less than 20-25m) (Hesp and Hilton 1996; Byrnes *et al.* 2004; Birkland and Wijsman 2005) (see Section 4.2). In addition, removal of the top surface layer will have direct ecological effects, given the majority of fauna are typically found within the top 30-40 cm of sediment (Phua *et al.* 2004; Birkland and Wijsman 2005), the impacts of which are discussed further in Section 4.3.1.

4.1.2 Changes in Sediment Characteristics

Seabed mining targets specific seabed particle fractions, which depends on the aggregate being mined. As mentioned in Section 2.4.3, it is economic to load and transport only the fraction of the aggregate that is required, and subsequently it is common to screen for unwanted fractions and return them to the seabed (Hitchcock and Bell 2004) (Figure 4-2). The removal of particular sized particles potentially changes the composition of the surface layer of sediments, depending on the fraction removed and the energy of the system it is removed from. Further implications arising from screening practices include changes in water quality, turbidity, biogeochemical cycles and increased sedimentation (see Section 4.4.1-4.4.3).



Figure 4-2 Screened sediment is returned to the water column (image courtesy of British Marine Aggregate Producers Association) (from Newell and Woodcock 2013).

A study was conducted by deJong *et al.* (2014) on the impacts of sand extraction between 2009 and 2013 relating to the harbour extension at the Port of Rotterdam in the Netherlands. They found that median grain size decreased in sand extraction sites and the fraction of very fine sands increased. Similar results were found in a study conducted along the 20m contour of the French coast off Dieppe, following monitoring (over a 10 year period) of an area subject to marine aggregate extraction (Desprez 2000). Sediments in the reference area were dominated by gravels and shingles, while sediments in the dredging and deposition areas were dominated by fine sands, as extraction progressively removed the original sandy gravel

The infill of pits and depressions caused from seabed mining are commonly dominated by the fine sediments that are capable of mobilisation due to the shear stress induced by waves and tidal currents (Newell *et al.* 1998). An example of this was reported in China's Pear River Estuary, where sand mining is conducted via water jets. This involves injection of high pressured water through a tube that has been inserted into the seabed ejecting sediment back through the tube onto the deck (Tang *et al.* 2011). Filtering of the particles was undertaken on board the vessel whereby gravel, shells and other coarse material were returned to sea and very fine particles were removed due to rinsing, so that only medium to coarse sand remained. Of the material returned to sea, the larger fractions settled on the seabed quite rapidly, while the finer particles entered the water column and became trapped in the pits created by the mining operations. Similar findings were reported in Uscinowicz *et al.* (2014) where fine sands were evident at the bottom of pits caused by stationary dredging in the Baltic Sea, combined with larger fractions from slope slippage. Uscinowicz *et al.* (2014) noted that over time, the finer sediments had become dispersed, so that the surface layers were covered with deposits similar to the pre-existing seabed.



Cooper *et al.* (2005) in their study off the south-east coast of England, UK, found that the sediment characteristics of mined areas may also be dependent on the intensity of dredging. Low impact areas were comparable to reference sites and sediments changed very little over the course of the four year investigation. However, samples from the heavily dredged areas were fine with less gravel and more coarse sand, before the gravel proportion increased and coarse sand decreased over time. The differences in sediment composition resulting from seabed mining are likely due to the composition of the extraction sites and the hydrodynamics of the areas affected (Uscinowicz *et al.* 2014).

4.2 Indirect Physical Impacts

Changes in physical dynamics are greatly dependent on the local conditions and the shape and size of the changes in bathymetry (Bray 2008). Movement of the water (such as from tides, currents and waves) creates energy that is dispersed and absorbed due to friction created by the sediment-water interface, which is altered when the pathway of the water changes (such as deepening of the ocean floor) (Bray 2008). An increase of water depth allows faster propagation of larger waves which allows it to move further inland (Bray 2008), potentially causing erosion of beaches and undermining shore stability. The potential impacts of changes in bathymetry will be largest in areas where large tidal ranges exist, with large waves and will be in proportion to the amount of energy contained in the system (Bray 2008). This is also of particular relevance to the NT marine environment given that most of the NT coastline is subject to a large tidal range in the order of 8 metres.

4.2.1 Changes to Wave and Tidal Patterns

Changes in bathymetry and topography due to seabed mining have the potential to impact hydrodynamics due to the changes in tidal currents, local flows, hydrology and sediment transport patterns of the area (McCook *et al.* 2015). Alterations in nearshore waves and currents have been shown to occur due to bathymetric changes, albeit bathymetry typically affects the surface hydrodynamics landward of the 20-25m depth contours (Hesp and Hilton 1996; Byrnes *et al.* 2004; Birkland and Wijsman 2005). More specifically, Bray (2008) suggests that the wave regime is affected by increasing water depth in the open sea, when the water depth is less than half of the wavelength.

An assessment of the impact coral flat excavation pits have on wave processes at Majuro Atoll, Marshall Island was conducted by Ford *et al.* (2013). Coastal zone protection is critical at the atoll, and reef is thought to be essential in protecting the coast from inundation during storm events. Pits in the coral flats (ranging from 15-35m wide) have been excavated as a source of armour stone and aggregate for construction, generally limited to inner sections of the reef. The study showed that wave energy was disrupted within the excavation pit and on the shoreline adjacent to the pit compared to the unmodified reef. The authors cautioned against applying their findings in a broader context, because wave energy attenuation and infragravity (IG) wave amplification would likely differ based on different reef morphologies, pit designs, and wave conditions. In the context of seabed mining, this highlights the importance of undertaking site-specific investigations to understand the potential hydrodynamic impacts caused by excavation pits on the seabed.

Byrnes *et al.* (2004) assessed the physical impacts of sand mining for beach replenishment on the New Jersey (USA) outer continental shelf (20km from shore and 10-20m deep). Potential dredging impacts at offshore borrow sites, involving between 2.1 and 8.8 million cubic metres of sand from seven locations were determined by wave modelling to estimate refraction, diffraction, shoaling,



and wave breaking along the Jersey coastline. The pre- and post-dredge modelling found that maximum wave height increased by 0.3m (approximately 20% increase compared to existing conditions) at the extraction sites, however wave heights dissipated and consequently wave heights reaching the coast were only 0.1m larger than existing conditions. They found that a steeper shore-face located to the north allowed more wave energy to reach the coastline than gently sloping areas to the south. The magnitude of change was also dependent on the quantity of sediment being extracted, in addition to the orientation of the site to the coastline. It is important to note that these findings were based on post-dredge modelling not post dredge monitoring, which would normally be undertaken to validate the findings.

deBoer *et al.* (2011) conducted a model of the impacts of a very large sand extraction trench on tidal dynamics for the Netherlands Continental Shelf. The trench was located along the 20m depth contour, approximately 12nm from the coastline. An idealised hydrodynamic model (with very simple basin geometry) was created with characteristics typical for the North Sea for a set of trenches with varying dimensions (in the order of hundreds of kilometres long, and width over 10 kilometres, and depth of several metres). The modelling confirmed changes in the coastal tidal range and currents, in the order of centimetres and centimetres per second respectively. Although small in magnitude, the change had significant flow-on effects throughout the whole basin, with 4-5% changes in tidal currents up to 10 kilometres from the coast and changes in tidal ranges, phases and currents affected up to tens of hundreds of kilometres from the extraction trench. The model also confirmed that the magnitude of impact increases with increasing extraction volume, with particular sensitivity to changes in trench width and depth, rather than length. Such changes to tidal currents may alter long-term sediment transport rates along the entire coast.

The study by deBoer *et al.* (2011) is of less relevance to the NT marine environment as mining is unlikely to be undertaken at the scale described above, however it does reinforce the importance of using modelling as a predictive tool to improve understanding of the likely changes that may occur to coastal processes as a result of changes to the seabed.

4.2.2 Changes in the Sediment Regime, Erosion and Coastal Shore Stability

Brampton and Evans (1998) identified five potential effects caused by offshore mineral aggregate extraction: (a) beach draw-down; (b) modifications to tidal currents; (c) changes in sediment transport; (d) variations in nearshore wave conditions; and (e) a reduction in shelter provided to the coastline. The most direct effect of mining activity on the coastline is beach draw-down, which can occur when material is extracted from within the active beach profile. Beach sand or gravel, transported seawards during storm events, remains trapped in the depression caused by the mineral extraction. It will not return to the upper beach during calm conditions, thus resulting in a net loss of beach material.

Hesp and Hilton (1996) assessed the coastline of Pakiri-Mangawhai in New Zealand, where sand mining occurs within the active nearshore system in depths of 3-8m. The impact of the pits formed from mechanical dredging are reversed in a matter of hours to days from the high energy swell and waves present in shallow waters (Hesp and Hilton 1996). The authors assessed the sediment processes occurring in the region (from the coastline to the farshore system) and concluded that it is likely that the coastline exhibits weak and inconsistent post-storm recovery due to sand mining occurring in the region as sediment is removed from the nearshore system that should otherwise be utilised to replenish beaches following storm events.



The modelling study conducted by Byrnes *et al.* (2004) identified that under normal wave conditions, offshore dredging (20km from shore, between the 10-20m depth contour) would change the longshore sediment transport by an average of 10% above existing conditions. These predicted changes were expected to have minimal impact along the shoreline. However, a study in the German Baltic Sea, found that small changes in bathymetry arising from sand extraction caused significant modifications of sediment transport along the coast and thus altering the patterns of erosion and accretion (Kortekaas *et al.* 2010). The differences may be related though to the depth in which extraction is occurring.

An Australian study from Moreton Bay, Queensland found that it was possible to dredge offshore sandbanks without causing erosion of beaches. Sand mining from the East Bank of Moreton Bay in 1983 for construction of Brisbane's airport removed an entire section of the sandbank and monitoring of the bank demonstrated that the bank had largely reformed by 1989 (Pattiaratchi and Harris 2002). However, current meters in the region, show that sand is supplied to the bank from the north, and Pattiaratchi and Harris (2002) caution that continued removal of sand from these areas could directly lead to erosion of Moreton Island's western beaches. It is suggested that to dredge sandbanks without affecting beach erosion, it would need to be demonstrated that there are no direct links between the sandbank system and coastal deposits.

To assess the impact of sand mining in the near-shore coastal zone of Southern Monterey Bay, US, Thornton *et al.* (2006) compared shoreline dune erosion rates during sand mining with rates after mining ceased. Storm waves and tides undercut the base of the dunes, affecting dune stability and subsequently caused the dune to slump onto the beach. Erosion rates varied between 0.5m per year to 1.5m per year along the coast. Longshore sediment transport is generally to the south in the area, so it would be expected that dredged areas would be infilled by sediment from the upcoast drift, and therefore beaches to the south of the mined area would be most affected. The primary cause of erosion along the coastline is reportedly a function of the alongshore gradient of mean wave energy, however sand mining was demonstrated to be exacerbating erosion rates.

In a similar study to Thornton *et al.* (2006), Kojima *et al.* (1986) investigated the correlation between seabed mining in water depths of 15-40m and beach erosion along the coast of the Genkai Sea in Japan. Comparison of annual variations in dredging volumes was compared with erosion trends which demonstrated an erosive effect of the dredging on the shorelines. Hydrographic surveys of the dredge hole profiles showed holes in waters deeper than 30m retained their shape, whereas dredge holes landward of 30m changed substantially, filling with sediment. On-offshore sediment transport does not occur, however the authors suggest that the infilling of dredged holes in the shallower regions from the ambient bed may intercept the littoral transport system to the beaches, leading to erosion and creation of a steeper beach slope.

An understanding of the local hydrodynamics of an area is essential in predicting the impacts of changing bathymetries and sediment characteristics regimes. Furthermore due to natural variability of coastline dynamics certain areas of shorelines will be more tolerant to changes in the wave climate and associated sediment transport (Kelley *et al.* 2004). A quantitative analysis was conducted by Kelley *et al.* (2004) to evaluate the fluctuations in coastal processes resulting from sand mining along the US east coast using three case studies. A model was created, enabling assessment of the sediment transport patterns at the shoreline utilising local hydrodynamics to determine the allowable excavation depth before inducing shoreline changes. Criteria for accepting or rejecting the dredge site dimensions was based on a range relative to ± 0.5 standard deviation about the mean natural sediment transport variability. The model predicted that sand extraction



volume, proximity to the shoreline and water depth had the greatest impact on near-shore sediment transport patterns.

Erosion of the shoreline is an ever-changing and natural phenomenon often accentuated by severe storm events. In a balanced system however, beaches should be replenished by natural processes that return the sediment from the near-shore system during calmer weather, though this is dependent on the system in which it is occurring (Newell and Woodcock 2013). Changes in sediment transport regimes of coastlines and near-shore systems resulting from seabed mining may lead to changes in wave and tidal patterns affecting the sediment regime, inducing altered patterns of erosion and accretion of beaches, dunes and shallow sub-tidal areas. These changes may ultimately lead to reduced coastal stability requiring coastal zone management and hazard mitigation.

Numerical and physical modelling of hydrodynamics and sediment transport will be a critical component of any environmental assessment of seabed mining in NT waters. Model verification and validation will also be important in providing predictive capacity and confidence in understanding potential impacts. This is considered in greater detail in Section 5.3.

4.3 Direct Ecological Effects

Regardless of the mining technique, the benthic community within the immediate vicinity of the mining operation will be destroyed, by either removal, smothering or physical destruction. The extent of destruction and its ecological significance, will depend on the bottom type, presence/absence of threatened or keystone species, recovery rate of the community and the local environment. For example, mining in a high-energy environment, characterised by mobile sands and opportunistic species, may be better adapted to frequent disturbance and thus recover quickly when compared with more stable, lower-energy communities that are characterised by long-lived, slower growing species (Newell *et al.* 1998). The loss of habitat and impacts to bottom-dwelling species resulting in mortality from seabed mining may have wider ecological effects throughout the food chain, as predator-prey relationships are altered, changing community composition and ecosystem functioning.

4.3.1 Removal and/or Alteration of Habitat

The most harmful direct effect of seabed mining is the removal of substrate and associated flora and fauna leading to a reduction in abundance, species diversity and biomass of benthic species (e.g. Poiner and Kennedy 1984; Kenny and Rees 1996; Desprez 2000; Boyd and Rees 2003). It is also likely to be the most persistent and wide-ranging impact of mining operations. Shallow seabed mining primarily occurs over mobile substrate (sands, gravels etc.) albeit; coral and coral sands are also mined in countries including Bali, Hawaii, Taiwan, the Maldives and Bahamas (Charlier 2002). Apart from the physical effect, habitat alteration encompasses an effect of changes in environmental conditions, including increased turbidity, and decreased water quality and light availability.

Removal of or alteration in habitat can lead to habitat fragmentation, creating patches and increasing the quantity of habitat edges which has demonstrated positive and negative effect on abundance and species density, particularly in seagrass meadows (e.g. Bell *et al.* 2001; Macreadie *et al.* 2009). A change in habitat structure may reduce habitat availability, disrupt predator-prey relationships and cause changes to community structure and function (Johnson *et al.* 2008). Such



impacts are discussed in detail within the scientific literature (e.g. Ewers and Didham 2006), however not in the context of seabed mining.

Impacts of substrate removal arising from seabed mining affect the flora and fauna associated with the habitats that are directly exposed to mining. In the case of dredging and excavation, the depth of the pit has been demonstrated to have an effect on the extent of biological impacts. For example, shallow dredging over a large area (such as in the case of trailer dredging) may lead to wider ecological effects, given most infauna live in the top layers of sediment, whereas shallow static dredging will affect less of the seabed surface area (Phua *et al.* 2004). The study of coastal sand mining by Byrnes *et al.* (2004) also suggests a similar approach where mining several small sites on a target ridge or shoal, or mining relatively small portions of several ridges or shoals, may help to ensure availability of nearby populations of potential colonizers.

Deep excavations however, are likely to affect benthic species composition at the base of the pit due to changes in hydrodynamic conditions, sediment composition and oxygen availability, altering the micro-ecosystem (Phua *et al.* 2004).

4.3.1.1 Impact to Species and Community Composition

Overall, experimental studies have shown that mining is accompanied by a change in species' abundance, density and biomass. While this is to the detriment of some species and communities (e.g. Eleftheriou and Robertson 1992; Desprez 2000), in others, it promotes an increase in abundance and biomass of selected species (e.g. deJong *et al.* 2014). Removal or a dramatic change in community ecology, often favours the establishment and growth of fast-growing, opportunistic (i.e. "r-strategist") species, in a niche that may have otherwise been unavailable (see Newell *et al.* 1998 for review; van Dalssen *et al.* 2000). The alteration in community structure and function that is likely to follow, may result in a new equilibrium, or eventually recover to a structure that is similar to pre-disturbance, frequently dominated by taxa that are longer-lived, slower-growing and larger (Kenny and Rees 1994, 1996; Newell *et al.* 1998; Byrnes *et al.* 2004; Phua *et al.* 2004). It can also provide the mechanism for introduction of invasive marine species (see Section 4.4.5).

Changes to species richness, density and biomass as a result of seabed mining, particularly dredging, is frequently reported in the literature. Poiner and Kennedy (1984) compared the benthic faunal assemblages in sand mining areas with reference sites following the dredging of sand for the upgrade to Brisbane's International Airport. A decrease in species richness (from an average of 33.0 to 16.6 species per site), a decrease in mean abundance (from 117.9 to 47.6 individuals per site) and a decrease in mean diversity (4.03 to 3.22 per site) for dredged areas was reported.

Examples from local dredging projects such as Inpex and the East Arm have limited relevance as monitoring of the excavation footprint is not usually undertaken in dredging projects, in preference to monitoring of impact and recovery at the spoil disposal area.

With the recent Inpex dredging project, the overall temporal and spatial patterns of abundance, taxon richness and assemblage of the subtidal infauna indicated that as expected, spoil disposal within the spoil disposal area (SDA) did have a significant impact on the infaunal assemblage. Within the SDA, the changes in median grain size and percent composition of fines also support the hypothesis of smothering and burial via spoil placement being the main cause and effect pathway impacting benthic infauna. However, following completion of spoil disposal, there was

significant recovery in both the abundance and taxon richness of these indicators back to baseline levels (Cardno 2015e).

Maintenance dredging and marine aggregates dredging can be expected to result in a 30–70% reduction of infaunal species diversity, a 40–95% reduction in the number of individuals, and a similar reduction in the biomass of benthic communities in the dredged area (Newell *et al.*, 1998). Gravels and shingles are extensively mined from an average depth of 15–20m off Dieppe, in the English Channel, and an empirical study conducted by Desprez (2000) has demonstrated that fundamental changes occurred to the community structure. Once dominated by coarse sands, the area progressively became more homogenous and was then dominated by fine sands (see Section 4.2.2). Subsequently, there was a 63% decrease in species richness, 86% decrease in abundance and 83% decrease in biomass. The dominant species also changed from Branchiostoma to polychaetes (*Ophelia borealis*, *Nephtys cirrosa*, and *Spiophanes bombyx*), with Echinocardium as complementary characteristic species.

Changes in benthic communities were monitored by Kenny and Rees (1994) as a result of gravel mining off Norfolk (Figure 4-3) (as presented in Newell *et al.* 1998). Changes to the community composition following dredging was markedly different to pre-dredging structure, and sites were highly variable, however over a period of months, the communities between sites became more similar to each other, the reference areas and pre-dredging samples. An extension of this study (Kenny and Rees 1996) further demonstrated that despite significant recolonisation of the area, the community remained distinctively different compared to pre-dredging conditions (Figure 4-4).

The ordinations used in Figure 4-3 and Figure 4-4 provide a means of visualizing the level of similarity of individual cases of a dataset. In these examples, the closer the site samples, the greater the similarity in terms of abundance and diversity.

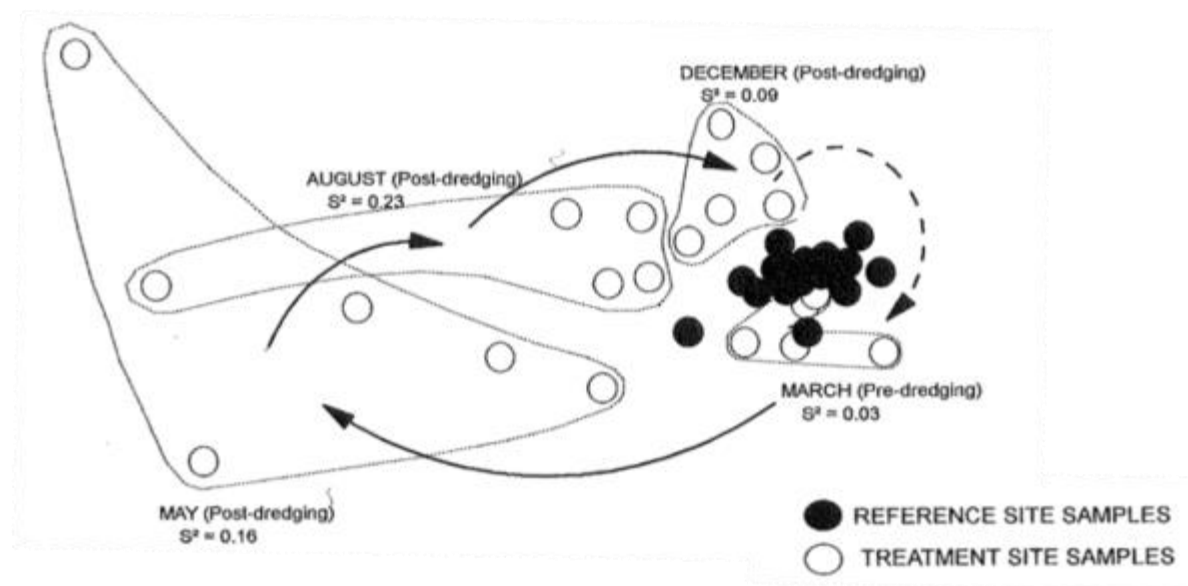


Figure 4-3 : Multidimensional scaling (MDS) ordination for the benthic community changes within a year of dredging in a gravel mining area off Norfolk (from Newell *et al.* 1998, as adapted from Kenny and Rees 1994)

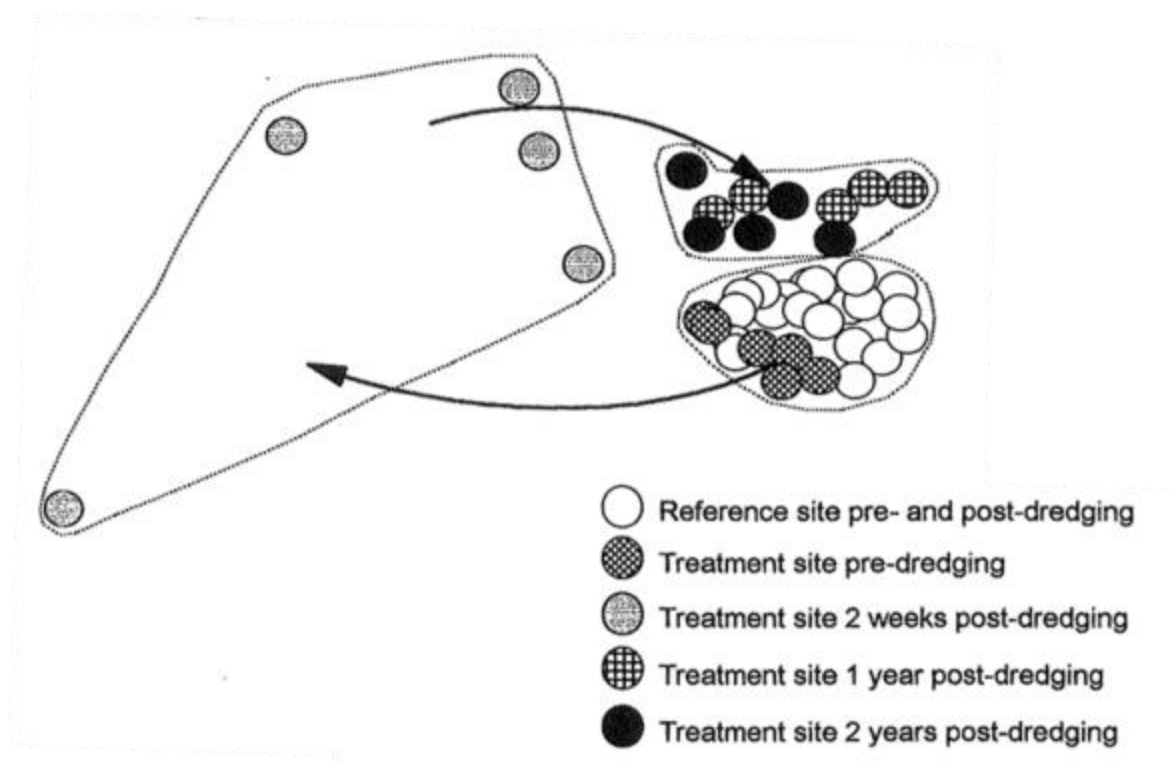


Figure 4-4 : Multidimensional scaling (MDS) ordination for the benthic community changes in a dredged gravel mining area off Norfolk, 2 years post-dredging (from Newell *et al.* 1998, as adapted from Kenny and Rees 1996)

Changes in the macroinvertebrate community may have long-term implications, particularly where the physical environment is altered. For similar faunal assemblages to occur at dredged sites, the original topography, sedimentary characteristics and hydrodynamics must be restored (Gubbay 2003; Boyd and Rees 2003). While the effect of habitat removal is relatively well known for marine fauna the smaller-scale alterations in topography and subsequent changes in sediment characteristics resulting from seabed mining and its effect on pelagic species has only recently received attention.

Fish may only be indirectly affected from mining activities, as generally their mobility enables them to avoid areas of disturbance. However, certain species may be susceptible if extraction occurs in breeding or spawning areas (Hanson *et al.* 2004). Changes in the benthic characteristics may also discourage spawning altogether. For example, gravel mining in the southern North Sea has been linked to impacts on the sand-eel and subsequently cod stocks, in addition to herring species. Given the sand-eel lays its eggs in the sand and once fully covered the embryo will develop, mining in the spawning areas will result in a disruption to the reproductive cycle of the sand-eel. This will undoubtedly affect cod stocks that rely on the sand-eel as a primary food source (de Groot 1979). Herring species have also been shown to avoid mining areas, attempting to lay eggs in different locations of which larvae were then destroyed due to wave action (de Groot 1979). Impacts to fish assemblages, including species of commercial importance have also been reported by Hwang *et al.* (2014). Sand mining in Gyeonggi Bay, Korea, has resulted in a decrease of species richness, diversity and abundance in impacted areas compared with reference locations (Hwang *et al.* 2014). Kim and Grigalunas (2009) have estimated the cost to commercial fisheries as a result of marine sand mining in Ongjin, Korea, considering short- and long- term effects in addition to food web effects. Predicted value of the loss in catch over a one-year period as a result of mining a single site with a



two-year burrow pit recovery was \$38,851. Estimated cumulative damages due to recurring mining, for a five-year period at 20 mining sites, yields estimates in the order of millions of dollars.

Fish may also be affected where key prey items are removed from the benthos (Gubbay 2003; Son and Han 2007; deJong *et al.*, 2014) or where physical changes to the seabed landscape alters the seabed macrofaunal composition (deJong *et al.* 2014) as extraction may increase food availability due to the increase of damaged invertebrates such as bivalves or crustaceans, thereby increasing the abundance of fish present (Phua *et al.* 2004). deJong *et al.* (2014) found that fish assemblages at the crests of landscaped sandbars varied compared to those within troughs at the Port of Rotterdam in the Netherlands. Furthermore, lower species richness yet higher biomass occurred in extraction sites when compared with reference sites and in the years immediately following sand extraction, but fish biomass reduced back to reference levels after two years. deJong *et al.* (2014) report that these changes in community composition occurred due to the reduction of larger particles in the dredging area. This led to an increase in the white furrow shell (Tennelid shellfish), due to its preference for fine sand and a subsequent increase in plaice (*Pleuronectes platessa*), which preferentially feed on white furrow shell. The decrease in species richness, but increase in biomass of demersal fish was directly related to the increased abundance of plaice. This demonstrates a link between alteration of the physical environment and biological assemblages.

Coral mining has shown to have an effect on wave processes (see Section 4.2.1), the reef structure and associated reef fishes (Brown and Dunne 1988; Ford *et al.* 2013). Direct removal of coral in the Maldives has led to a transformation in the reef; which has transitioned to filamentous algae covered - dead branches and coral rubble (Brown and Dunne 1988). Additionally, the maximum diameter of colonies was smaller compared to control locations. It is therefore not surprising that diversity of reef fish and fish abundance (adults and juveniles) was lower at mined sites compared with non-mined sites (Brown and Dunne 1988). Abrasion and smothering of new recruits has led to minimal recovery of the reef, as settlement on the mobile rubble is near impossible (Brown and Dunne 1988). Reef mining in the Maldives has also had a measurable impact, reducing overall biomass and abundance across three trophic levels; planktivores, benthic herbivores and omnivores (Shepherd *et al.* 1992). This was shown to be an effect of habitat type and structure preferences of the sampled species, which had been altered due to mining operations. Contrastingly, reef slopes adjacent to mined areas, supported a higher biomass and abundance of reef fishes than reef slopes adjacent to non-mined areas, primarily due to higher availability of plankton and reduced competition (Shepherd *et al.* 1992).

4.3.2 Noise, Blasting and Vibration

Noise impacts relating to mining activities may comprise noise from mining vessels, seismic exploration, hydraulic noise, the crushing or breaking of substrate prior to its removal, dredge dragheads, in addition to any blasting or from the loading or dumping of rocky material. No studies specifically relating to noise impacts from seabed mining were identified, however there is an abundance of data and studies that discuss the impacts of dredging noise and seismic noise on fish and marine mammals, with an overwhelming majority regarding cetaceans. An in-depth review of the impacts of seismic surveys on marine life has been completed by Weilgart (2013) and DNV (2007). A brief review of these impacts and other noise, blasting and vibration impacts relating to seabed mining follows.

Any elevated continuous background noise has the potential to mask megafauna communication systems and listening capabilities for predator avoidance, as well as disturb normal behaviour due to displacement from critical habitats. Thus the noise from dredges is a potential risk, especially to

turtles because their hearing range falls within the main energy band. Research from elsewhere suggests that, compared to other sources of underwater noise, dredging is within the lower range of emitted sound levels. Shorebirds seem unlikely to be affected by underwater noise, and the migratory species in particular are not heavily reliant on acoustic communication while they are in Australia during the nonbreeding season (refer to McCook *et al.* 2015).

Anthropogenic underwater noise may impact on marine fauna by:

- Mortality, or pathological organ damage (e.g. André *et al.* 2011);
- Temporary or permanent shifts in hearing thresholds (e.g. McCauley *et al.* 2003);
- Behavioural changes or displacement from areas (e.g. Perry 1998); or
- Masking or interference with other biologically important cues (such as communication or echolocation) (e.g. Hawkins and Chapman 1975).

Impacts to fauna will depend on the frequency range and intensity of the noise being produced, distance from the noise source, species' sensitivity, water depth and the length of time the sound is being emitted. Factors such as temperature, water depth, frequency, amplitude and seabed composition will affect how the sound propagates through the water column and therefore the area in which impacts occur may be larger than the mining footprint (Swan *et al.* 1994; Phua *et al.* 2004). Furthermore, the most important aspect of the effects of sound exposure is the presence or absence of a swim bladder, as those species lacking swim bladders are less susceptible to fatal impacts (Casper *et al.* 2012; Popper *et al.* 2014). The severity of the response of fauna not only relates to the distance to the noise source, but also factors such as (but not limited to) prior exposure, age, gender, health and the current behavioural state (Erbe 2012). A schematic of the extent to which zones of impact may occur around a noise source has been illustrated by Erbe (2012) (Figure 4-5).

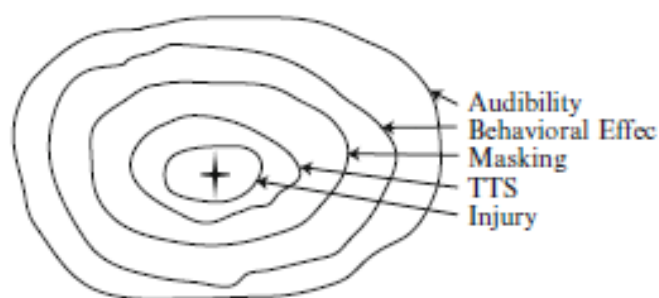


Figure 4-5 : Relative extent of different zones of impact around a noise source (TTS: temporary threshold shift) (from Erbe 2012).

Noise above a species' threshold of sensitivity may lead to a temporary auditory threshold shift (TTS, i.e. temporary hearing impairment) or permanent shift (PTS) (resulting in permanent deafness or loss of some frequencies) (Ketten 2002; Popper 2003; DNV 2007; Weilgart 2013). Where the duration of intense sound is short and the noise is narrow, the loss will be limited and recoverable. Generally there is evidence to suggest a signal intensity of 80dB over a species' threshold at each frequency will result in a significant threshold shift (Ketten 2002). McCauley *et al.* (2003) found that fish sustained severe damage to their sensory epithelia from air gun exposure at distances of 500 m to several kilometres with no evidence of repair 58 days following damage. The TTS threshold of the harbour porpoise (*Phocoena phocoena*) was determined by Lucke *et al.* (2009) by measuring the response to airgun arrays. The male harbour porpoise consistently showed aversive behavioural reactions at received sound exposure levels (SEL) above 145 dB re 1 $\mu\text{Pa}^2\text{s}$.



Erbe (2012) has also compiled the sound spectrum measurement of various sources of anthropogenic sound, including seismic airguns and dredging and material dumping which has been back-propagated to 1m from the source (Figure 4-6). Sound levels from some large TSHDs operating in rocky areas have been recorded in excess of 150 dB re1 μ Pa at 1km, while large CSDs can emit noise audible 20-30 km away (Richardson *et al.* 1995; Dames and Moore 1996). Hanson *et al.* (2004) report that sand mining sound pressure levels are expected to vary between 130-140dB at frequencies ranging 300-400 Hz between 30-40ft depth and are projected to reduce to 120 dB 1km from the source and to 112 dB at 3km. Robinson *et al.* (2011) assessed noise levels by marine aggregate dredges (n=7) in the UK (Figure 4-7) and also found that there is considerable variation between dredges at frequencies less than 500 Hz and that gravel extraction generated higher noise levels compared to sand extraction. Robinson *et al.* (2011) also summarised the noise levels reported in the literature (Figure 4-8) in addition to reporting the sound recorded at 100m from the TSD 'Sand Falcon' compared to background levels during full dredging, pumping of water only, and when the pump was off (Figure 4-9). The primary cause of noise in higher frequencies is shown to be a result of material being extracted and being sucked through the pipe.

Monitoring of underwater noise during the Ichthys Nearshore Environmental Monitoring Program (Salgado-Kent *et al.* 2015) recorded a broad range of noise, from approximately 20 Hz to nearly 5 kHz with intensity levels reaching close to 160 dB re 1Pa during some periods. Received noise levels produced during dredging at Walker Shoal were a focus due to the hard substratum being dredged. The analyses revealed noise levels close to approximately 145 dB re 1 μ Pa at distances between 630 m and 680 m from the source.

Noise during conventional sand mining from San Francisco Bay was expected to be around 130-140 dB at frequencies ranging around 300-400 Hz at a depth of approximately 10-15m. At approximately 1 km from a sand mining operation sound pressure levels are expected to have decayed to 120 dB and a further drop in sound pressure level to 112 dB by approximately 3 km. These levels are expected to decay to ambient background levels by 15 km from the point of operation. Water depth and substrate, among other factors, will affect the time and distance taken for sound levels to reach ambient background levels (Hanson *et al.* 2004).

To begin to understand the levels at which impacts may occur (such as masking), an understanding of species' individual bandwidths of sound emitted, and the bandwidths of the cues they respond to is required. Few studies have determined this, and information is sparsely available, limited to few species. For example, Ketten (2002) have synthesised audiograms from the literature for the beluga whale, killer whale, harbour porpoise, bottlenose dolphin, false killer whale, Risso's dolphin, pinnipeds and several seals and sea lions. Similarly, Popper *et al.* (2014) has compiled information for Atlantic salmon, plaice, dab and Atlantic cod.

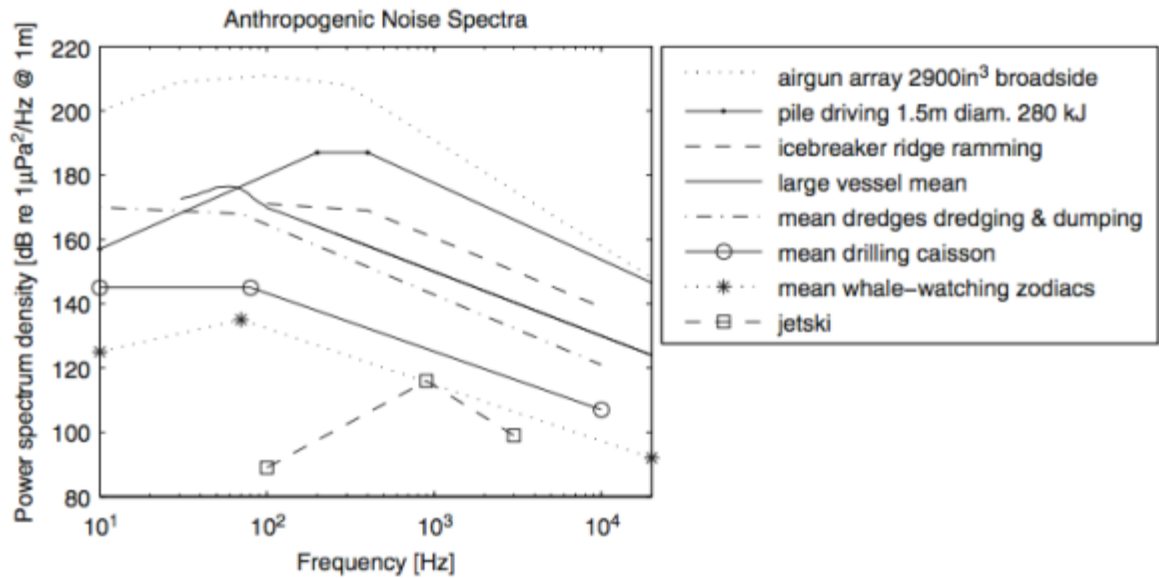


Figure 4-6 : Sound pressure levels of various anthropogenic sources (from Erbe 2012)

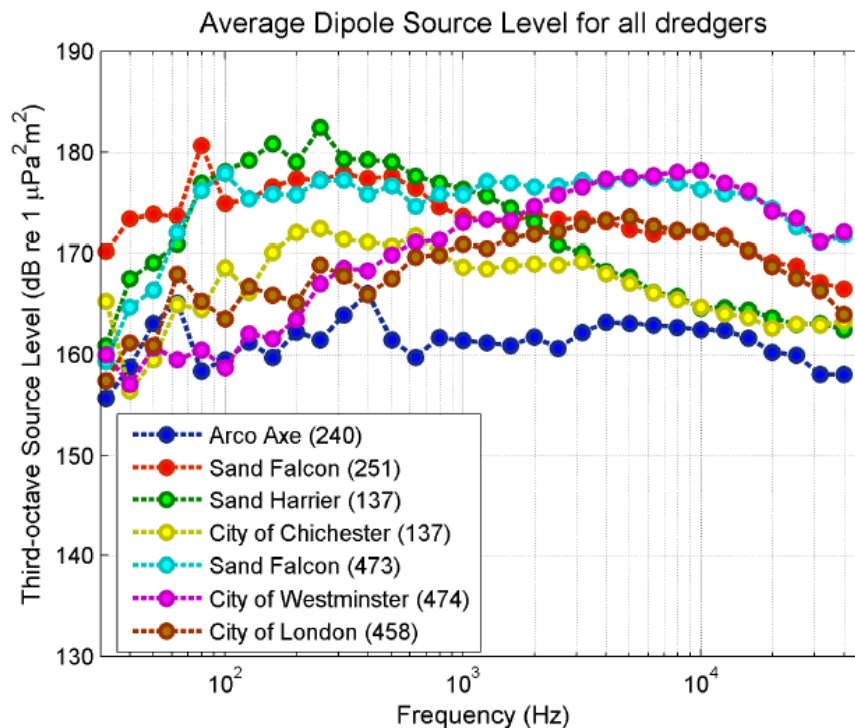


Figure 4-7 : Source levels calculated for marine aggregate dredges in the UK (from Robinson *et al.* 2011)



Dredger name	Hopper capacity (m ³)	Total installed power (kW)	Survey location	Water depth (m)	Sediment type	Source Level (dB re 1 μ Pa ² m)		Reference
						Peak TOB	Broad band	
Beaver Mackenzie (cutter suction)	-	-	Beaufort Sea	13	-	167	172	Miles et al, 1987/Richardson et al, 1995
Aquarius (cutter suction)	2,500	15,620	Beaufort Sea	46	-	178	185	Miles et al, 1987/Richardson et al, 1995
Cornelis Zanen (TSHD)	8,530	12,064	Beaufort Sea	20	-	-	-	Miles et al, 1987/Richardson et al, 1995
Gerardus Mercator (Large TSHD)	18,000	-	Sakhalin	-	-	183	188	Sakhalin energy report/ Ainslie et al, 2009
Taccola (TSHD)	4,400	6,300	-	-	-	-	188	Nedwell et al, 2008 (from Langworthy et al, 2004)
Arco Adur (TSHD)	2,700	2,940	Great Yarmouth Cross Sands (Area 328)	-	-	-	-	Defra/Cefas report
Arco Adur (TSHD)	2,700	2,940	Hastings Shingle Bank	~18	Gravelly sand	-	-	Defra/Qinetiq report
City of Westminster (TSHD)	2,999	4,080	Hastings Shingle Bank (Area 460)	~18	Gravelly sand	170*	186	Parvin et al, 2008

*Obtained from Parvin et al, 2008 TOB Received Level data at 514 m and Propagation Loss equation used by Parvin et al, 2008, with no frequency dependent absorption.

Figure 4-8 : Summary of the dredging noise surveys reported in the literature (from Robinson *et al.* 2011)

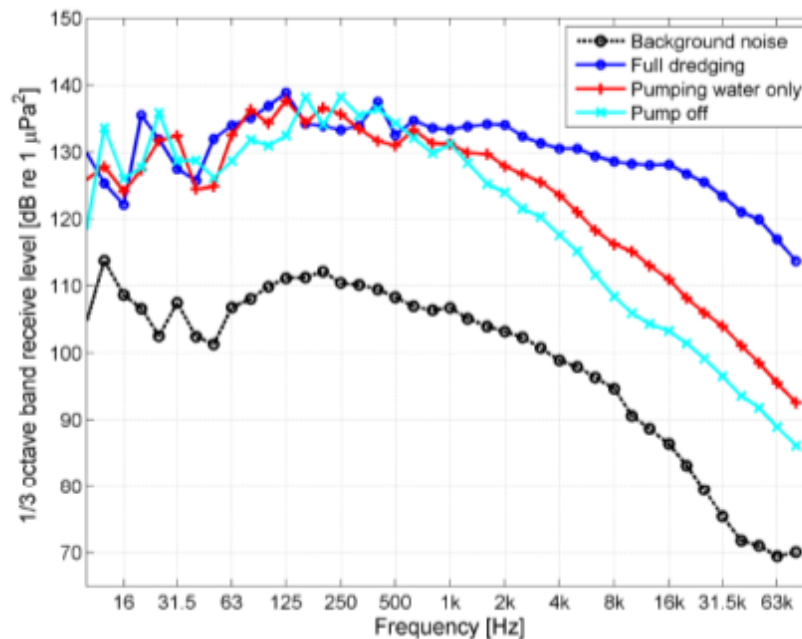


Figure 4-9 : Noise levels recorded from the 'Sand Falcon' dredger at 100m distance (from Robinson *et al.* 2011)

While no direct causal evidence exists, strandings of both Humpback whales in Brazil and Beaked whales in the Galapagos and Gulf of California have been linked to seismic surveys (Gordon *et al.*



2003; Engel *et al.* 2004). Furthermore, relationships have also been made between seismic air guns and mass giant squid strandings in Spain's Bay of Biscay, as evidenced by internal injuries and severe ear damage (MacKenzie 2004). Of equivalent importance are non-lethal impacts which may modify species' growth and reproduction which can threaten whole populations and their ongoing survival of a species (Tyack 2008). For example, exposure to seismic energy in an experiment in Canada, has led to bruised organs, abnormal ovaries, in addition to bleeding, stress, delayed embryo development and smaller larvae in snow crabs (DFO 2004).

Behavioural changes have been noted for several marine species due to anthropogenic noise which may also be relevant to seabed mining activities. Fish have shown to drop to deeper depths, become motionless or more active, or form a compact school (Dalen and Knutsen 1987; McCauley *et al.* 2000; Pearson *et al.* 1992; Santulli *et al.* 1999; Skalski *et al.* 1992; Slotte *et al.* 2004). Furthermore, Wardle *et al.* (2001) found seismic air guns cause fish to be startled by an air gun being fired inducing a "C-start" response, however they quickly resumed the same prior behaviour and no movement of fish or invertebrates away from the gun was noted. Nomadic or migratory fish may behave differently in the presence of the seismic surveys (Wardle *et al.* 2001). So called, "C-start" responses have also been seen in few other fish species (DNV 2007). Harris *et al.* (2001) also found that seismic airguns had a minimal effect on the behaviour of ringed, bearded and spotted seals off Northern Alaska, with only partial avoidance of the area (<150m from the vessel) during full-array seismic events.

McCauley (1994) reports the distances and sound levels at which behaviour responses of fish may be induced from seismic activity:

- Subtle behavioural responses expected for several kilometres from the source at ~160 dB re 1 μ Pa;
- Alarm responses up to 2 km (at ~180 dB re 1 μ Pa); and
- Startled responses within approximately 300m, followed by a flight response (at ~200 to 205 dB re 1 μ Pa).

Most marine fauna are dependent on sound for many factors essential for their fitness and survival, including feeding, predator avoidance, reproduction and navigation. Given underwater anthropogenic noise is predominantly at low frequencies (below 1 kilohertz and can reach sound pressure levels of over 200dB re1 μ Pa at the source) (Perry 1998), it is particularly concerning for marine mammals and cetaceans, such as the baleen whale which commonly use low-frequency signals for communication (Perry 1998; Richardson & Wursig 2007; Tyack 2008). Weilgart's review (2013) and citations within discuss a number of varied behavioural impacts in marine mammals as a result of seismic activity including;

- Fin whales who ceased singing, which resumed following cessation of seismic activity;
- Modification of vocalisation in Blue whales;
- Avoidance of the area;
- Reduced swimming effort of sperm whales;
- Shorter dive times and lower respiration rates in bowheads;
- Faster swim rates and increased respiration rates in western grey whales;
- Switching from foraging to transiting behaviours in grey seals; and
- Increased stress and weakened immune systems.



Displacement has also been documented in a number of marine cetaceans in relation to noise exposure. Grey whales, beluga whales, fin whales, sperm whales, bottlenose dolphins and harbour porpoises, have been linked to displacement as a result of seismic exploration and vessel traffic (Bryant *et al.* 1984; Finley *et al.* 1990; Evans *et al.* 1993; Evans *et al.* 1994; Mate *et al.* 1994; Castellote *et al.* 2012; Pirotta *et al.* 2013) along with humpback whales, blue whales, and bowhead whales that have abandoned areas in response to boating, aircraft and other industrial activities (Perry 1998). Perry (1998) state that cetacean behaviour varies dependent on an individual's age, sex, state of activity, location, season and time of day and the significance of noise and how the individual responds therefore also will vary. This means that baseline surveys to measure disturbance effects can be complicated, particularly as they are often less responsive when feeding or mating, compared with resting (Richardson and Würsig 1997). Furthermore, reactions and impacts tend to be higher for continuous noises, and erratic moving signals, compared with static pulses.

Species that lack a swim bladder, or other air-filled cavity, e.g. elasmobranchs (sharks, rays and skates) are incapable of detecting sound pressure, so may rely on particle motion, which detect sound when the hair cells under the otoconia are bent when sound waves travel through their acoustically transparent body (Casper *et al.* 2012). This means that they may not be as sensitive to noise as other species. No behavioural experimental studies have been conducted exploring skate or ray responses to sound, nor studies on any elasmobranch for effects of anthropogenic noise (Casper *et al.* 2012). However there is evidence to suggest sharks are often attracted to certain noises, although exhibit flight responses when presented with a sudden onset of intense sound nearby (such as that exhibited by the scream of a killer whale) where the intensity was considerably higher than the preceding sound (Banner 1972; Myrberg *et al.* 1978; Myrberg 2001).

Anthropogenic noise relating to marine mining may mask sounds that species rely on (DNV 2007; Erbe 2012; Weilgart 2013). Masking will depend on the spectral and temporal characteristics of the signals and anthropogenic noise; at low signal-to-noise ratios a signal may indeed be audible, whereas high signal-to-noise ratios will potentially result in environmental cues being masked (Lucke *et al.* 2009; Erbe 2012). Casper *et al.* (2012) states that it is unlikely that damage result in elasmobranchs as a result of typical shipping vessels, however the sound would be sufficient in masking detection of biologically relevant sounds.

Very little is known regarding the hearing of sea turtles (Popper *et al.* 2014), however they have been shown to exhibit an avoidance response to seismic surveys (McCauley *et al.* 2000; Lenhardt 2002; DeRuiter and Doukara 2010). Trials with captive marine turtles demonstrated that generally an alarm response is exhibited at an estimated 2 km range from operating seismic vessels and avoidance behaviour at 1 km (McCauley *et al.* 2000). A study on loggerhead turtles (*Caretta caretta*) by Lenhardt (2002) demonstrated turtles increased their swimming speed and avoidance occurred as a result of seismic air gun trials. Loggerheads tracked during seismic surveying in the Mediterranean found that 57% of individuals dove at or before their closest point of approach to airguns with some individuals also exhibiting startled responses (DeRuiter and Doukara 2012). McCauley *et al.* (2000) indicates turtles may begin to show behavioural responses to an approaching seismic array at received sound levels of approximately 166 dB re 1 μ Pa, and avoidance at around 175 dB re 1 μ Pa.

Blasting is described as a sudden onset of a loud noise, and is generally from a single exposure to an explosive shock wave with a massive, but short-lived increase in pressure, followed by a refractive wave where pressure drops back below the baseline (Ketten 2002). Humpbacks have



been known to die following underwater explosions (Ketten *et al.* 1993) and it is speculated that blasts are capable of inducing significant hearing losses (Ketten 2002).

4.3.3 Entrainment and Collisions

Recent studies have compiled data on the collision rates between ships and cetaceans, albeit predominantly whales (Laist *et al.* 2001; Jensen and Silber 2004; Van Waerebeek *et al.* 2007; Neilson *et al.* 2012). While these reviews are not specific to collisions from vessels associated with mining activities, it demonstrates the relationship between increased vessel traffic and the risk of collisions with marine fauna. Fin whales (*Balaenoptera physalus*) are struck most frequently, and southern right whales, humpback whales (*Megaptera novaeangliae*), sperm whales (*Physeter Catodon*) and grey whales (*Eschrichtiis Robustus*) were also subject to strikes. Strikes were most common by ships more than 80 meters in length, and those travelling more than 14 knots (Laist *et al.* 2001; Neilson *et al.* 2012). Juveniles and calves were also found to be most vulnerable (Laist *et al.* 2001; Neilson *et al.* 2012). Only one known incident of a dredge vessel is known, a southern right whale (*Eubalaena australis*), that occurred in East London Harbour (Best *et al.* 2001). Collisions may occur with not just whales, but other marine mammals, including dolphins, dugongs, and seals, however sea lions and seals are not considered to be at risk of large vessel collisions (Pidcock *et al.* 2003).

Vessel strikes may result in propeller cuts, damaged flukes, skeletal damage, strandings and acute trauma (Laist *et al.* 2001; Van Waerebeek *et al.* 2007). In the case of the incident of the southern right whale in South Africa, the calf surfaced in front of the dredge, was struck by the propeller, and subsequently stranded and died (Best *et al.* 2001; Jensen *et al.* 2004). The calf was with its mother at the time, which was seen within the vicinity of the stranded calf, and efforts had to be made to ensure prevention of the mother beaching herself (Best *et al.* 2001). If marine activities are timed to avoid key resting areas and known migration paths during the relevant seasons, the likelihood of collision substantially decreases (Todd *et al.* 2014).

Within the literature, entrainment is considered only in reference to dredging, particularly from the suction of hydraulic dredges. Benthic fauna and demersal fish associated with the benthos are considered to be more vulnerable to entrainment compared with more mobile pelagic species (Todd *et al.* 2014). Undoubtedly, entrainment in a dredge will result in significant trauma, if not death of the individual (Reine *et al.* 1998). Drabble (2012b) has conducted estimates of vulnerability to entrainment in hydraulic dredges for a selection of fish species specific to the East Channel Region of the UK based on traits such as sensitivity to noise, burst speed, burial and fecundity. Furthermore, those species identified as vulnerable, have recorded marked population changes over the 2005-2008 monitoring period (Drabble 2012b). Furthermore, a zone of potential impact, the Entrainment Impact Zone (EIZ), was proposed by Drabble (2012b) in which plankton and nekton are subject to possible entrainment (Figure 4-10). It is a function of the Active Dredge Zone (ADZ) and calculated based on the one seventh power law calculated at 34m of an average 35m of water column.

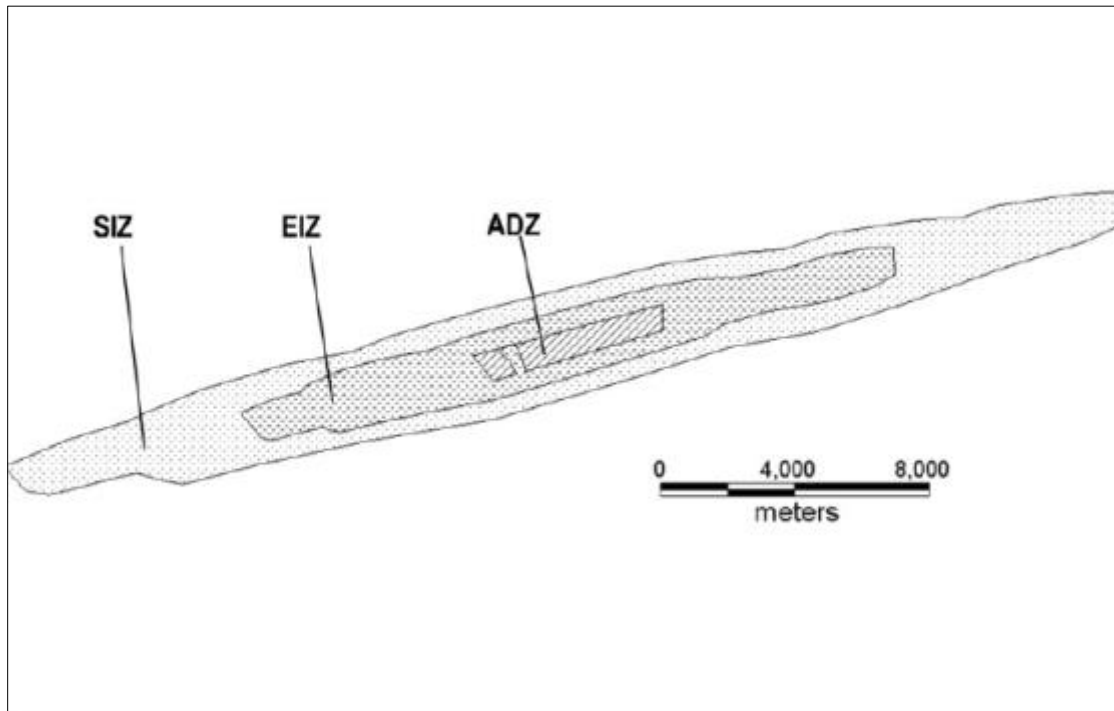


Figure 4-10 : Conceptual model demonstrating the Active Dredge Zone (ADZ), Entrainment Impact Zone (EIZ) and Secondary Impact Zone (SIZ) (from Drabble 2012b)

Entrainment by hydraulic dredges, particularly in harbours and estuaries has been reviewed by Reine *et al.* (1998) which demonstrated entrainment effects, and rates are highly species and location dependent. Dredging in spawning areas is likely to increase the likelihood of entrainment of eggs, larvae and juveniles, particularly those relying on the benthos.

Turtles are also susceptible to entrainment by dredging vessels. During active dredging operations, hopper dredge drag heads are slow moving and nearly silent while suctioning bottom sediments, thereby potentially causing injuries or death to sea turtles that are entrained into the draghead (Dickerson *et al.* 1991; Banks & Alexander 1994; Dickerson *et al.* 2004; Fitzpatrick *et al.* 2006). To reduce the environmental effect associated with turtle entrainment, two different deflector types were developed for use on a draghead: rigid deflectors and flexible chains. These mitigation measures and other management strategies are discussed in greater detail in Section 6.

4.4 Indirect Ecological Effects

Indirect (secondary) impacts generally arise from increased sediment load, or nutrients into the water column and from the deposition or settlement of sediments on the sea floor (Newell and Woodcock 2013).

4.4.1 Turbidity

Turbidity refers to the amount of sediment or organic matter present in the water column, which may be induced naturally through storms or tidal flows, or anthropogenically, for example through seabed mining or tailings disposal (Section 4.5). Turbidity is most often measured in terms of nephelometric turbidity units (NTU), Jackson turbidity units (JTU), or as the concentration of total suspended solids (TSS, mg/L) (Erftemeijer *et al.* 2012; IADC 2015). In the case of dredging, disturbance of the sediment occurs from both the dredge- or drag- head, in addition to the

material being discharged as a result of screening or from overspill (or overflow) (Figure 4-11). Seabed mining methods, as described in Section 2.4.2 will disturb at least the top layers of sediment on the seabed and result in sediments entering the water column (Continental Shelf Associates Inc. 1993).

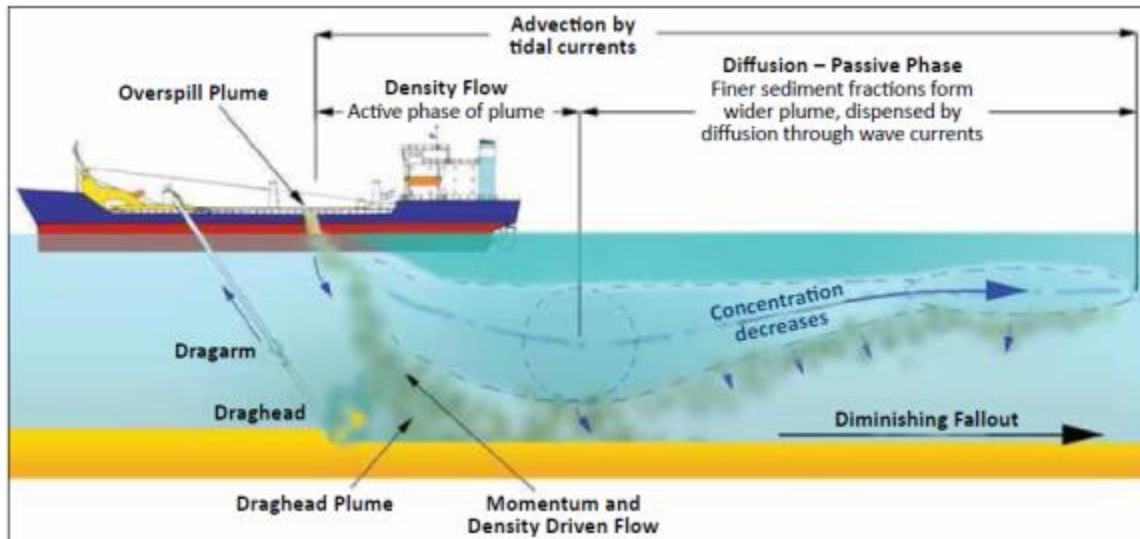


Figure 4-11 : Sediment plumes and turbidity as a result of overspill/screening and the draghead of a dredge during aggregate extraction (from Newell and Woodcock 2013, image © ENTEC)

Suspended sediments will form a sediment plume with larger particles settling more rapidly, while fine sands may be suspended in the water column for some distance before settling out, increasing the impact footprint beyond the limits of the extraction area (Hitchcock and Bell 2004). Depending on the hydrodynamics of an area, very fine sand suspended due to dredging may be carried up to 11 km from the dredging site, fine sand up to 5 km, medium grained sand up to 1 km and coarse sand just 50 m (Hitchcock and Drucker 1996). Capital dredge monitoring of turbidity from three projects in tropical Western Australia (Barrow Island, Burrup Peninsula and Cape Lambert), demonstrated that the zone of potential impact exceeding the 80th percentile of baseline turbidity, was predominantly within three kilometres from the dredging area, although in one instance, up to 20 kilometres at Barrow Island (Fisher *et al.* 2015). Acoustic Doppler Current Profilers (ADCP) can be used to track plumes as shown in Figure 4-12 where sections of a plume from a marine diamond mining vessel in Namibia, demonstrate that most of the sediment disturbance is in the immediate vicinity of the vessel, and in a narrow band at the seabed (Penney *et al.* 2008). It is important to note that use of acoustic backscatter (ABS) techniques do not preclude the need to collect baseline water quality data from a site to determine background water clarity.

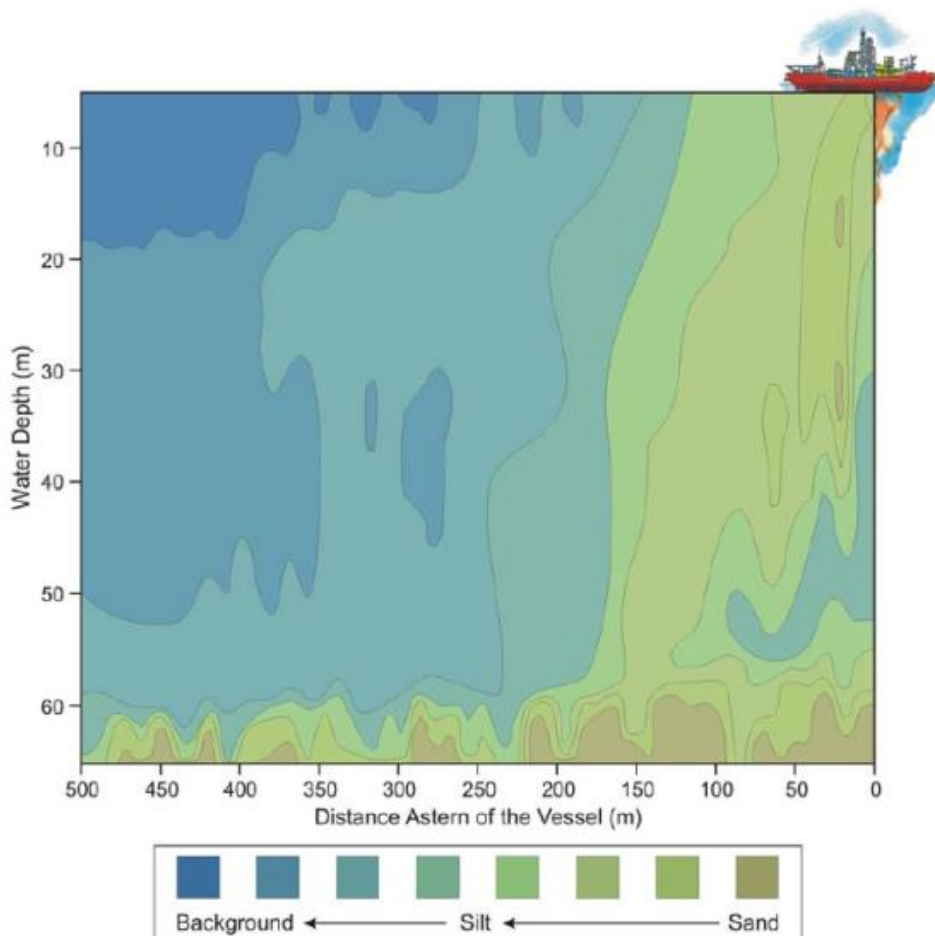


Figure 4-12 : Acoustic Doppler Current Profiler (ADCP) longitudinal section along a plume from a marine diamond mining vessel in Namibia, demonstrating the size and depth of sediment particles behind the vessel (from Penney *et al.* 2008)

The size and depth of a sediment plume resulting from seabed mining will be determined by the concentration and grain size of particles being suspended in the water column, in addition to current velocity and direction (Hanson *et al.* 2004). The lasting effects of the sediment plume are generally short-lived, lasting usually between 4-5 tidal cycles (between high and low), though this will depend on the hydrodynamic regime of the area (Hitchcock and Bell 2004). Similarly, Hanson *et al.* (2004) state that an overflow plume as a result of sand mining typically dispersed after 3-4 hours. These types of plumes are generally short lived as sand mining deliberately targets material that is coarse and composed of sand and gravels. Clays and silts which are generally responsible for causing more persistent plumes represent a much smaller proportion of the material dredged. For example, the target resource in the Hitchcock and Bell (2004) study constituted less than 1-3% silt.

Ecosystems adapt to long-term natural conditions and therefore communities in high energy environments, such as an open, wave-exposed coast, may be more tolerant of increased levels of turbidity and subsequent reduced light penetration (Miller *et al.* 2002). Conversely, species that are present in these types of conditions may also be at the limit of their adaptability and may be vulnerable to an increase in turbidity & sedimentation above normal variation.

Sensitivity and tolerance to turbidity however is species-specific and varies dependent on typical ambient conditions, sediment properties, and the duration and intensity of the turbidity increase

(Erftemeijer *et al.* 2012). Ecological impacts may result directly from increased turbidity but also indirectly due to reduced light attenuation (e.g. Erftemeijer and Robin Lewis 2006) and burial by the sediment as it falls to the seabed, although effects only occur when turbidity is elevated above background concentrations for an extended duration (Gubbay 2003; Phua *et al.* 2004).

The impact to corals from elevated levels of turbidity have been comprehensively reviewed by Erftemeijer *et al.* (2012) and Jones *et al.* (2015b) and range from low-level stress related physiological responses to large-scale changes in community structure. Impacts on the reproduction of corals are also reviewed (Jones *et al.* 2015a) and increase of coral disease under turbid conditions (Pollock *et al.* 2014). Listed impacts include:

- Reduced photosynthetic efficiency
- Changes to heterotrophic feeding
- Increased ciliary/polyp activity
- Increased energy expenditure for cleaning;
- Mucus production
- Sediment necrosis
- Change in coral colour and bleaching due to changes in zooxanthellae and photosynthetic pigments
- Darkening in colour due to photoacclimation
- Injury to coral tissue, loss of polyps and partial mortality of the colony
- Increased bacterial growth
- Decrease in polyp fecundity
- Spawning asynchrony
- Inability for coral larvae settlement and decreased recruitment rates
- Decrease in live coral cover, density and diversity
- Mortality of individuals and colonies; and
- Changes in community structure, with shifts towards sponge and algae dominance.

A conceptual model of the effects of turbidity on corals was created by Jones *et al.* (2015b). The model demonstrates all known linkages between the causes and effects of anthropogenically-generated turbidity on corals, including elevated suspended sediment, changes in light penetration and smothering by sediment as it settles on the benthos (Figure 4-13).

As explained by Jones *et al.* (2015b), the relative influence of the stressors can vary on an hourly, daily, and seasonal basis, depending on the dredging (or mining) activities, diel and tidal cycles, and sea-state. Four different exposure scenarios for symbiotic corals exposed to dredging plumes are symbolized in Figure 4-14: (B) A scenario whereby a buoyant plume drifts over the reef with little contact with the corals and where light availability only is likely to be the key proximal stressor (see photograph Fig. 3D), (C) A turbidity event during relatively turbulent water conditions where (wave+current) shear stresses are sufficient to inhibit the deposition of most sediment and so suspended-sediments and reduced light availability are the predominant influences. (D) A scenario where elevated SSCs has occurred during very calm conditions and where sediment has subsequently fallen out of suspension and smothered corals, (E) represents a night time scenario of high SSCs in turbulent conditions where suspended-sediment is the sole proximal stressor.

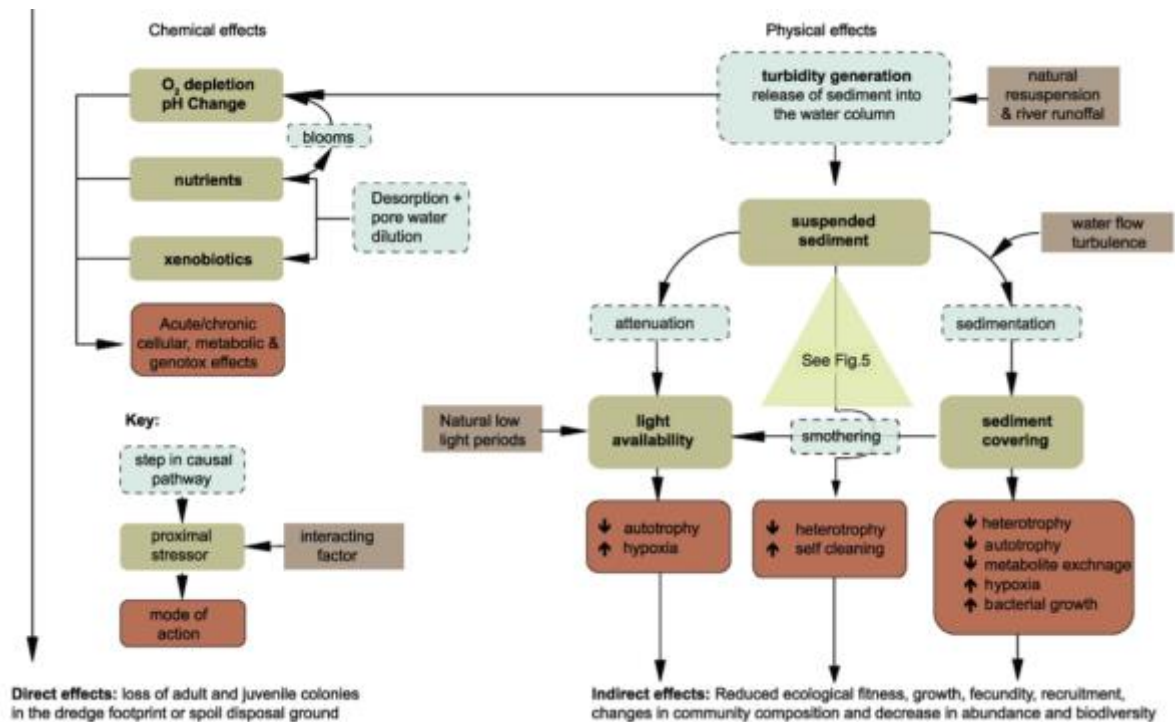


Figure 4-13 : A conceptual model identifying the stressors and biological impacts resulting from chemical and physical effects of turbidity on corals (from Jones et al. 2015b) where Fig.5 refers to the ternary diagram in Figure 4-14.

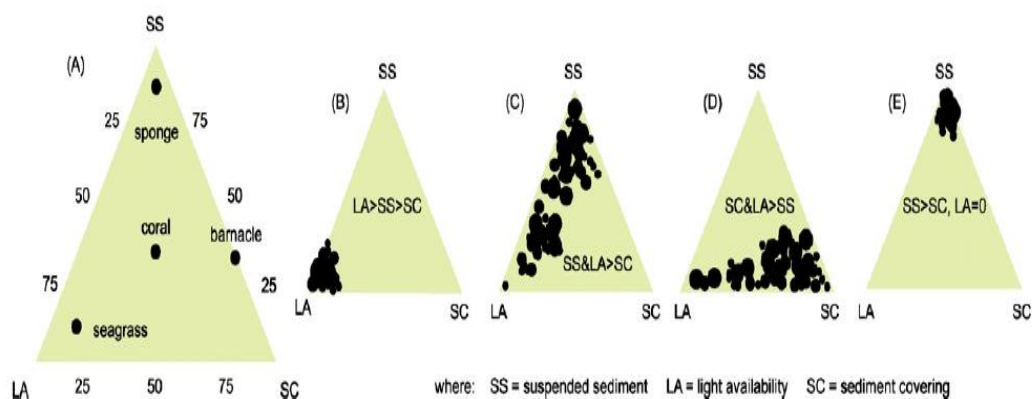


Figure 4-14 Ternary Diagram diagrams representing the relative influence of 3 key proximal stressors associated with turbidity generation activities i.e. suspended sediment (SS), light availability (LA) and sediment covering (SC) on shallow tropical benthic organisms such as corals, sponges, seagrass and filter feeders (from Jones et al. 2015b).

Seagrasses are light-limited and occupy areas based on the penetration depth of photosynthetically active radiation (PAR) (i.e. sunlight) (Erftemeijer and Robin Lewis 2006). Other additional factors such as salinity, temperature, currents, exposure, sediment characteristics and nutrients in the water column and in the sediment can also limit the distribution of seagrasses. Increases in turbidity will affect light availability and therefore the efficiency and depth-limits of seagrasses. Minimum light requirements, in addition to the duration of time that various seagrass



species can survive in light intensities below their minimum requirements, is summarised by Erftemeijer and Robin Lewis (2006). Relevant details for local species are discussed in Section 5.4. In general, impacts to seagrasses as a result of increased turbidity and decreased light attenuation include a decrease in shoot density, above and/or below ground biomass, decreased canopy height, reduced growth and reproduction and subsequent meadow distribution, reduced photosynthesis and mortality (Longstaff & Dennison 1999; Longstaff *et al.* 1999; Ralph *et al.* 2007). Similarly to seagrasses, macroalgae are light dependent, and therefore changes in turbidity leading to altered light attenuation, will impact their growth, distribution and photosynthetic efficiency (Aumack *et al.* 2007). Reductions in PAR penetration will also affect phytoplankton productivity (Dankers 2002). Changes in photosynthesis have implications to the broader ecosystem as it is responsible for the production of oxygen and carbohydrates in the water column and forms the basis of the food chain (Phua *et al.* 2004; Ralph *et al.* 2007).

Turbidity may result in the clogging of fish gills, which may lead to infections and ultimately mortality (Phua *et al.* 2004; Wong *et al.* 2013). Mobile marine species, such as fish, have the capability to move away from an area with environmental stressors higher than their tolerance limits. For example, Westerberg *et al.* (1996) found that both Atlantic cod (*Gadus morhua*) and herring (*Clupea harengus*) had an avoidance threshold to suspended solids of approximately 3mg/L, of which was confirmed by a non-visual component as similar results were achieved at night. Newcombe and Jensen (1996) suggest that typically, sensitive fish species exhibit a threshold limit between 50 and 100mg/L for behavioural avoidance. Furthermore, impacts of suspended sediments on fish have also found to include reductions in the suitability of spawning habitat through impacts of egg and larval development, reduction in food availability, reduced growth rates, changes in migration patterns and the reduced efficiency of prey detection and foraging success (Bruton 1985; de Robertis *et al.* 2003; Hanson *et al.* 2004; Phua *et al.* 2004). Increased suspended sediment has also found to increase fish species richness and abundance in the surf zone, due to increased shelter from predators (Clark *et al.* 1998). A review of the literature conducted by Hanson *et al.* (2004) tabulates the biological responses of a variety of fish and invertebrate species, including egg and larval effects, to suspended sediment exposure.

Increased turbidity in the water column may affect invertebrates through abrasion, clogging of filtration mechanisms, decreased respiration rates or behavioural changes (Gubbay 2003; Birkland and Wijsman 2005; Jones *et al.* 2012; Todd *et al.* 2014). As the food to sediment ratio is also decreased, feeding efficiency of filter-feeding organisms will be affected and more energy is required to sort through particles (Todd *et al.* 2014). A variety of studies have been conducted assessing the impacts of increased suspended sediment on both freshwater and marine invertebrates. Summaries of these impacts have been compiled in reviews (e.g. Jones *et al.* 2012 and Hanson *et al.* 2004). Effects are species-specific and range from no effect, to feeding inhibition, reduced growth and mortality.

Suspended sediments may adhere to eggs of some species, which affects hatching success. Impacts to eggs will depend on the eggs' distribution in the water column, concentration and duration of exposure (Hanson *et al.* 2004; Birkland and Wijsman 2005). Laboratory studies suggest that larvae are less tolerant to excess concentrations of suspended sediments, than eggs of the same species (Hanson *et al.* 2004). Hatching of striped bass (*Morone saxatilis*) and white perch (*M. americana*) (both of which spawn eggs in freshwater environments) was delayed in tests of suspended sediment concentrations of 100 mg/L for 1 day, however egg hatching of Atlantic herring (*Clupea harengus*) (an oceanic spawner) under exposure of sediment concentrations up to 7000 mg/L showed no impact to hatching success (Clarke and Wilber 2000). However, *C. harengus* larvae reared at concentrations above 540 mg/L tended to be small, and those exposed to 19,000

mg/L for 48 hours suffered 100% mortality (Clarke and Wilber 2000). As in adults however, the tolerance thresholds for larvae and eggs will be species-specific.

4.4.2 Sedimentation

Sedimentation is when suspended sediments settle on the seabed, which may bury benthic sessile organisms, or sensitive habitats such as seagrasses and coral reefs. The spatial extent to which sedimentation occurs near a mining site will depend on the quantity of suspended sediment above baseline levels in the water column, the size of the particles and hydrodynamics of the area. Larger particles will settle out more rapidly, while fine sands may be suspended in the water column for some distances, increasing the impact footprint of mining outside the limits of the extraction area (Hitchcock and Bell 2004). Impacts arising from sedimentation include burial/smothering, abrasion, and changes in the physical characteristics of the seabed, which may reduce habitat suitability. Species' life history, morphological and physiological traits, community composition, environment and habitat preferences have been shown to determine their tolerance to sedimentation.

Benthic species are adapted to natural background levels of sediment movement, however increases above their threshold may mean, particularly for sessile invertebrates, they are susceptible to smothering if they are unable to escape or emerge from burial (Miller *et al.* 2002; Gubbay 2003; Jones *et al.* 2012; Hendrick *et al.* 2016). Hendrick *et al.* (2016) demonstrated that the ability for macroinvertebrate species to resist mortality or emerge from burial was dependent on the species and its coping mechanisms, the individual's size, particle size and duration and depth of burial. Increased duration and depth of burial, along with decreasing sediment size resulted in increased mortality. The laboratory study conducted tests on six macro-invertebrate species due to their conservation or commercial importance and distribution in areas where mining occurs; the brittle star (*Ophiura ophiura*), Ross worm (*Sabellaria spinulosa*), Queen scallop (*Aequipecten opercularis*), green urchin (*Psammechinus miliaris*), yellow sea squirt (*Ciona intestinalis*) and an anemone (*Sagartiogeton laceratus*). The least sensitive was the Ross worm, followed by the anemone and green urchin. The brittle star, queen scallop and sea squirt were all highly intolerant to burial. While none of these species are known to exist in the Northern Territory, another species within the genus of *Ophiura* is known to be an indicator species for fine-sandy mud substrates in Anson Bay (*Ophiura kinbergi*) (Smit *et al.* 2000). In the absence of species-specific information from a particular locality, the use of indicator species provides a reliable starting point for setting trigger thresholds for impact mechanisms such as sedimentation.

A review of sedimentation on rocky coast assemblages was conducted by Airoidi (2003). The review revealed that the literature commonly reports on changes in species composition and distribution, inhibition of settlement and recruitment and the decline/mortality/removal of species as the top three effects attributed to sedimentation on rocky coasts. A list of morphological, physiological and life-history traits that have been suggested to increase tolerance to sedimentation in species on rocky coasts was also compiled. These include characteristics such as regeneration of upright portions from remnant bases tolerant to sedimentation, opportunistic cycles of reproduction and growth or erect morphology. Additionally, the presence of species such as mussels and small macroalgae forming turfs that are able to bind and trap sediments may assist in the removal of sediment before depositing on less tolerant species (Airoidi 2003).

Corals are sensitive to sedimentation, though similar to invertebrates, some species are more tolerant than others depending on their ability to self-clean and clear sediment settling on their surface (Ports Australia 2014; McCook *et al.* 2015). Cleaning mechanisms include a combination of polyp distension, ciliary action in addition to the production of mucus sheets (Erftemeijer *et al.*



2012; Pollock *et al.* 2014). Corals tend to be more impacted by low-levels of sedimentation than other benthic species and habitats. Certain life forms, such as horizontal foliose, encrusting and massive forms are more likely to be susceptible to smothering, while upright, branching forms are less likely (Gilmour *et al.* 2006). A review of sedimentation on corals with an emphasis on dredging, conducted by Erftemeijer *et al.* (2012), found that sedimentation rates that can be tolerated by different species ranges from less than 10 mg cm⁻² d⁻¹ to more than 400 mg cm⁻² d⁻¹ and in some species, are able to tolerate more than a month of high sedimentation, and more than two weeks of complete burial. Detrimental impacts to corals as a result of sedimentation include physiological stress, decreased growth, reduction in larval settlement, death of underlying tissue, bleaching and mortality (Erftemeijer *et al.* 2012; McCook *et al.* 2015).

Coral monitoring as part of the Ichthys Nearshore Environmental Monitoring Program (Cardno 2015d) was undertaken over a 2.5 year period. At the majority of monitoring sites, there were no differences indicative of a decline in coral cover from the baseline to the post-dredging phase, with the exception of South Shell Island and Weed Reef 2, as a result of a natural thermal bleaching event for the latter. Overall, the measured potential impacts of dredging was summarised as causing:

- A temporary increase of sediment on corals at the end of the Dredging Phase at South Shell Island. Partial mortality decreased or remained unchanged in the subsequent surveys indicating the sediment cover was temporarily overlaying live coral tissue that was exposed in subsequent surveys; and
- Potential suppression of recruitment at South Shell Island due to indirect effect of increased turbidity (from dredging) at the site, and the susceptibility of coral recruits to sedimentation.

Sponges show comparatively higher resistance to impacts from both turbidity and sedimentation than corals. Similar to corals however, sponges show morphological adaptation in connection with high sedimentation rates (DHI 2010). A recent review has been completed by Schonberg (2016) as part of the Western Australian Marine Science Institution (WAMSI) dredging science node. From an ecological perspective, turbidity and sedimentation were likely to alter the structure of filter feeding communities by reducing fitness and survival from both dredging related and natural processes. There are sponges that are well adapted to living in turbid environments and may continue to survive at dredging sites. These include endopsammic sponges (living partially buried within sediments), fast growing species that are able to change their morphology (highly plastic), and species with erect growth forms. Sediment tolerance may also be related to species that are more capable of keeping their surfaces sediment-free (Schonberg 2016).

A review of the literature on the effects of sedimentation on seagrasses by Cabaço *et al.* (2008) found that the capacity of seagrasses to tolerate sedimentation and burial was size-dependent with mortality rates related to shoot mass, rhizome diameter, above-ground biomass, horizontal elongation rate and the size of leaves. Larger species, such as *Posidonia* have a greater capacity to withstand burial. Impacts to seagrasses as a result of sedimentation include an increase in shoot mortality, decreases (and increases) in growth rates, decreased biomass, increased branching frequency and decreased productivity (Cabaço *et al.* 2008).

Seagrasses monitored as part of the Ichthys Nearshore Environmental Monitoring Program (Cardno 2015a), were predominantly *Halophila* spp. (including mostly *Halophila decipiens*) and *Halodule* spp. (including *Halodule uninervis*). The post monitoring study concluded that there was no potential influence from project dredging activities on seagrass distribution and the



documented changes in the distribution and percentage cover of *Halodule* and *Halophila* were primarily driven by the influence of climatic seasons (dry and wet season).

Halodule sp. can tolerate moderate disturbance and are ephemeral with rapid turn-over and high seed set. It is also considered a pioneer species, growing rapidly and surviving well in unstable and depositional environments. *Halophila* spp. can propagate through budding, but primarily relies on a buried seed bank for population re-establishment in seasonally fluctuating or high disturbance environments. It is a highly fecund, annual and opportunistic species (Kenworthy 1993) that may be favoured by disturbance and is tolerant of sedimentation.

Newell and Woodcock (2013) also reviewed the impacts of sedimentation on benthic species. They concluded that generally, fauna found in a particular location are adapted to survive in the environmental conditions of that area. For example, species that characterise mobile sandy gravels tend to be more capable of emergence from burial or tolerant to sedimentation, while those in high energy, environments, with coarser sediments are less tolerant (Newell and Woodcock, 2013; Todd *et al.* 2014). Therefore, the type of dredging and the environment in which it occurs will have varying effects. Aggregate dredging, without screening, is unlikely to result in changes of community composition provided there is sufficient material remaining and species are able to recover and recolonise the area. However, long-term changes in the sediment composition occurring, for example from material returned to the seabed (discussed in Section 4.1.2) as a result of screening, may have more permanent effects on the benthic communities tolerance and ability to recolonise an impacted area.

4.4.3 Changes in Water Quality

As a result of sediment disturbance associated with seabed mining, organic matter, nutrients, heavy metals and other contaminants may be released into the water column, altering water quality in the vicinity of operations. The mining process may also reduce oxygen availability for fauna in the water column and at the seabed.

4.4.3.1 Organic Enrichment

During extraction, or from overspill and screening, organic matter and nutrients bound to sediments will be released into the water column. This can lead to eutrophication, particularly where nitrogen and/or phosphorous is introduced given plankton growth is nitrogen and phosphorous limited (Boesch 2001; Garrison 2007). The increase in nutrients can stimulate the growth of phytoplankton. Once the organisms begin to decay however, they deplete oxygen levels, which can produce hypoxic or anoxic conditions (Charlier 2002; Phua *et al.* 2004; Garrison 2007; Johnson *et al.* 2008). As the decaying matter accumulates on the seabed, oxygen is consumed at the base of the water column, which is not replenished from surface waters (Boesch 2001). Deep excavation pits, such as those caused by static dredging, may have increased water residence times, particularly where the water in the pit is stratified from the rest of the water column (Birkland and Wijsman 2005). Organic matter may accumulate in pits, further decreasing the availability of oxygen or creating anoxic conditions of the water near the seabed which may render the area unsuitable for fauna to spawn, feed, settle and recolonise (Birkland and Wijsman 2005; Johnson *et al.* 2008). Decreased oxygen content at the seabed can also arise from the release of contaminated or disturbance of anaerobic sediment layers during extraction (Gubbay 2003; Johnson *et al.* 2008).



Hypoxic (and anoxic) conditions can kill organisms or lead to persistence of stress-tolerant species, reducing the biological quality of the ecosystem (Garrison 2007; Johnson et al. 2008). Community composition may change, as species that have a higher tolerance to organic loading, such as polychaetes may become dominant compared with less-tolerant species, such as arthropods (Lenihan et al. 2003). Hanson et al. (2014) report that dissolved oxygen concentrations below 6 ppm may adversely affect exposed fish or macro-invertebrates. Nutrient enrichment can also lead to increased macroalgal growth in shallow waters with ample light penetration (Boesch 2001). It may also result in blooms of microalgae.

Birkland and Wijsman (2005) suggest that nutrient and organic content of sediments are an inverse function of the sediment particle size; that is, sand and gravel, particularly in high-energy environments, have lower organic content and are well-oxygenated, compared with fine sands in low-energy systems. This is supported by a study conducted by Krause et al. (2010) who found that following dredging, fine sands became dominant and organic content increased as a result. Furthermore, oxygen deficiencies persisted in the furrows, 6-10 months following cessation of dredging.

Organic enrichment resulting in widespread impacts on water quality from the release and mobilisation of nutrients in dredged sediment usually only requires detailed consideration where dredging occurs in urbanised areas or in enclosed waterways and embayments that receive large catchment run-off. This is consistent with McCook et al. (2005) who indicate that quantification of the contribution of released/mobilised nutrients should be part of assessments of large dredging projects or where there is a potential for high sediment nutrient concentrations, for example close to river mouths or nutrient point sources.

4.4.3.2 Release of contaminants

Dredging in particular has received recent attention, due to the removal and disposal of potentially contaminated sediments (Hanson et al. 2004; Bray 2008). The mobilisation of sediments into the water column as a result of dredging may release heavy metals that would have otherwise remained buried (Phua et al. 2004).

Sediment-bound contaminants (both organic and inorganic) could potentially desorb during the dredging process or when sediment is entrained in plumes associated with the dredging and disposal operations. However, this is unlikely to be a major issue with dredging, as studies have shown that metals that desorb from disturbed sediment (e.g. by dredging) tend to bind to fine silt and clay; therefore, screened fractions are particularly likely to contain elevated levels of contaminants (Hanson et al. 2004). Johnson et al. (2008) however suggest that mining of offshore sand and gravel do not typically release high levels of contaminants compared to estuaries or nearby coastal areas, due to the distance from sources, such as rivers and Ports. Irrespective, where mining involves the extraction of an ore body (rather than sand and gravel) the chemical properties of tailings and waste rock that are disposed to sea require detailed assessment to ensure that release of metals into the water column can be quantified. This is considered in greater detail in Section 4.5.

One of the more severe impacts relating to the release of contaminants from the seabed is where toxins are accumulated through the food chain (Bray 2008; Jakimska et al. 2011) however organisms may also be affected through direct absorption, or consumption of suspended sediment. The toxicity of metals however, is dependent on the contaminant, its concentration, chemical/geochemical state and the bioavailability of the toxin to uptake by marine organisms



(Johnson et al. 2008). Dissolved forms are regarded as the most bioavailable, but is species-dependent (Jakimska et al. 2011). Some heavy metals are essential for the growth and functioning of many organisms but have the potential to become toxic in high concentrations, such as iron and manganese, while others such as cadmium and mercury are toxic even in low concentrations (Jakimska et al. 2011). Some metals in toxic concentrations can adversely impact cellular functions and lead to dysfunction of the endocrine system, which can affect reproduction, metabolism, the immune system and growth (Hylland 2006; Jakimska et al. 2011).

Hydrogen sulphide is a highly toxic, naturally occurring constituent of sediment pore water. It is not treated as a contaminant of concern for the evaluation of dredge material as it normally undergoes rapid oxidation and dilution during the dredging and disposal process (Sims and Moore 1995). Contaminant concentrations in open oceans are likely to be low as they are diluted within the water column (Penney et al. 2008). An example of bioaccumulation of toxins through the food web is the consumption of seagrasses by fish or mammals, such as dugongs or turtles (Lanyon et al. 1989). Seagrasses are known to accumulate metals (e.g. Filho et al. 2004) which in the case of metals like mercury, lead and cadmium have the potential to bioaccumulate in species that feed on seagrass that can result in lethal and sub-lethal effects on higher order consumers. Penney et al. (2008) suggests that the impacts associated with the release of contaminants as a result of diamond mining is low and chemical contamination is unlikely however this will need to be assessed on a case by case basis depending on the type of product mined and the environmental conditions at the mine site.

4.4.4 Emissions and Discharges

Emissions and discharges resulting from seabed mining are likely to be analogous to those commonly encountered by the general shipping industry and artificial light and air emissions are inevitable. Shipping is a major source of oil spills worldwide, from normal operations, but also as a result of refuelling.

4.4.4.1 Light emissions

Light emissions relating to seabed mining in the Northern Territory would predominantly relate to the use of transient vessels and any ancillary plant where mining and offshore processing may occur and would therefore be localised. On-board lighting would emanate 24 hours a day for safety purposes in accordance with the requirements of the *Navigation Act 1912*. Lighting would be visible, particularly during the evening and night from the coastline.

Artificial lighting has the potential to influence the behaviour of marine fauna, by avoidance, disorientation or interrupt reproduction (Davies *et al.* 2014). Light is important in the settlement of invertebrate larvae (Davies *et al.* 2015), is an essential cue in broadcast spawning of coral species (Jones *et al.* 2015b; Kaniewska et al. 2015) and thus, artificial light may therefore disrupt the development of ecological communities. Furthermore, artificial light is known to disorient migratory birds and can result in vessel strikes (Merkel 2010).

Marine turtles, particularly reproducing females and hatchlings can be affected by artificial light (EPA 2010; Davies et al. 2014). Artificial light close to nesting beaches disrupts the orientation of hatchling turtles after they emerge from the nest, either causing movement in the wrong direction or preventing the hatchling from discerning which direction to travel (see McCook et al 2005). Marine turtles in the North Marine Region, specifically the flatback, green, hawksbill and olive ridley turtles (DSEWPaC 2012) are most vulnerable. Recently a buffer of 1.5km has been



recommended for industrial development nearby known turtle nesting sites (Kamrowski et al. 2014; Pendoley and Kamrowski 2016). While less of an issue for transient vessels, if the mining plant is permanently moored or operating offshore, then assessment of the risk should be considered for each locality.

4.4.4.2 Air emissions

Air emissions and subsequent changes in air quality relating to seabed mining will be related to the use of vessels. Changes to air quality resulting from vessel use is analogous to the impacts associated with commercial shipping, in that burned fuel in vessel engines and other on-board equipment (i.e. incinerators, generators etc.) will result in the gaseous emissions of CO₂, CO, SO_x and NO_x. Such emissions can lead to a reduction in local air quality, and in high concentrations may be toxic, odoriferous or aesthetically displeasing, though impacts of such emissions generally only affect human health (e.g. Corbett et al. 2007). Atmospheric fallout of particulates are another potential source of contaminants.

4.4.4.3 Chemical discharges and hydrocarbon spills

Spills and accidental discharges of any chemical, including fuel, may arise as a result of accidental collisions, refuelling, during transfers between vessels and barges or carriers or from leaks or spills from equipment. The release of potential contaminants depends on the type of material being mined, chemicals used in the preparation and extraction processes, and poor environmental management practices.

Diesel is the most commonly used fuel in commercial shipping and Michel et al. (2013) list the following general behaviour, fate and effects of diesel spills in open waters:

- Low viscosity and therefore will be readily dispersed through the water column given diesel is considerably lighter compared with seawater. Diesel cannot sink to the seabed and accumulate, though can adhere to fine-grained sediments and subsequently settle. This is unlikely in open marine waters.
- Shoreline clean-up is rarely required as diesel can be washed off quickly by waves and tidal flushing.
- Diesel is readily degraded by naturally occurring microbes within 1-2 months in open water.
- Diesel is one of the most acutely toxic oil types. Fauna and flora in direct contact with diesel will likely die, though small spills are rapidly dispersed and therefore fish kills have not been reported in open water contexts.
- Crabs and bivalves can bioaccumulate the oil, but will also depurate it within weeks after exposure.
- Sea birds may be exposed through direct contact, particularly if spills occur near nesting colonies or areas of high abundance. Ingestion may occur during preening.

The impacts of an oil spill on biological resources will depend on the ambient conditions of the area, the presence (or absence) of sensitive species and the accumulation or bioavailability of a contaminant to susceptible marine species. A review of the biological impacts that may arise as a result of ship-based oil spills is described by ITOPF (2011) and is summarised below.



Plankton - Plankton, comprising bacteria, plants, and the larvae of fish and invertebrates, are unlikely to be sensitive to oil spills at a community or population level, given the likely over-production of young life stages that provide a buffer for recruitment from adjacent unaffected areas.

Fish - Juveniles are likely to be sensitive to oil spills, however adult fish are considered much more resilient as effects on wild fish stocks have not been detected and mass mortalities are rare, particularly in the open ocean. High and localised concentrations along a shoreline, or in rivers may have more catastrophic effects.

Seabirds - Seabirds are considered the most vulnerable to oil spills, particularly those who raft together in flocks. Fouling of plumage is the highest effect, as direct exposure may affect their buoyancy and insulation, leading to hypothermia and/or drowning. Preening to clean themselves will lead to ingestion of oil, which will have serious effects on the animal.

Marine Mammals and Reptiles - Cetaceans and turtles may be exposed to floating oil while breathing or breaching. Reports of effects to manatees or dugongs are rare. Species that rely on fur to regulate their temperatures will be the most vulnerable. Loss of turtle eggs and hatchlings may occur if oil reaches nesting shorelines. Adult turtles may suffer mucus membrane inflammation.

Seagrass - Floating oil is likely to pass over seagrass meadows without affect, however if oil is mixed at sufficiently high concentrations and moves to the seabed, then seagrass and associated fauna may be impacted.

Corals – corals are highly sensitive organisms, the highest risk is when turbulence from waves breaking disperses spilt oil or when dispersants are used.

Rocky and Sandy Shores – scouring effects of wave action and tidal currents mean that these shorelines are the most resilient to oil spills. It is noted that generally, recovery is quick and the species are able to reoccupy and recover within a matter of weeks.

Soft Sediment Shores – fine sands and muds typically occur in areas with low wave energy and are therefore any oils that flocculate and penetrate vertically through the sediment may persist for many years increasing the likelihood for long-term effects on the community.

Saltmarshes – A single event of contamination is unlikely to cause more than temporary effects, however persistent longer-term damage may occur by repeated or chronic oiling, or by clean-up activities. Clean-up activities that may impact saltmarshes include trampling, use of heavy equipment or sediment removal. Where bulbs or roots are not impacted, seasonal regrowth should be expected.

Mangroves – mangroves are highly vulnerable to oil spills as inundation of root systems may block oxygen supply, leading to mortality of the mangrove system. Mangroves with higher aeration may have a higher tolerance to oil. Toxic components of oil may interfere with species' ability to maintain salt balance.

Recovery time after oil spills is a function of the affected systems complexity and resilience (ITOPF 2011). Table 4-2 shows that the time taken to recover will depend on the quantity and type of oil, and the extent of the damage inflicted.

Table 4-2 : Indicative recovery times after oiling for various habitats (from ITOPF 2011)

Habitat	Recovery Period
Plankton	Weeks/months
Sandy beaches	1-2 years
Exposed rocky shores	1-3 years
Sheltered rocky shores	1-5 years
Saltmarsh	3-5 years
Mangroves	10 years +

4.4.5 Marine Pests

Marine (or aquatic) pests are described by the Northern Territory Government as 'non-indigenous plants or animals, whose introduction, does or is likely to, have a significant detrimental impact on the environment' (DPIF 2014), and therefore not all non-indigenous species are considered to be pests or invasive species. Any vessel entering Northern Territory waters has the potential to act as a dispersal vector for marine pest species, and the risk is not limited to seabed mining vessels. Introduction of non-native species is largely related with Ports and therefore coastal environments are more susceptible than open ocean habitats (Paulay *et al.* 2002). Introductions may occur via ballast water exchange or through fouling of vessel hulls or gear. Bax *et al.* (2003) list hull fouling as the most probable mechanism for introduction and establishment of marine pest species in Australia.

It is important to note however, that not all introduced species will successfully colonise and establish in their receiving environment. This may be due to the fact that they cannot find suitable substrate; conditions may not be optimal for the species settlement and growth; or due to lack of prey availability. Therefore, pest species tend to be opportunistic, have high tolerance for a variety of environmental conditions, are generalists and pioneering (Hutchings *et al.* 2002). Seabed communities and benthic environments are particularly susceptible to colonisation from marine pests in areas where mining is occurring as the landscape is disturbed and fragmented, enhancing the likelihood of a pest species establishing (Sakai *et al.* 2001; Paulay *et al.* 2002; Hanson *et al.* 2004).

4.4.5.1 Marine pests in the Northern Territory

Due to its proximity to Asian ports and its use as the first port of call from Asia, Darwin in particular is considered at high risk of invasive marine pests (DRDPIFR 2008). The Department lists the following three species as major threats to Northern Territory Waters (Northern Territory Government 2016):

- Asian bag mussel (*Musculista senhousia*). This species has not been found in Northern Territory waters, but has successfully established in the Swan River, WA and in Port Philip Bay, Western Port Bay and Portland in Victoria. It has also been reported in the Tamar River in Tasmania.
- Asian green mussel (*Perna viridis*). No known established populations exist in Australia, though it has been detected in Cairns in 2001 and 2013. Native to tropical Indo-Pacific, the species has successfully spread to other countries via ballast water exchange.



- Black-striped mussel (*Mytilopsis sallei*). This species was first identified in Darwin marinas in 1999 and has a high potential to cause economic and environmental damage due to its ability to settle on almost any surface to the exclusion of all other species (DRDPIFR 2008). The eradication of the species from Darwin Marinas was the world's first successful eradication of an established marine pest population.

Australia's National System for the Prevention and Management of Marine Pest Incursions has developed National Control Plans for six of the nation's highest threat marine pest species, including the Asian bag mussel (*M. senhousia*) (Department of Agriculture 2014). Presence of *M. senhousia* has the potential to impact clam and cockle fisheries (Aquenal Pty Ltd 2008a). Its impacts are likely to be limited to the preferred low energy habitats, avoiding open coastal areas (Aquenal Pty Ltd 2008a). The Asian green mussel (*P. viridis*) and black-striped mussel (*M. sallei*) are not part of the National Control Plan. The Asian green mussel and black-striped mussels are known to be detrimental commercially by clogging water intake pipes (Ray 2005; Northern Territory Government 2016). *P. viridis* may also harbour microalgal species that produce toxic shellfish poisoning (Ray 2005).

Surveys (2007/2008) by the Northern Territory's Department of Regional Development, Primary Industry, Fisheries and Resources (DRDPIFR) did not find any marine pest species in Darwin Harbour (DRDPIFR 2008). A targeted survey completed in 2010, however, found two potential pest species; a single pacific oyster shell (*Crassostrea gigas*) was found (though likely a discarded shell that was imported for human consumption) and the tubeworm *Hydroides sanctaecrucis* (Golder 2010). Investigations of Darwin Harbour relating to the INPEX project also found no marine pest species that posed a threat to biosecurity (Cardno 2015b).

4.4.5.2 Risks and impacts of marine pest introduction

Successful establishment of a marine pest involves introduction to a new habitat, colonisation/establishment and range expansion or dispersion (Sakai *et al.* 2001). Furthermore, not all introduced non-native species will become established and impact the existing ecological environment (e.g. the Asian green mussel has failed to establish in Cairns, despite detection on two occasions (Northern Territory Government 2016)). Impacts arising from marine pests are commonly discussed in the literature as it is identified as a key anthropogenic threat, though no known direct link has been found between seabed mining and marine pest introductions or dispersal.

The risks that currently exist with dredging vessels that are sourced from overseas would equally apply to seabed mining vessels. Marine pests may cause displacement of native species. For example, the alga *Caulerpa taxifolia* (which has both a native and non-native distribution in Australia) has been shown to compete with native seagrasses as it directly competes for light and space, colonising large areas very rapidly (de Villele and Verlaque 1995). *C. taxifolia* is more tolerant to increased turbidity and reduced light attenuation compared with seagrasses (Burfeind *et al.* 2009) and it has been suggested that sparse seagrass beds of *Posidonia* are susceptible to invasion by *C. taxifolia* compared with more dense beds (Villele and Verlaque 1995; Cresse *et al.* 2004). *C. taxifolia* have also been shown to be able to colonise areas void of vegetation where seagrass once existed (Burfeind & Udy 2009), reducing the ability for the seagrass to recolonise an area thereby potentially altering ecological function. The Japanese Seaweed, *Undaria pinnatifida*, while in Australia has not been shown to have negative environmental or economic impacts, however internationally has been shown to compete with native seaweeds for space, altering species richness and diversity (Aquenal Pty Ltd 2008b).



Marine pests have the ability to drive ecological change by creating or modifying existing habitats and their assemblages, altering ecosystem function and trophic structures to the potential detriment of native species in the area, reducing overall biodiversity. A great example of this is the zebra mussel, which has become a serious pest in the US. The species attaches itself to native mussels in extreme densities, outcompeting natives for food, space and oxygen (Pimentel *et al.* 1999).

Marine pests can also have economic ramifications, particularly biofouling species. Of threat to the Northern Territory, is the black-striped mussel which fouls boat hulls and subsequently leads to inefficiencies due to increased drag, reduced speed in addition to the direct damage of the hull surface (Northern Territory Government 2016). Furthermore, black-striped mussels previously have clogged cooling water intakes of vessel engines, causing damage to motors from overheating (Northern Territory Government 2016).

Economic consequences also arise when marine pest species predate upon or compete with commercially important species, affecting their abundance and distribution. In Australia, the European green shore crab (*Carcinus maenas*) and the European fan worm (*Sabella spallanzanii*) are listed as a marine pest of concern (Department of Agriculture 2014). The shore crab is known to affect commercially grown species of mussels, oysters and clams, though at this stage, the impact on commercial fisheries in Australia is considered low, likely due to animals being grown in cages or on lines suspended in the water column (Aquenal Pty Ltd 2008c). The fan worm has known to have impacted the scallop fishing industry in Port Phillip Bay, Victoria, as dredges used to recover the scallops became clogged with *Sabella* thereby increasing the sorting time required for catches (Aquenal Pty Ltd 2008d). The current impacts on the fishery are not of concern since it closed in the mid-1990's following increased concerns over the environmental impacts of dredging.

Certain habitats and communities are less susceptible to pest invasions. Ecosystems with high species diversity may be more persistent and stable as there are fewer niches available for establishment (Hutchings *et al.* 2002; Tan & Morton 2006).

4.5 Impacts from Tailings Disposal

The mining of ore from the seabed could require disposal of overburden or tailings back to the seabed. While it may be possible or practical to dispose of material to land, the following section provides a quick overview of the impact pathways associated with disposal to sea.

There are many potential risk pathways that have been discussed in relation to the potential impacts associated with seabed mining and they can also equally apply to the process of tailings disposal as shown in Figure 4-15.

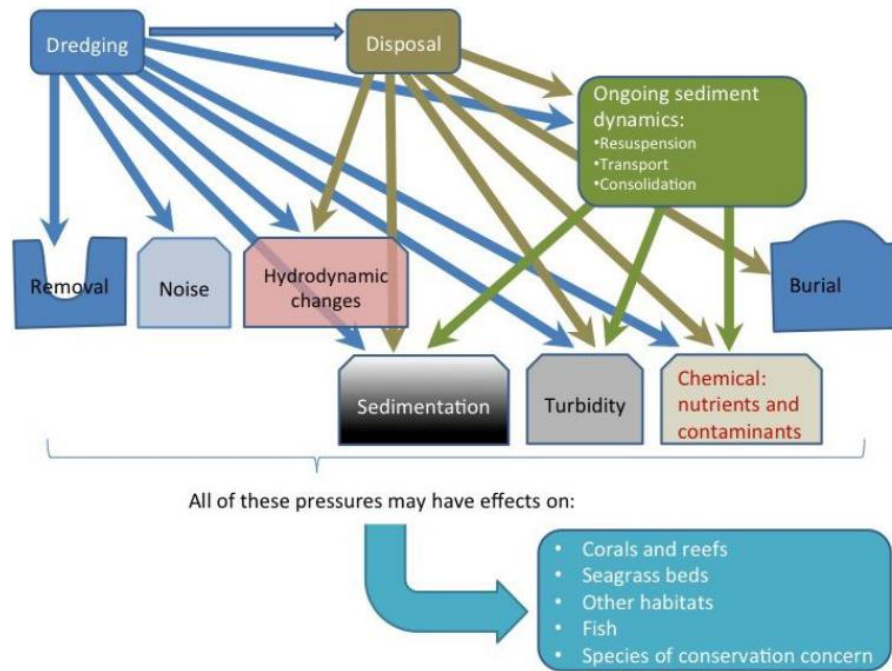


Figure 4-15 : Relationships between the activities of dredging and dredge material disposal, the major (potential) pressures and their effects on key habitats and other biodiversity values of the Great Barrier Reef World Heritage Area (from McCook et al. 2015)

There are four common options for offshore disposal of dredged material which are also likely to apply to disposal of mine tailings and overburden, being;

- Unconfined disposal into licensed disposal areas
- Confined disposal by controlled dumping into seabed depressions or redundant borrow pits
- Confined disposal onto seabed between constructed bunds, and
- Formation of islands.

Potential impacts associated with offshore disposal of dredged material vary according to the disposal method employed and the characteristics of the receiving environment. Potential detrimental effects generally fall into two categories, namely water column effects (i.e. exposure to suspended sediments) and sedimentation effects. Sedimentation is the deposition of sediment over benthic habitat and is measured as either the rate of sediment accumulated per unit area of substrate (i.e. g/m²/hr), or as overburden thickness (i.e. millimetres above the pre-existing sediment horizon). Sedimentation can affect submerged aquatic vegetation, mangroves, shellfish (crustaceans and molluscs), corals and tropical coral reefs and fishes and have been considered in greater detail in Section 4.4.2.

Unconfined offshore disposal can result in temporary impacts, which include increased turbidity, reduced water quality, smothering of seabed flora and fauna, damage to fish and fish breeding areas and the possible fouling of nearby beaches, and permanent impacts, such as altered water and sediment movements from changes to seabed topography and the possibility of increased coastal erosion at near-shore disposal sites. Additionally, onshore disposal at reclamation sites can result in changes to the hydrodynamic, sediment transport and groundwater regimes of the area, odour and visual impacts and various ecological impacts, including habitat destruction.

The study by Asmend and Johansen (1999) provides an example of the short and long term environmental effects from marine tailings and waste rock disposal into the marine environment. The study was based on the discharge of tailings from a lead/zinc mine into sea below 60m water depth, after a land-based tailings disposal system could not be constructed. Widespread contamination of bottom waters ensued as the dissolution of lead and zinc from the tailings and waste rock was underestimated, which then led to bioaccumulation of metals in seaweeds and mussels. Dust from the ore crusher and conveyor that transported concentrate from storage to the bulk carrier was also identified as a third significant source of metal pollution in the marine environment. The issue of dust may not be a significant issue for mining that is undertaken exclusively at sea but would require consideration where land-based processing is required.

Other studies such as Stronkhorst et al (2003) assessed the environmental impact of the long-term disposal of moderately contaminated dredged material into dumping sites located in 18-21m water depth in the North Sea. Abundance and diversity of benthic species were relatively low for the duration of disposal activities. One year after the cessation of the disposal of silty harbour sediments, the sediment texture at dumping site North had recovered rapidly, almost attaining the reference conditions. The rapid redistribution of the fine sediment layers from the dumping site to other areas of the North Sea was achieved by dynamic transport along the seabed due to the shallow and high-energetic nature of the environment. Rapid recolonization of the benthic invertebrates ensued and was the result of the ending of physical disturbances (burial, smothering) and a decrease in silt content of the sediment. The study concluded that benthic macrofauna at and near the disposal sites was primarily and adversely affected by physical disturbance (burial, smothering) rather than the presence of contaminants.

Overall, the most common observation from disposal operations is a decrease in macrofaunal abundance and species richness at the disposal site (Bolam et al., 2006; Powilleit *et al.*, 2005; Witt *et al.*, 2004) during the disposal activity, followed by recovery that can vary from months to years. Sedimentologic composition is usually the primary determinant of community structure at the disposal area as sediments are often homogenised and dispersed as they settle to the seabed.

4.6 Cumulative Impacts

The definition of cumulative environmental impacts varies within the literature, but the consensus is that generally, it is the successive, incremental and combined effects on a resource, ecosystem or society that accumulates through space and time (Franks *et al.* 2010). These impacts can be either, or both positive and/or negative and understanding the causation and linkages between impacts is a critical aspect in the management of them (Franks *et al.* 2010). For the purpose of this report, only negative environmental cumulative impacts will be discussed, however it is acknowledged that there will be social, commercial and economic cumulative effects arising from and interacting with environmental impacts. Impacts from climate change, including sea level rise are also a consideration in assessment of cumulative impacts but are outside the scope of this study.

4.6.1 Types of Cumulative Impacts

The cumulative environmental impacts of seabed mining may arise via:

- The compounding activities of a single operation
- Multiple mining operations; and the
- Interaction of mining operations with other activities.



There is also likely to be some combined effects of one, two or all three of these in which impacts may further accumulate and compound in the environment. The interaction of these cumulative impacts will operate differently, depending on the activities and effects being considered and may have direct or indirect consequences (Halpern *et al.* 2008).

4.6.1.1 A single operation

A single mining event may produce multiple environmental stressors or a single operation (i.e. the mining of one particular resource in a single location) could also trigger successional impacts (Franks *et al.* 2010). Any one impact or stressor by itself, may have minimal impact to exposed flora and fauna, however the interaction of several factors are also likely to have a larger, longer-lasting overall effect (Hanson *et al.* 2004; Halpern *et al.* 2008). An example of a triggered successional impact is increased turbidity inducing eutrophication, which may lead to hypoxic or anoxic conditions on the seabed (see Section 4.4.3.1).

Multiple types of stressors when combined are also likely to induce a species, or community to reach its critical threshold before it otherwise may have. For example, a species or community already living at its upper tolerance limit may reach a critical limit when subjected to further disturbance such as seabed mining, even if it is just a once off event or operation (Hitchcock and Bell 2004).

4.6.1.2 Multiple operations

Cumulative impacts induced by multiple seabed mining operations may involve either multiple concurrent mining operations in space and/or multiple operations through time. Gubbay (2003) notes that recovery times of impacted species may be increased where there is disturbance in adjacent areas. Furthermore, cumulative effects can occur on the seabed where resources are already being exploited. For example, bottom trawling occurs for the Northern Prawn Fishery over extensive areas in the Northern Territory (Deng *et al.* 2005). Similar to shallow seabed mining, the primary environmental impact of bottom trawling is direct disturbance to the seabed (see Jones 1992 for review) and therefore, cumulative impacts from multiple operations (either occurring concurrently or through time) affecting the seabed may occur and should be considered.

Impacts can also accumulate through time, where sites may be repeatedly mined, inducing different community impacts compared with short term or once-off events (e.g. Fraser *et al.* 2006). An increase in general vessel traffic in the ocean will increase the risk of vessel strikes to marine fauna and migratory birds (Phua *et al.* 2004), in addition to increasing ambient noise and light levels. However, while it is somewhat well understood that light affects sea turtles (EPA 2010; Davies *et al.* 2014), data on the hearing of sea turtles is quite limited (Lenhardt 2002; Popper *et al.* 2014). Therefore, the cumulative impacts of increased vessel traffic need to be interpreted with caution. Furthermore, the length of time a dredge vessel spends in an area is considerably longer than transient merchant ships (Robinson *et al.* 2011) and therefore, the effects of increased vessel traffic in the coastal waters of the Northern Territory, particularly from mining vessels will have the potential to compound.

4.6.1.3 Interactions with other activities

Seabed mining impacts may interact and accumulate in areas where other activities are providing additional stressors on the ecosystem. Activities that seabed mining may interact with resulting in



cumulative impacts include fishing, coastal runoff, coastal developments, maintenance dredging and general commercial vessel traffic. The impacts of these, and other activities when combined with seabed mining are summarised in Table 4-3.

Table 4-3 : Possible impacts of other activities when combined with seabed mining (adapted from Posford Duvivier Environment and Hill 2001)

Potential Impact	Fishing	Eutrophication	Coastal Alteration	Waste and spoil disposal	Capital & Maintenance Dredging	Large Vessel Anchoring	Other Aggregate Extraction Areas
Changes in Hydrodynamics	✓		✓	✓	✓		✓
Changes in Sediment Transport			✓	✓	✓		✓
Increased Turbidity	✓	✓		✓	✓		✓
Sedimentation	✓	✓		✓	✓		✓
Removal/Alteration of Habitat	✓	✓	✓	✓	✓	✓	✓
Modification of Sediment Characteristics	✓	✓		✓	✓	✓	✓
Decreased Water Quality		✓		✓	✓		✓
Increased Primary Production	✓	✓			✓		✓
Marine Pests	✓				✓	✓	✓
Other Emissions and Discharges	✓			✓	✓	✓	✓
Increased Noise	✓				✓	✓	✓
Entrainment and Collisions	✓				✓	✓	✓

4.6.2 Cumulative Impacts from Seabed Mining in the Northern Territory

Multiple and successive environmental impacts from existing pressures combined with the potential impacts arising from future seabed mining may result in significant cumulative impacts that may not have otherwise been expected. Furthermore, impacts to a single organism or species may be less concern compared with the cumulative effects on the wider ecosystem. The total cumulative effect of an event, activity, operation and the mechanisms in which it will interact with other existing stressors will be location based and specific to the activities that are occurring. However, examples of environmental cumulative impacts that may arise as a result of seabed mining in the Northern Territory can be hypothesised based on the review of environmental impacts and the existing environment. These may include:



- Modification of seabed topography, increasing wave attenuation and subsequent beach and coastal erosion
- Modification of sediment characteristics (such as sediment granulation), altering species richness, biomass and diversity
- Increased suspended sediment loading leading to reduced light attenuation, particularly in shallow waters
- Introduced marine pest species displacing native species through competition for light and space, thus changing the community ecology of the area
- Alterations to predator-prey relationships and distributions as a result of removal or alteration of species' habitats
- Modification of the spatial distribution and extent of habitats (through direct removal or fragmentation of habitats) alters the distribution and extent of the fauna that utilise those habitats (i.e. impacts to seagrass meadows affecting critical dugong and turtle feeding grounds)
- Exacerbated environmental impacts as a result of several operations within close proximity to each other; or
- Repeated mining operations through time increase recovery periods, and subsequently the likelihood of the ecosystem returning to its pre-impacted state is reduced.

Overall, understanding the effects of these cumulative pressures is extremely challenging as the multiple pressures may interact in complex ways, generating effects which are greater, and much more difficult to predict, than a simple summation of individual impacts (McCook et al. 2015).

5 Establishing Acceptable Level of Impact

At the heart of environmental impact assessment is the question of how much environmental impact should be tolerated. This is a difficult issue to address and while the overall environmental objective of a project is clearly "sustainability" or "sustainable use," the operational definition of these terms is situational dependant on the type of project being assessed and possibly any number of social, economic and environmental factors (<http://www.gdrc.org/uem/e-mgmt/6.html>). In determining an acceptable level of impact, it is important to understand the extent and duration of the mining activity and the sensitivity and vulnerability of the receiving environment. Once understood, one way of offsetting the threshold of environmental impact is to implement appropriate mitigation measures which can be applied to eliminate, reduce or control the adverse environmental effects of a project.

5.1 Use of Conceptual Models for Defining Impact

Conceptual models are used in the environmental management of industrial activities, particularly in the management of industrial contaminated sites in the United States, Europe and Australia. The idea however can be equally applied for seabed mining. As the name suggests, it is a concept of the current understanding of the environmental conditions related to a project location, and as such includes both a repository of information and an interpretation/synthesis of this data. The conceptual model for a project contains all information necessary for decision- making related to the mining project and its environmental management, including environmental data and syntheses, the mine plan, past and present mining activities and equipment, and previous decisions (Durden et al. 2017). Most importantly, the conceptual model allows for defining exposure pathways and impact to sensitive receptors (including cumulative impact). Typical sources of information that would be relevant to a conceptual model for a seabed mining project are summarized in the table below:

Table 5-1 : Information required for Conceptual Model (modified from Durden et al 2017)

Type of Information	Parameters
Location and size of project area	Location, size
Bathymetry	Depth, other benthic features
Climate	Physical, Chemical
Pelagic, midwater (metocean) conditions	Physical, Chemical, Biological
Benthic Conditions	Physical, Chemical, Biological
Surface and sub-surface geology	Physical, Chemical, Biological
Hazards (potential and historical)	Natural and anthropogenic, scale and type of impact
Protected areas within (and outside) claim area	Physical, Chemical, Biological

Type of Information	Parameters
Details of planned mining activity	Scope, design, type, timeline, environmental risks
Risks and Areas of Potential Conflicts	Location, size, frequency and nature of conflict
Planned mitigation works	Location, type, size, anticipated result details of monitoring to measure results
Results of current and previous monitoring programs	Aims, scope, results, implications
Adherence to current regulations	Information of actions to adhere to regulations and records of any pollution incidents
Implementation of BAT	Details of technology and its implementation
Stakeholder inputs	Formal and informal, details of input
Summary of findings to date	Synthesis of all information above, including rationale for interpretation
Uncertainties and items requiring further investigation	Scale and relationship to items above and timeline required for certainty

The main benefit of the conceptual model is to have all available data and interpretations at hand as an evidence base for decision-making. It also facilitates ensuring that outcomes of previous decisions are adequately addressed, thus avoiding the repetition or propagation of errors. In a phased or staged mining scenario, the conceptual model becomes central to managing activities in different phases at different locations within a mining lease. The conceptual model is thus a growing repository of site and activity specific information supported by transparent data which is updated regularly. By updating the model with information as it becomes available it also allows a more reliable and realistic assessment of the potential impacts associated with the proposed mining project, rather than relying on a worst case scenario which may be totally unrealistic. This in turn can allow for a more targeted approach to monitoring and mitigation (Durden et al. 2017).

5.2 Predicting Zones of Impact

The Western Australian (WA) EPA dredging guideline (EPA 2011) which is also referenced in the NT dredging guideline (NTEPA 2013) makes use of a spatially-based zonation scheme for assisting with identification of the predicted extent, severity and duration of impacts associated with dredging (Table 5-2, Figure 5-1). This classification also provides a good basis for spatially-defining impacts associated with seabed mining as they relate to direct physical disturbance and indirect impacts such as turbidity and sedimentation.

Table 5-2 : WA EPA zonation scheme of impacts associated with dredging (from EPA 2011).

Zone	Description
Zone of High Impact	<p>The area in which impacts to benthic organisms are predicted to be irreversible. The term irreversible is defined in accordance with EPA (2009) as <i>'lacking a capacity to return or recover to a state resembling that prior to being impacted within a timeframe of five years or less'</i>. Areas within and immediately adjacent to proposed dredge and disposal sites are typically within zones of high impact.</p>
Zone of Moderate Impact	<p>The area in which predicted impacts on benthic organisms are sub-lethal, and/or the impacts are recoverable within a period of five years following completion of the dredging activities. This zone abuts, and lies immediately outside of, the zone of high impact. Proponents should clearly explain what would be protected and would be impacted within this zone, and present an appraisal of the potential implications for ecological integrity of the impacts over the timeframe from impact to recovery (e.g. through loss of productivity, food resources, shelter).</p> <p>Where recovery from the impact predicted in this zone is likely to result in an 'alternate state' compared with that present prior to development, then this outcome should be clearly stated in environmental assessment documents, along with justification as to why the predicted impacts should be included within this zone (rather than the Zone of High Impact) and an appraisal of the potential consequences for ecological integrity. The outer boundary of this zone is coincident with the inner boundary of the next zone, the Zone of Influence</p>
Zone of Influence	<p>The area in which changes in environmental quality associated with dredge plumes are predicted and anticipated during the dredging operations, but where these changes would not result in a detectable impact on benthic biota. These areas can be large, but at any point in time the dredge plumes are likely to be restricted to a relatively small portion of the Zone of Influence.</p> <p>The outer boundary of the Zone of Influence bounds the composite of all of the predicted maximum extents of dredge plumes and represents the point beyond which dredge-generated plumes should not be discernable from background conditions at any stage during the dredging campaign.</p>

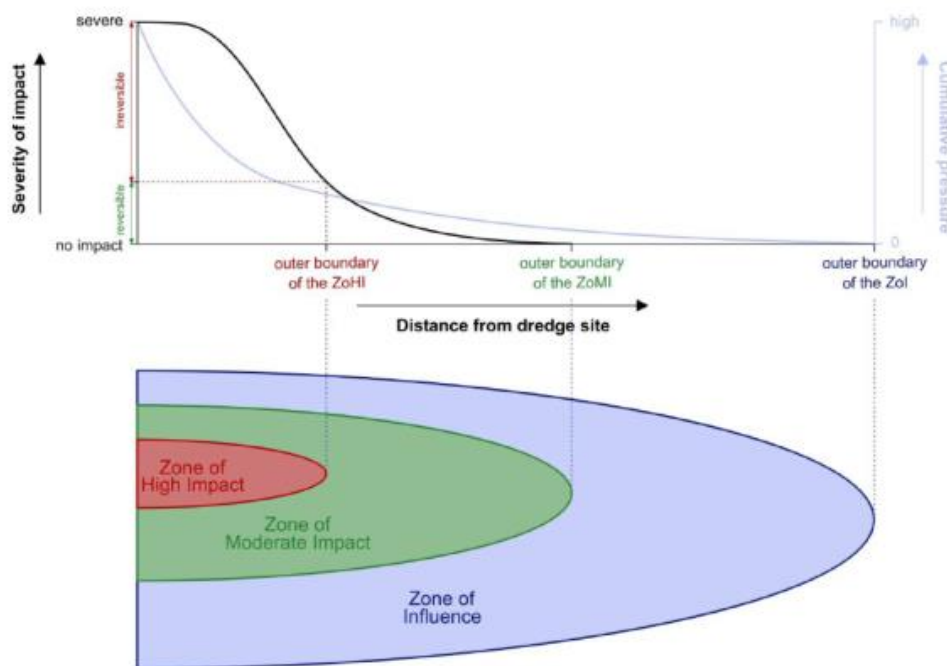


Figure 5-1 : Representation of the degree of change in environmental quality (grey line) and the level of resulting impact (black line) as the distance from the impact site increases (from EPA 2011)

5.3 Modelling to Inform Impact Assessment

A range of mathematical models are available or being developed to predict a broad array of processes related to dredging and dredge material disposal that would be equally applicable to seabed mining activities. These include models to predict:

- Changes in hydrodynamic processes (water levels, current and waves)
- Sediment release from subsea mining activity and the dynamic behavior of the sediment plumes
- Sediment transport and associated morphological impacts
- Contaminant release, and
- Response of flora and fauna to changes in water column parameters and deposition of sediments.

As stated by Bray (2008), the more sophisticated the models become, the more data is needed for setup and validation. Common to all models is the requirement for high quality input data to produce reliable and high quality results. The lack of good quality data is often a major obstacle for use of numerical models. Another important consideration is that the complexity of interactions between abiotic and biotic parameters that remains a challenge for producing realistic and acceptable predictions in the assessment process.

Quantifying and modelling the transport and fate of sediments released during dredging (and mining) operations is essential to predicting the environmental impact of large-scale coastal developments. Predicting zones of impact is an essential component of Environmental Impact Assessment (EIA) for proposed dredging operations and is a very important component in determining impacts from large scale mining projects that could occur in the coastal environment



(Sun et al 2016). As considered in Section 5.2, defining the various zones of impact is reliant on modelling to determine the extent and duration of sediment plumes that would be generated by a subsea mining activity.

Numerical modelling is a vital tool for predicting potential environment impacts of dredging activities and dredge spoil disposal. However, the processes governing dredge plume generation and transport is complex and depend on many factors, which are often site- and substrate-specific. Many different hydrodynamic and sediment transport models are used by proponents, some are open source, but many are licensed models and the formulation and parameterizations of important processes are not always known. There are currently no industry standards for calibrating the models. Usually hydrodynamic models are calibrated with field measurements of currents, water levels and waves and capable of reproducing the dynamics in the study region. However, sediment transport models are not always calibrated and ways of calibration vary greatly, partly due to differences in quality and quantity of data. Therefore, large uncertainties in modelling results often exist ((Sun et al 2016).

Irrespective of any limitations, the predictive aspect of the modelling is critical in informing the impact assessment and to assess the acceptability of the impacts prior to commencement of the activity. Ongoing modelling is then used during the mining activity to validate or adjust the model. An example of an actual 3D numerical hydrodynamic model and sediment transport model is shown in Figure 5-2.

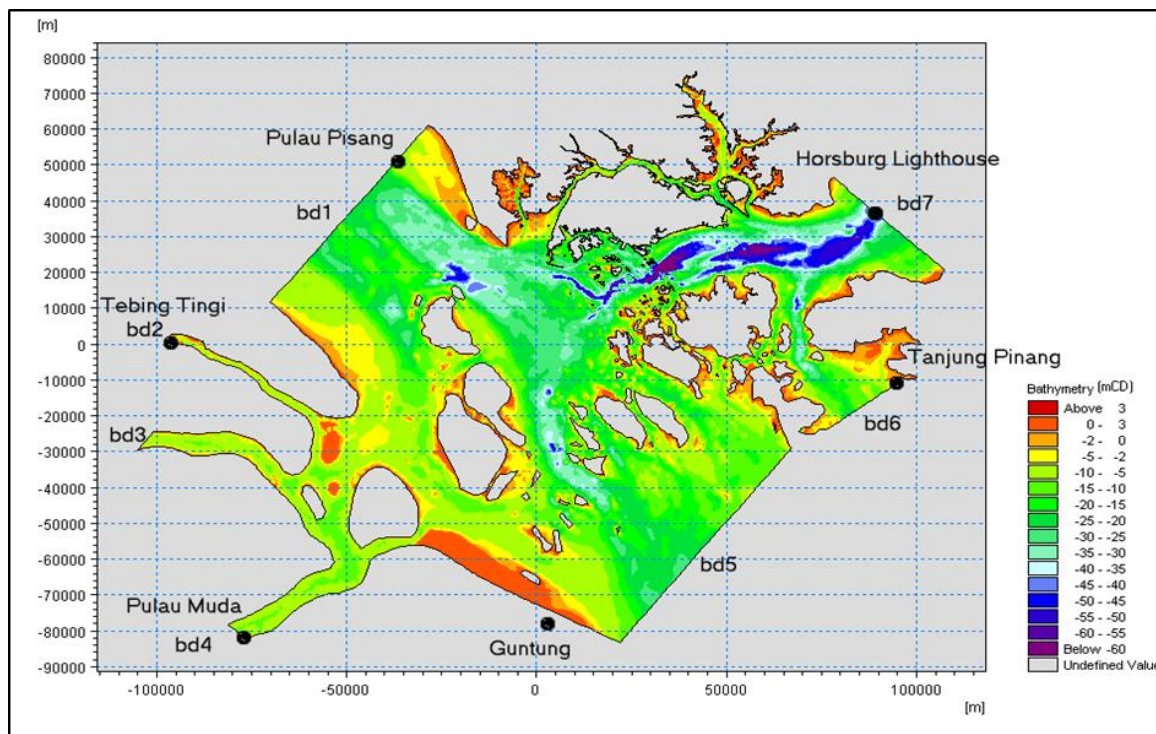


Figure 5-2 3D Hydrodynamic Model and Sediment Transport Model domain associated with bathymetry. Tidal stations used for boundary marked with solid dots along with seven open boundaries.

The calibrated and validated model is modified and refined to provide forcing for the sediment plume mixing and dispersion to simulate the activity and the potential environmental effects of the activity on the surroundings area. The model is then subject to ongoing calibration and validation.

Sediment plume modelling integrates engineering and environmental inputs and simulates the potential sediment plume concentrations and sedimentations associated with spills generated from the mining activity. The sediment plume modelling process is illustrated in Figure 5-3. The model outputs will indicate the size and distribution of each impact zone.

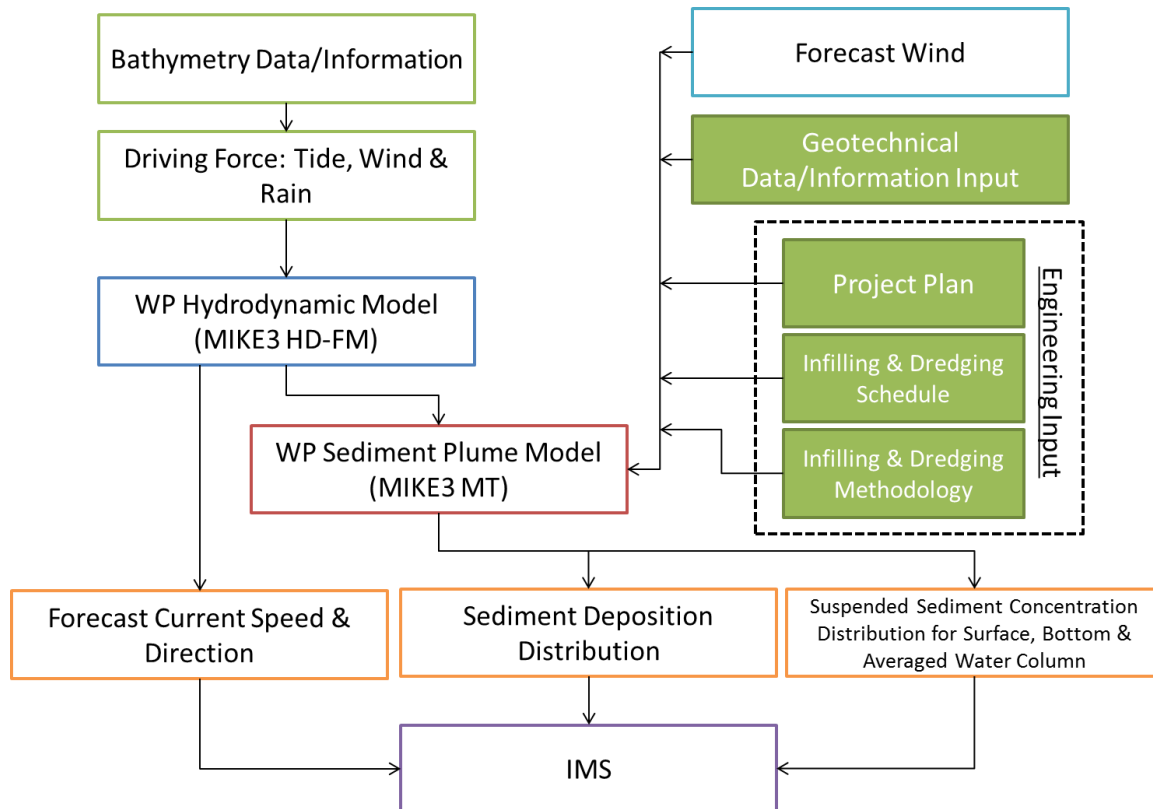


Figure 5-3 Schematic of the sediment plume modelling process

Both hydrodynamic and sediment transport model calibration and validation is then undertaken by comparing the modelled results against actual field based measurements e.g. ADCP current measurements and sediment flux measurements. The IMS refers to the Information Management System that is essentially a database for all monitored parameters.

5.4 Tolerance Limits (Thresholds)

Depending on the sensitivity and recoverability of species to environmental disturbances and associated impacts, they will have varying tolerance limits on changes to their environment before experiencing or exhibiting symptoms of stress. As discussed in Section 4, there are a range of potential impacts associated with seabed mining. The main source of impact to environmental receptors in the NT marine environment from seabed mining (outside the direct excavation footprint) is most likely related to suspended sediments (turbidity) and sedimentation.

In order to manage the impact to species and habitats, limits can be implemented of which are below the species' tolerance threshold in order to trigger management methods (i.e. cessation of dredging should turbidity exceed defined limits) or prevent known impacts to species (e.g. Jones *et al.* 2015).



The relevant stressors applicable to seabed mining in coastal waters for which tolerance limits can be defined include:

- Sedimentation
- Turbidity
- Water quality
- Light; and
- Noise.

In Australia, determination of water quality trigger levels for monitoring are often set based on the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC/ARMCANZ 2000), which outlines an approach for developing local (site specific) trigger levels based on long term (typically 12-24 months) of monitoring data from a suitable reference site. The trigger level is then determined by using the 80th percentile of the long term dataset.

Alternatively, determination of trigger levels may also be based on literature values and more recently based on site-specific natural variability (McArthur et al. 2002). This is then coupled with the use of numerical models to predict suspended sediment concentrations and sedimentation due to dredging, in order to predict the level of impact to sensitive receptors at various distances from the dredging works (DHI 2010).

The approach by McArthur et al. (2002) proposes the development of TSS or turbidity threshold limits based on naturally occurring levels at the project site and takes into account the following:

- Review of long-term (12 months) good quality TSS or turbidity measurements from the project location
- Analysis of long term data to determine seasonality in data (is highly relevant where there are distinct dry and wet seasons as there are in the NT)
- Set the 95th percentile as the threshold concentration and the 99th percentile as the highest allowable value
- Combined frequency of natural and mining related events exceeding the threshold concentration should not be significantly greater than would normally occur.

The key assumption of this approach is that the communities in the study area are adapted to the naturally occurring levels of excessive turbidity or TSS (due to storms) and as long as the project related TSS do not result in dose-duration events that significantly exceed the intensity, duration or frequency that the communities are accustomed to, then the project should not cause any additional stress (DHI 2010).

5.4.1 Tolerance Limits for Turbidity and Sedimentation

Threshold definition is particularly difficult for inshore areas, where there can be naturally high and variable background conditions of turbidity and sedimentation and benthic communities may show high tolerance to increases in turbidity and sedimentation caused by dredging (Ports Australia, 2014). The same report also reiterated that research to develop site-specific thresholds, particularly for subtropical and tropical inshore communities that often naturally experience high levels of turbidity and sedimentation can be time-consuming and expensive. The use of locally-derived tolerance thresholds is generally only feasible for major, long term projects. In this context, it is not uncommon for thresholds from similar locations to be adopted for different project monitoring programs. This has been done quite successfully in the Pilbara (Western



Australia) where there have been a number of large dredging projects undertaken over the last ten years. The tolerance limits set for these historical projects are all relatively similar in magnitude, although the allowable duration and frequency of events tends to vary from project to project (DHI 2010).

The following section focusses on tolerance limits and triggers for species that have known sensitivities to stressors that are most likely to manifest during a seabed mining project in the coastal environment.

5.4.1.1 Seagrass

Erftemeijer and Robin Lewis (2006) synthesised the literature on critical thresholds of seagrasses to light availability and sedimentation. Most species listed in their review are not known to exist in the waters of the Northern Territory. The limits for those species that are known to the region are listed in Table 5-3.

Table 5-3 : Thresholds of seagrass species to light limitation and sedimentation (from Erftemeijer and Robin Lewis 2006) and sediment burial (from Cabaço *et al.* 2008).

Species	Surface Irradiance	Duration Of Time Below Minimum Light Requirement		Sedimentation (cm/year)	Burial Level (cm)	
		Light availability	Period survived		50% mortality	100% mortality
<i>Cymodocea rotundata</i>	Unknown	Unknown		1.5	2	8
<i>Cymodocea serrulata</i>	Unknown	Unknown		13	2	-
<i>Enhalus acoroides</i>	Unknown	Unknown		Unknown	4	-
<i>Syringodium isoetifolium</i>	Unknown	Unknown		Unknown	8	-
<i>Halodule univervis</i> *	Unknown	Unknown		Unknown	4	-
<i>Halophila decipiens</i> *	2-8%	Unknown		Unknown	Unknown	
<i>Halophila ovalis</i> *	16%	0	1 month	2	2	2
<i>Thalassia hemprichii</i>	Unknown	Unknown		Unknown	4	-
<i>Zostera capricorni</i>	30	5% SI	1	Unknown	Unknown	

* refers to species present in NT waters



The use of light thresholds for monitoring and management of seagrass has been used more recently for management of turbidity from dredging campaigns. The work by Collier et al. (2016) provides a very comprehensive synthesis of thresholds that can be applied to ensure protection of seagrasses in the Great Barrier Reef World Heritage Area (GBRWHA) from activities that impact water quality and the light environment over the short-term.

An example of the turbidity thresholds applied for the protection of seagrass during the Ichthys Development Project is summarised in Table 5-4.

It is uncertain how protective these triggers actually were because monitoring of the sensitive receptors was confounded by limited baseline and no controls. A review of the effectiveness of the monitoring program is outside the scope of this report, however it does demonstrate the importance of having access to good baseline data for determining effective threshold limits.



Table 5-4 : Turbidity thresholds applied for seagrass in Darwin for the Ichthys Program (Cardno 2015a)

Components		Level 1 Trigger Daily Average Turbidity			Level 2 Trigger		Level 3 Trigger	
Previous (EA DSDMP Rev. 1)	Trigger value (Wet Season) (1 Nov to 30 Apr)	Intensity (95%ile)	Duration (90%ile)	Frequency (90%ile)	Loss in seagrass distribution (percentage cover): > level of detection AND	Loss in leaf/shoot density (leaves/m ²): >20% net detectable loss	Loss in seagrass distribution (percentage cover): > level of detection + 10% AND	Loss in in leaf/shoot density (leaves/m ²): >30% net detectable loss
	Trigger value (Dry Season) (1 May to 31 Oct)	Intensity (99%ile)	Duration (95%ile)	Frequency (95%ile)				
Revised (EA DSDMP Rev. 4 and GEP DSDMP Rev. 7)	Trigger value (Wet Season) (1 Nov to 30 Apr)	Intensity (95%ile)	Duration (90%ile)	Frequency (90%ile)	Risk to Receptor Assessment Outcome of risk assessment to inform the potential risk of impact to <i>Halophila</i> and <i>Halodule</i> resulting from the Project's dredging and / or spoil disposal activities. moderate or high risk rating = exceedance		Observed impact assessment Outcome of reactive seagrass monitoring and impact assessment to assess the impact to <i>Halophila</i> and <i>Halodule</i> resulting from the Project's dredging and / or spoil disposal activities. moderate or high observed impact rating = exceedance	
	Trigger value (Dry Season) (1 May – 31 Oct)	Intensity (99%ile)	Duration (95%ile)	Frequency (95%ile)				

5.4.1.2 Corals

Coral reefs with high coral cover and diversity occur in inshore areas where very high levels of turbidity (<100NTU) can occur over short durations as a result of wave induced re-suspension that occur during storms and cyclones (Browne *et al.* 2012). Similarly, periods of high sedimentation rates (as high as 100mg/cm²/day) may occur naturally for several days to weeks without any major negative effects to inshore corals (Benson *et al.* 1993). The durations that corals can survive high sedimentation rates range from < 24 hours for sensitive species to > 4 weeks for very tolerant species. Thresholds for sedimentation rates in individual coral species range from < 10mg/cm²/day to > 400 mg/cm²/day (Erftemeijer *et al.* 2012).

Erftemeijer *et al.* (2012) have reviewed the critical thresholds of corals to light availability (Table 5-5), suspended sediments (Table 5-6) and turbidity (Table 5-7). These tables relate to the percentage of surface light reaching the coral. Given the large number of coral species in the Northern Territory (in the order of 250 species) not all are included below. The information detailed below demonstrates how site- and species-specific critical thresholds can vary. This also demonstrates the importance of collecting suitable baseline water quality data from locations of interest as a basis for developing effective threshold limits.

Table 5-5 : Thresholds of coral species to light availability (from Erftemeijer *et al.* 2012).

Species/type	Surface Irradiance	Location
Plate corals	0.15%	Florida, USA
Star corals	1%	Curacao
Star and brain corals	20%	Florida, USA
Branching corals	60%	Florida, USA
Scleractinian corals	2-8%	South China Sea
Individual corals	10%	Worldwide
Coral reefs	35%	Worldwide

Table 5-6 : Thresholds of Australian coral reefs to Total Suspended Sediments (TSS) (mg/L) (from Erftemeijer *et al.* 2012).

Description	TSS (mg/L)	Location
Coral reef	3.3	Great Barrier Reef, QLD
Marginal reef	40	Paluma Shoals, QLD
Nearshore fringing reef	75-120	Magnetic Island, QLD
Nearshore fringing reef	100-260	Cape Tribulation, QLD

The relative sensitivity of corals according to their response level to turbidity was compiled from a review of various coral species from around the world (Table 5-7). The review by Erftemeijer *et al.*



(2012) provides a good summary of key findings relating to impacts on corals based on 77 published studies on the effects of turbidity and sedimentation on 89 coral species.

Table 5-7 : Relative sensitivity of corals according to their type of response to different levels of turbidity (mg/L) (from Erftemeijer and Robin Lewis 2006).

Response category	Turbidity level (mg/L)				
	<10	10-20	20-40	40-100	>100
No effect	Most species	Intermediate	Tolerant	Very tolerant	Very tolerant
Minor sublethal effects	Sensitive	Sensitive	Intermediate	Tolerant	Very tolerant
Major sublethal effects	Very sensitive	Sensitive	Intermediate	Tolerant	Tolerant
Lethal effects (partial mortality)	Very sensitive	Very sensitive	Sensitive	Intermediate	Tolerant
Major lethal effects (mass mortality)	Very sensitive	Very sensitive	Sensitive	Intermediate	Most species

Minimum light requirements of corals range from <1% to as much as 60% of surface irradiance. Reported tolerance limits of coral reef systems for chronic suspended-sediment concentrations range from <10mg L⁻¹ in pristine offshore reef areas to >100mg L⁻¹ in marginal nearshore reefs. Some individual coral species can tolerate short-term exposure (days) to suspended-sediment concentrations as high as 1000mg L⁻¹ while others show mortality after exposure (weeks) to concentrations as low as 30mg L⁻¹. The duration that corals can survive high turbidities ranges from several days (sensitive species) to at least 5–6 weeks (tolerant species).

Turbidity and sedimentation also reduce the recruitment, survival and settlement of coral larvae. Maximum sedimentation rates that can be tolerated by different corals range from <10mg cm⁻² d⁻¹ to >400mg cm⁻² d⁻¹. The durations that corals can survive high sedimentation rates range from <24 h for sensitive species to a few weeks (>4 weeks of high sedimentation or >14 days complete burial) for very tolerant species. Differences in sensitivity between different coral species has been related to the growth form of coral colonies as shown in Figure 5-4.

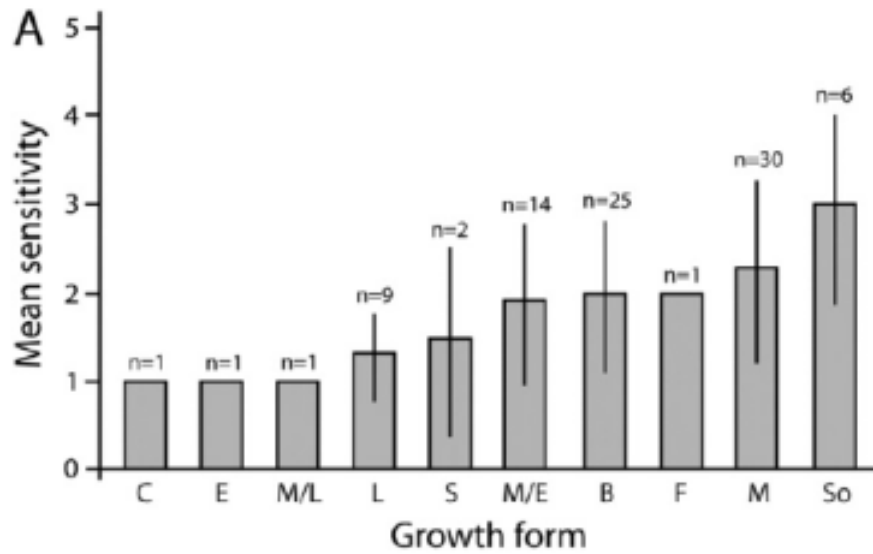


Figure 5-4 : Sensitivity of the growth form of corals according to their response effect (1 = very tolerant, 2 = tolerant, 3 = intermediate, 4 = sensitive, 5 = very sensitive). B = branching, C = columnar, E = encrusting, F = foliaceous, L = laminar, M = massive, S = solitary, So = soft corals and gorgonians. (from Erftemeijer and Robin Lewis 2006)

Examples of trigger levels adopted for coral monitoring during the Ichthys dredging program in Darwin Harbour are shown in



Table 5-8. The effectiveness of these trigger levels in protecting corals has not been assessed for the Ichthys project, however it is important to note that there has been a tendency to take an overly conservative approach when setting trigger levels for these types of projects and conversely, several instances where trigger levels have been underestimated (DHI 2010).



Table 5-8 : Coral trigger values from the Ichthys Nearshore Monitoring Program, Darwin (Cardno 2015d)

Components	Normal Situation	Level 1 Trigger			Level 2 Trigger		Level 3 Trigger
			Daily Average Turbidity				
		Intensity (95%ile)	Duration (90%ile)	Frequency (90%ile)	Coral Bleaching	Coral Mortality	Coral Mortality
Channel Island							
Trigger value (Wet Season) (1 November to 31 April)	Not triggered	>44 NTU	>26 NTU over 7 consecutive days	>26 NTU > 3 days per 7-day rolling period	>20% gross coral bleaching	Recorded mean partial mortality greater than upper 95% confidence limit of the predicted mortality	For two consecutive surveys recorded mean partial mortality greater than upper 95% confidence limit of the predicted mortality
Trigger value (Dry Season) (1 May to 31 October)	Not triggered	>21 NTU	>15 NTU over 5 consecutive days	>15 NTU > 3 days per 7-day rolling period			
Weed Reef 1 and Weed Reef 2							
Trigger value (Wet Season) (1 November to 31 April)	Not triggered	>65 NTU	>46 NTU over 6 consecutive days	>46 NTU > 3 days per 7-day rolling period	>20% gross coral bleaching	Recorded mean partial mortality greater than upper 95% confidence limit of the predicted mortality	For two consecutive surveys recorded mean partial mortality greater than upper 95% confidence limit of the predicted mortality
Trigger value (Dry Season) (1 May to 31 October)	Not triggered	>14 NTU	>11 NTU over 4 consecutive days	>11 NTU > 3 days per 7-day rolling period			

N.B. Trigger levels 1 to 3 are associated with different management responses and the extent of intervention increases with the trigger level.

5.4.1.3 Filter Feeding Species

There are many other benthic species that are vulnerable to impact from excessive turbidity and sedimentation including soft corals (gorgonians) and sponges; however these are unlikely to be as sensitive as light dependent species such as hard corals and seagrass. They are therefore not the focus of most environmental monitoring projects where most attention is placed on monitoring the health of the most sensitive species in the marine environment.

A recent review of impacts on filter feeding species (including sponges) by Schonberg (2016) concluded that sediment associated with dredging activity can affect the physiology of filter feeders in very complex ways, which are not yet adequately understood. The responses are manifold, difficult to quantify and can vary significantly between taxa, life stages, and with other environmental factors. It is thus very difficult to recognise trends and patterns from fieldwork or even from controlled experiments. At a community or ecological level it is also difficult to predict how filter feeders would respond to dredging-related pressures when considering the wide range of responses to sediments, the large variability in sensitivity between taxa and the large range of interacting environmental variables, which cannot always be separately assessed.

In the context of this uncertainty and the lack of published literature on the tolerance of sponges and other filter feeding species such as octocorals (soft corals), no tolerance limits are available.

5.4.1.4 Mangrove and Estuarine Habitat

Mangroves are less-commonly used for monitoring dredging-related impacts, as they mostly occur in areas that often experience high levels of turbidity and rates of sedimentation so are therefore comparatively tolerant to these types of conditions.

Monitoring programs do not normally include turbidity thresholds for mangroves but usually include sedimentation thresholds or trigger levels based on 50mm net sedimentation (Ellison 1998) as recently used for the Ichthys mangrove monitoring program (Cardno 2015c).

Estuarine food webs, and planktivores in particular, are inferred as being sensitive to habitat removal and burial, and having moderate to low sensitivity to turbidity and sedimentation changes, based on their biological and ecological properties. Mangroves can adapt and increase in response to low to moderate levels of sedimentation, but excess accumulation may lead to mortality. There is a need to consider the sensitivity of mangrove biota, and mangrove seedlings, as well as the trees themselves.

Assessment of risks to estuarine and mangrove habitats (from dredging related impacts) is considered low (McCook et al 2015).

5.4.1.5 Summary of Sensitivities

A generalised summary of which invertebrate characteristics may be vulnerable to dredging (and excessive sedimentation and turbidity) for various life history stages is shown in Table 5-9 below. Detailed information on the life history characteristics (LHCs) of major invertebrate taxa is shown in Appendix 6.I of Short et al. (2017).

Table 5-9 Life history characteristics to determine vulnerability to dredging for invertebrates, seagrasses and macroalgae (after Short et al. 2017)

Group	Characteristic	Vulnerability Score		
		High	Medium	Low
Invertebrates				
	Feeding strategy	Autotrophs/filter feeders	Grazers/predators	Deposit feeders
	Movement	Sessile	Weakly mobile	Mobile
	Lifespan	Short-lived		Long-lived
	Reproductive strategy	Semelparous		Iteroparous
	Reproductive season	Discrete		Protracted
	Developmental strategy	Brooders	Lecitho- /planktotrophs	Asexual
Seagrasses				
	Growth rate	Slow-growing		Fast-growing
	Time to sexual maturity	Long		Short
	Turnover time	Slow		Fast
	Seed bank presence	Absent		Present
Macroalgae				
	Growth rate	Slow-growing		Fast-growing
	Lifespan	Longer-lived (years)		Shorter-lived (days- months)
	Reproductive strategy	Less complex (fewer stages)		More complex (more stages)

5.4.2 Tolerance Limits for Toxicants

Tolerance limits and thresholds relating to toxicants in water are usually addressed by relevant water quality guidelines (ANZECC/ARMCANZ 2000). These guidelines use risk-based techniques, wherever possible, for deriving guideline trigger values for protection of complex ecosystems using single-species toxicity test data. The resultant trigger value should represent the concentration of chemical that would not cause a *significant adverse effect* on an ecosystem. Extrapolating from laboratory toxicity data to effects in the field involves uncertainties and value judgements. All



guideline values for individual chemicals are, at best, estimates of maximum concentrations unlikely to cause adverse environmental effects (ANZECC/ARMCANZ 2000).

5.4.3 Tolerance Limits for Underwater Noise

Tolerance limits for underwater noise vary according to a range of factors including the source of the noise and the sensitivity of the receptors in the marine environment. The following discussion and thresholds have been taken from the underwater piling noise guidelines (DPTI 2012) as an example of how tolerance limits have been applied for the protection of cetaceans and pinnipeds which are amongst the most sensitive of marine species to underwater noise.

The following zones of impact can be defined (Richardson et al. 1995):

- Zone of audibility – Area within which marine mammal might hear the source noise but not show any significant behavioural response. The size of the zone of audibility is highly dependent on the ambient noise environment.
- Zone of responsiveness – Area within which the considered marine mammal might react behaviourally to the noise source. This zone can be smaller than the zone of audibility as marine mammals usually do not show significant behavioural responses to noises that are faint but audible.
- Zone of hearing injury – Area closest to the noise source where the noise levels may be high enough to cause a physiological impact such as TTS or PTS.

The zones of impact define the likely environmental footprint of a noise source and indicate how far away a noise source is expected to have an impact on a marine mammal species, either behaviourally or physiologically. This information, together with information on the biological importance of the marine site as a habitat for the considered species, e.g. breeding, calving or resting areas, or confined migratory routes or feeding areas, is used to assess the likely impact of a noise source.

A summary of safety zones that are applied to impact piling in the marine environment are shown below.



Table 5-10 : Safety zones applied to impact piling in marine environments (DPTI 2012)

Species	Noise exposure threshold	Observation zone	Shut-down zone	Zone of behavioural response
Impact piling	SEL in dB(M) re 1 μPa²s for single impact			
Low-frequency cetaceans	≤ 150 dB(M_{lf}) at 100 m	1 km	100 m	≤ 150 m
	≤ 150 dB(M_{lf}) at 300 m	1.5 km	300 m	≤ 500 m
	> 150 dB(M_{lf}) at 300 m	2 km	1 km	≤ 3 km
Mid-frequency cetaceans	≤ 150 dB(M_{mf}) at 100 m	1 km	100 m	≤ 150 m
	≤ 150 dB(M_{mf}) at 300 m	1.5 km	300 m	≤ 500 m
	> 150 dB(M_{mf}) at 300 m	2 km	1 km	≤ 3 km

Species	Noise exposure threshold	Observation zone	Shut-down zone	Zone of behavioural response
High-frequency cetaceans	≤ 150 dB(M_{hf}) at 100 m	1 km	100 m	≤ 150 m
	≤ 150 dB(M_{hf}) at 300 m	1.5 km	300 m	≤ 500 m
	> 150 dB(M_{hf}) at 300 m	2 km	1 km	≤ 3 km
Pinnipeds	≤ 150 dB(M_{pw}) at 100 m	1 km	100 m	≤ 150 m
	≤ 150 dB(M_{pw}) at 300 m	1.5 km	300 m	≤ 500 m
	> 150 dB(M_{pw}) at 300 m	2 km	1 km	≤ 3 km

5.5 Resilience, Recovery and Rehabilitation

5.5.1 Recovery

Seabed mining in the coastal waters of the Northern Territory has the potential to affect a range of marine habitats directly and indirectly through flow-on effects (such as turbidity and changes in water quality) that may affect other nearby habitats that are sensitive to disturbance, such as coral reefs and seagrass beds (Newell *et al.* 1998). Therefore, the assemblages most affected by seabed mining in the region will be the benthic macroinvertebrate community and pelagic predators that rely on it. Changes in the macroinvertebrate community may have long-term implications, particularly where the physical environment is altered. For similar faunal assemblages to occur at dredged sites, the original topography, sedimentary characteristics and hydrodynamics must also be restored (Gubbay 2003; Boyd *et al.* 2004).

The ability and time taken for an ecosystem to 'recover' can differ, but it's largely reflective of the definition of 'recovery' that is used. Return of the community of an impacted site to one that is similar in species composition, density and biomass to that of pre-disturbance or a reference non-impacted site is common (Kenny and Rees 1994; Boyd and Rees 2003; deJong *et al.* 2014), however the recovery of ecosystem function could also be deemed adequate (Cooper *et al.* 2008; Barrio Froján *et al.* 2011). Presumably, should community composition and structure be restored, then so

too would the ecosystem's function, however the converse will not always be true (Michel *et al.* 2013).

Recovery following seabed mining generally supports ecological theories of succession (Figure 5-5).

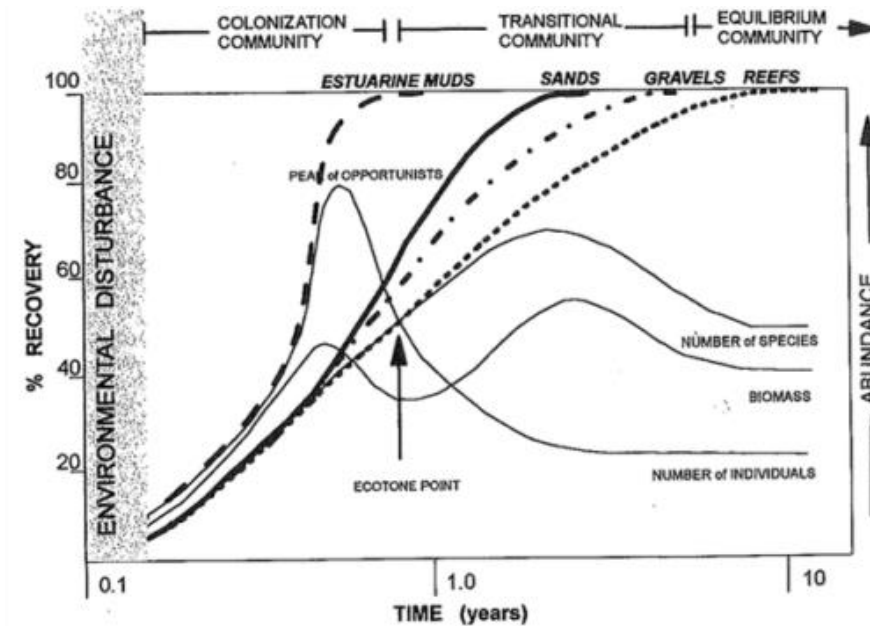


Figure 5-5 : Schematic demonstrating the likely recolonisation time of benthic macrofauna following disturbance in estuarine muds, sands and reef areas superimposed over the changes in abundance of individuals, biomass and species as the community transitions through the successional phases of recovery (from Newell *et al.* 1998)

That is, it goes through a process of recolonisation in the first instance by opportunistic, fast growing "r-strategist" species (Newell *et al.* 1996; VanDalfsen *et al.* 2000; Newell and Woodcock 2013), and in turn elevates measurements of species abundance and biomass (Desprez 2000; deJong *et al.* 2014). Overpopulation or increased predation then causes a sharp decline in abundance, owing to the reduction of the opportunists leading to the transitional community phase (Newell *et al.* 1996; Michel *et al.* 2013). It is then that metrics such as species richness returns and abundance and biomass measures stabilise, as the community again reaches equilibrium (Newell *et al.* 1996). This equilibrium community is typically characterised by more selective, larger, slower-growing and longer-lived species (Kenny and Rees 1994, 1996; Newell *et al.* 1998; Byrnes *et al.* 2004; Phua *et al.* 2004).

Seabed stability will largely determine the rate at which recovery will occur, with less stable substrates recovering more rapidly compared with stable substrates (Figure 5-6). Recovery will vary and take anywhere between less than one year (Robinson *et al.* 2004) to 2-3 years following the completion of dredging (deGroot 1979; Kenny and Rees 1994). Reef communities or deep-water sand and gravel sites tend to experience little natural disturbance and are therefore characteristic

of “K-selected” species and not adapted to or tolerant of disturbance. Subsequently, the recovery of these systems may take many years.

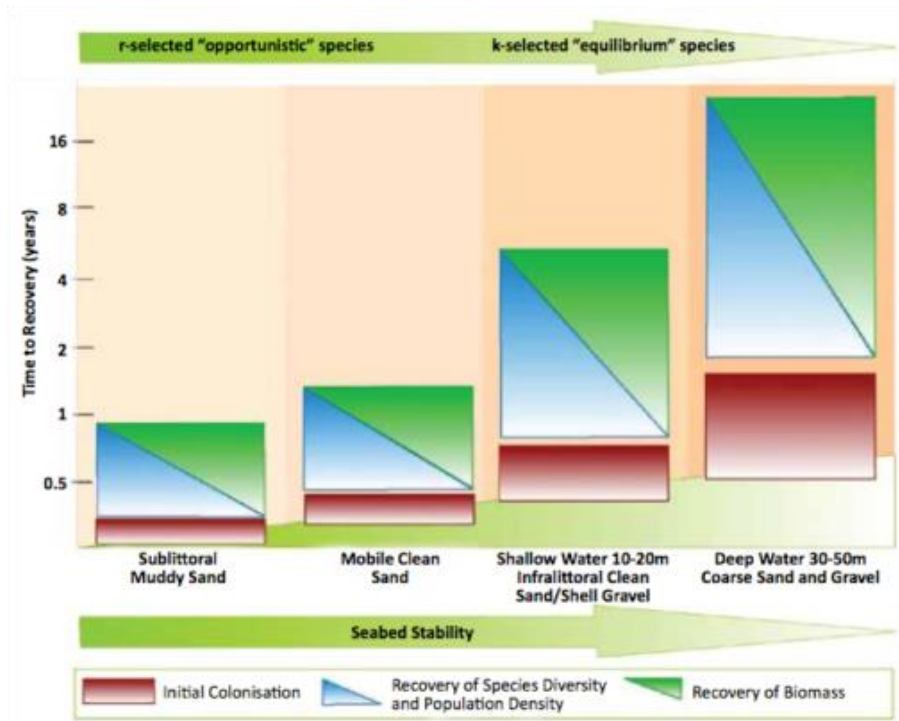


Figure 5-6 : Schematic of the general sequence in the rate of recolonisation of benthic macrofauna following dredging for various substrate types and depths (from Newell and Woodcock 2013)

The ability for an area to recover biologically from disturbance is dependent on the:

- Type and the spatial and temporal extent of the disturbance (e.g. Phua et al. 2004; Van Dalssen et al. 2000)
- Recovery and stability of physical characteristics (e.g. Desprez 2000) and
- Structure and reproductive traits of the pre-disturbed community (ie. dominance by opportunistic vs. slow-growing, long-lived species) (e.g. Robinson et al. 2005).

An understanding of the benthic invertebrates that inhabit the proposed mining area will assist in the subsequent recovery of the affected system. For example, Diaz *et al.* (2004) found that dredging before spring/summer would favour the recolonisation of crustaceans that dominated the benthic assemblages and reduce the impact on demersal fishes whereas dredging before the autumn/winter season would favour polychaetes. Given fauna can exhibit strong seasonal patterns of reproduction and recruitment, the natural recovery of a system could be completed earlier and more similar to that of the pre-disturbed community where reproductive information of dominant taxa is known.

Previous studies have indicated that assemblages in the coastal waters of the Northern Territory contain a range of sediment types, characterised by a diverse range of flora and fauna. It is



therefore expected that seabed recovery will also vary accordingly and range from months to years. This excludes recovery from any impacts outside the immediate mining footprint and presumes that the physical characteristics of the seabed are retained. If the physical characteristics are permanently altered, then full natural recovery may never occur but recovery to a new altered state will most likely occur.

5.5.2 Resilience

Resilience describes the capacity of a population or community to its original state after being disturbed. Elasticity and amplitude are two common properties or measures of the resilience of a population or community. Elasticity measures the speed of return to an original state whereas amplitude measures maximal disturbance from which a population or community can return to its original state (Minchinton 2007). Not all populations and communities will return to their original state following natural disturbance and many chronic human disturbances can result in phase shifts in populations. Direct impacts from seabed mining are likely to result in permanent changes on resident populations as it is highly unlikely that the seabed will revert to its original pre-mining condition. Indirect impacts from excess turbidity and sedimentation are less likely to result in permanent change where the impacts are within the disturbance thresholds for a particular species or population. Human disturbance can also interact with natural disturbance to influence the resistance and resilience of populations and communities.

5.5.3 Rehabilitation

Rehabilitation of the sea floor following seabed mining is very challenging, given the wide-ranging potential impacts and interrelationship between the varying effects. Furthermore, the structure of the pre-existing community needs to be well understood to determine suitable rehabilitation measures. Due to the fluidity and interconnectivity of the ocean, restoration of seabed habitats following seabed mining is a relatively new concept. Where seabed mining occurs in an area dominated by soft sediments, the physical characteristics of an affected area may be rehabilitated using the following methods (Cooper *et al.* 2013; Newell and Woodcock 2013):

- The use of unwanted dredge material derived commercially (i.e. from maintenance dredging) to use as infill for mined depressions or troughs, or use in beach replenishment
- Capping to replace the lost sediment character, i.e. gravel seeding (Cooper *et al.* 2011), or
- Bed levelling to remove artefacts in topography.

Two known studies exist assessing manipulative rehabilitation of soft sediment habitats following seabed mining. Both experimental studies have occurred in the UK due to the requirement of the nation's Marine Minerals Regulation, which requires dredging to aim to "leave the seabed in a similar physical condition to that present before dredging started" (Cooper *et al.* 2011). Collins and Mallinson (2006) used waste shell material from shellfish processing to promote faunal recolonisation at two dredged sites to the east of the Isle of Wight, UK. In 18 m of water, approximately 200kg of crushed whelk and scallop flat were deposited. The crushed whelk was found to be too mobile for successful recolonisation, however the scallop flat promoted quick



rehabilitation by epifauna. After only 7 months, 70% of epifauna found at other dredged locations more than five years after the cessation of dredging, had returned to the rehabilitated area.

Cooper *et al.* (2011) trialled the effectiveness and practicality of gravel seeding in order to recreate gravel habitat lost due to aggregate extraction and screening processes in the southern North Sea of the UK. The study was conducted in water depths between 22 and 33m using a commercial aggregate dredging vessel in which 5000 m³ of gravel was dredged from an active dredge zone and seeded over the treatment area via a conveyor belt at the stern of the vessel. Four surveys of the treatment area occurred; 2 months prior to seeding (t-2), immediately post seeding (t0), and 12 and 22 months following seeding (t+12 and t+22 respectively). Acoustic, video and grab surveys showed a 22% increase in the mean gravel content of the seabed compared with the control site, though there was still more sand present compared with the external reference area. A biological enhancement was also evident, which appeared to be consistent with expected patterns of colonisation and succession, with an increase in species, individuals and biomass 12 months following seeding.

Finally, in order to enhance faunal recovery, bed levelling has also been trialled off the French coast, in an area that is characterised by dredge furrows (Pers. Comm. Dr. Michel Desprez, as cited in Cooper *et al.* 2013). It is not yet clear whether these techniques however are practical on an industrial scale or whether the benefits outweigh the costs.

Arguably, the effectiveness of such rehabilitation measures may be limited in a holistic sense; for example where the acquisition of gravel through gravel mining, or sand for beach replenishment, results in the transfer of impacts from the remediation site to the borrow site (Newell and Woodcock 2013). A cost-benefit analysis was conducted by Cooper *et al.* (2013) for the rehabilitation of physical characteristics of the outer areas of the Thames Estuary in the UK following aggregate dredging. Water depths in the extraction site are between 27-35 m and extraction had occurred for over 25 years by both trailer suction and anchor dredging techniques. In that assessment the cost benefit of rehabilitation could not be justified.

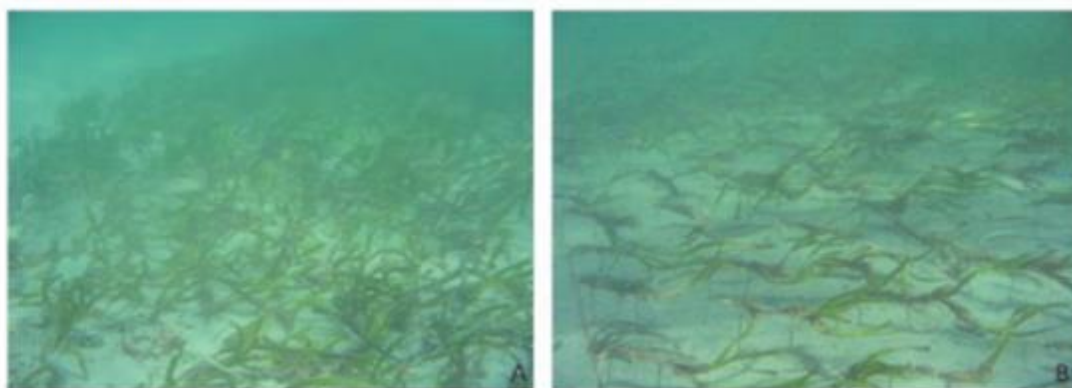
Cooper *et al.* (2011) suggest before restoration of the seabed, a site-specific feasibility assessment should be conducted, establishing the:

- thickness and extent of the overburden resulting from dredging
- potential for natural recovery
- significance of the changes for the health of the wider ecosystem and other legitimate interests
- quantity of material required for restoration, compared with the quantity of material extracted over the history of the site
- source and nature of the gravel material to be used for gravel seeding and any requirement for screening
- impact of the screened sediments on restoration efforts
- likelihood of long-term success, accounting for local conditions, and
- the financial and environmental costs and benefits of restoration.



Seagrass rehabilitation in Australia has occurred since the 1990's and mostly in temperate parts of the country such as Albany, Cockburn Sound and Botany Bay (Butler and Jernakoff 1999; Oceanica Consulting Pty Ltd 2011). Rehabilitation of seagrass involves transplantation, and a manual exists for the rehabilitation of *Posidonia* specific to temperate Western Australia, likely resulting from the extent of transplantation that has occurred in the region (BMT Oceanica 2013). Transplantation of seagrass seedlings has been shown to be effective under appropriate conditions (eg. Figure 5-7, Oceanica Consulting Pty Ltd 2011).

January 2007



July 2010

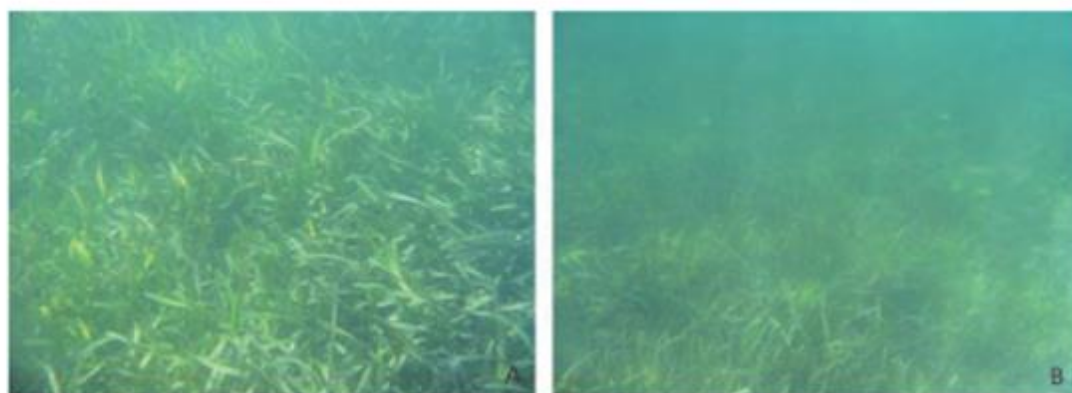


Figure 5-7 : Density experiments with *Posidonia australis* transplants following planting (top: January 2007) and monitoring (bottom: July 2010) in Cockburn Sound, Western Australia (from Oceanica Consulting Pty Ltd 2011)

No data on transplantation of seagrasses occurring in the Northern Territory could be found. While there have been documented losses due to dredging e.g. Port of Bing Bong, regeneration has been based on natural recovery not active transplantation. Relocation/transplantation of coral reefs have also been documented (e.g. Kenny *et al.* 2012 and Naughton and Jokiel 2001) and given the sensitivity of both seagrasses and coral reefs to disturbance the following factors should be considered prior to restoration attempts (Butler and Jernakoff 1999):

- selecting suitable sites, e.g. suitable sea floor, protected location, adequate light, good water quality and clarity
- developing site-specific methodology appropriate to conditions
- improving seagrass spreading and coverage rates
- minimising donor bed damage
- overcoming high labour and time costs, and
- attracting the desirable functional attributes.

While both seagrass and coral transplantation and relocation is technically feasible, it is not always successful and avoidance of direct impact to these habitats should always be the preferred option.

5.6 Scientific Uncertainty

Scientific uncertainty occurs where existing science is incomplete or where no consensus exists regarding a particular threat (Hunter et al. 2002). Where there is a threat of serious or irreversible environmental damage and scientific uncertainty as to the environmental damage, there is a need to apply the precautionary principle. The precautionary principle is usually applied when one of these two conditions occur. Either condition can be usually addressed through further study, modification to the proposed activity, and/or adoption of control measures. With respect to seabed mining in NT waters, the threats and exposure pathways associated with mining are generally well understood, whereas the baseline scientific knowledge of much of the marine and coastal environments of the Northern Territory remain incomplete and poorly described outside selected areas such as Darwin.

A similar judgement was also reached during the assessment of TTR's marine consent to recover and export iron sands from the South Taranaki Bight in New Zealand (refer to <https://www.epa.govt.nz/assets/Uploads/Documents/Marine-Activities-EEZ/Activities/TTRL-Marine-Consent-Decision-EEZ000011-FINAL-version.pdf>). Although only recently approved (August 2017), the original application was rejected due to the considerable uncertainty as to the nature of the marine environment that might be affected by the proposal and the environmental performance standards necessary to ensure that significant adverse effects are avoided, remedied or mitigated. This included the application area but also the area of the sediment plume's potential impact. The NZ EPA indicated at the time that:

"It was generally agreed by the experts, and accepted by the applicant, that natural temporal variability in the relevant marine environment is not well understood, despite the fact that the applicant has been working on this proposal for over seven years. It was generally agreed that this 'gap' would need to be filled before appropriate trigger values and compliance limits relating to the effects of the applicant's proposed mining operations could be set. The generally accepted view was that this baseline environmental monitoring would take at least two years."

5.7 Application of Best Practice

There are currently no recognised international best practice guidelines for minimising or mitigating environmental impacts from seabed mining.

The definition of what constitutes 'best practice' does vary according to the domain for which it is developed. In general terms, best practice may be a technique, method, process or activity that is believed to be more effective at delivering a particular outcome than any other technique, method, process or activity, when applied to a particular condition or circumstance. Best practices may also be defined as the most efficient (least amount of effort) and effective (best results) way of accomplishing a task. Additionally, a given best practice may only be applicable to a given circumstance or condition and may have to be modified or adapted to achieve best practice under different circumstances or conditions. It is also logical to assume that best practice will change over time with development of improvements.

With respect to dredging which could equally apply to seabed mining, the Hartman Consulting Group (1996) defined "Best Management Practices" as the actual practices – including the forms, procedures, charts, software references etc – actually used by dredgers to minimise consequences of dredging and disposal on water quality. Water quality is but one of several potential perturbations associated with dredging and spoil disposal and the Hartman definition, whilst limited, could therefore, be considered to represent one of many possible best practices that could be applied to seabed mining.

This broader concept of multiple objectives was recognized by the World Association for Waterborne Transport Infrastructure (PIANC 2009) in the distinction between dredging management practices, as follows:

- **Management Practice (MP):** A practice intended to improve the environmental performance of a dredging project, inclusive of excavation, transport, and placement of dredged material.
- **Best Management Practice (BMP):** A management practice, or combination of management practices that is determined after impact assessment, examination of alternative practices, and appropriate stakeholder participation to be the most effective, practical, and sustainable means (including technological, economic, social, and institutional considerations) of achieving an environmental performance objective.

With respect to achieving "environmental performance objectives", in the context of dredging projects such would typically refer to a means of preventing, or reducing, the potential environmental impacts associated with dredging related operations. The PIANC (2009) BMP definition recognises that many factors influence the selection of appropriate MPs for improving outcomes of dredging activities. The perceptions and relative importance of potential impacts can vary amongst different stakeholders (e.g. project owner or sponsor, resource agency with regulatory authority, conservation organisations and the public), however an understanding of the physical changes that result from dredging and the likely impacts these changes might have on the environment are prerequisites to the selection of effective MPs. Whilst BMPs need to be determined on a project-by-project basis, ultimately the economic factors and environmental



consequences driving the project should optimise the various costs and benefits of project implementation and subsequent derivation of the preferred BMPs.

Most marine dredging projects (like mining projects) are likely to involve the dislodgement of seabed material and its subsequent recovery, transfer and disposal. Dredging projects (and therefore mining projects) are likely to vary with respect to:

- nature of the project
- jurisdiction (national and international regulatory environment)
- local marine communities and sensitive species
- prevailing physical features and hydrodynamic characteristics
- social expectations and
- any funding constraints or lack thereof.

In addition to the above, factors that should be considered in scoping a seabed mining project include:

- access to and cost of mining equipment
- the availability of and/or ability to develop information on the physical, ecological and socio-economic characteristics of the areas to be affected
- the design and implementation of mitigation opportunities, and
- the development of environmental monitoring and auditing programs.

Combined with each of these are issues related to scale of the project and the costs of approvals, mining and monitoring. As a consequence of the above, there are no universally accepted guidelines that could be directly applied to all mining projects. Each project is unique in the sense that the principal project drivers will vary according to local conditions (physical, ecological and socio-political) and cost relative to overall project budget.

In spite of this, a series of generic 'best practice' measures to reduce the environmental and potential health, safety and community impacts from mining activities can potentially be applied to all seabed mining projects.

In essence, every dredging and reclamation project consists of two inter-related elements; the dredging process and the constructed project. The dredging process is generally short-term (months to years) with multiple short-term effects, some of which however may persist over the longer term. (It should be noted however that for a mining project, the extraction process is likely to be over a much longer period years to tens of years). The constructed project will have long-term environmental effects, some positive and others potentially negative. Ultimately, the proponent of a seabed mining project should aspire to having no "unacceptable" environmental effects.

6 Mitigation Measures

The choice of the most appropriate mitigation measures depends largely on the actual conditions at the mining or disposal site. Some of the measures will be ineffective or not possible to implement depending on a range of factors. A thorough environmental impact analysis of the planned operations would be required to determine the correct infrastructure and procedures for an optimal project both in terms of economics and protection of the environment.

6.1 Innovations in Plant and Equipment

As part of considering appropriate elimination, substitution and engineering controls, any of the following mitigation measures may be applied to good effect on a seabed mining project where conventional dredging plant and equipment are utilised:

- Specially designed cutterheads to modify spillage and excessive suspended sediments at the dredge site
- Specially designed dragheads to improve suction efficiency
- Green valves to reduce turbidity when using overflow on a TSHD
- Degassing systems (to avoid cavitation in the pump)
- Specially designed grabs and buckets (to limit loss while raising material) - closed or "environmental" buckets are designed to reduce or eliminate increased turbidity of suspended solids during retrieval of dredged material
- Use of monitoring and autonomous systems for better control of operations and undertaking precision dredging. Achieved using special tools and techniques to restrict the material dredged to that specifically identified. This may mean thin layers, either surficial or imbedded, or specific boundaries
- Silt curtain frame attached to barge
- Use of monitoring systems to alert crew of leaks
- Limit speed (revolutions and swing speed) of the cutter and ladder of the CSD respectively, to reduce the generation of suspended sediments and turbidity
- Control pump speed
- Careful navigation in shallow water to avoid turbulence
- Limit overflow
- Limit hoisting speed of grabs and backhoes to avoid spills
- Specially designed dragheads to minimise harm to turtles
- Larger capacity dredgers designed to carry larger loads. This provides for less traffic and fewer dumps, thereby providing less disturbance at a disposal site.



6.2 Environmental Windows

Environmental windows are routinely recommended by resource agencies with the intent to protect sensitive biological resources or their habitats from potentially detrimental effects of dredging and disposal operations. Environmental windows are periods of time when dredging activities may exert minimal influence on special status species.

The focus of current dredging research, planning and management is the identification of environmental time windows for dredging to minimise cumulative impacts and avoid sensitive life history phases (especially periods of mass coral spawning) or to minimise exposure (e.g. by avoiding certain tidal phases). For example, scheduling operations during periods of high ambient turbidity and sedimentation (e.g. during the wet season) may have less additional impact than operating during periods of otherwise low turbidity (dry season): that is increasing the duration of high suspended sediments may be more serious than smaller proportional increases in the magnitude of high suspended sediments. However, implementation of such strategies, and demonstration of their effectiveness, is much more difficult in practice than in principle (McCook et al. 2015).

The recent work by Short et al. (2017) examined the effects of dredging-related pressures on critical ecological processes for a range of marine organisms. The assessment included identification of potentially critical periods and locations when mitigating scheduling and processes, environmental windows (EWs) could be employed to reduce the impact of dredging on non-coral and non-fish biota, which would also equally apply to seabed mining impacts.

EWs, or the cessation of dredging during ecologically sensitive periods, can be an effective management tool if they are set properly. In addition to an understanding of environmental conditions, this requires location-specific knowledge of the timing of sensitive periods in the life histories of the key or dominant habitat forming organisms present.

The selection of effective EWs is highly dependent on the particular habitat and species present. These may be highly diverse, with correspondingly diverse life history characteristics and variable vulnerabilities to disturbance. Thus, the first step in the selection of EWs for seabed mining is to assess the ecological, social and economic 'value' of the species present in order to prioritise protection.

Marine invertebrates can play important roles in the habitats in which they occur. The filter feeders, in particular, are a highly diverse and ecologically important group, providing food and shelter for other sessile and mobile organisms. Habitat forming primary producer taxa such as seagrasses and macroalgae should also take priority for protection and management (Short et al. 2017). Seagrass meadows are highly important habitats in shallow coastal and estuarine ecosystems. They provide food, shelter and other ecological services to many ecologically and commercially important marine organisms and are amongst the most productive aquatic communities. Similarly, macroalgal beds are extremely ecologically important in most shallow temperate marine ecosystems, supporting diverse communities of fish and invertebrates.

In WA, it is known that many marine organisms exhibit an increased vulnerability to disturbance during the late spring to early autumn period (Oct. – April) due to the timing of sensitive life history periods (periods of reproduction and recruitment), such that winter is a period of the year when dredging would pose the lowest risk to critical life cycle processes for a number of taxa. However, this does not hold true for ephemeral seagrasses. Furthermore, local information on potentially critical periods and detailed knowledge of life history characteristics are missing for many dominant WA species of invertebrates, seagrasses and macroalgae.

A conceptual model of the process involved in determining EWs for dredging for a particular taxon or sub-taxon is shown in Figure 6-1. The level of accuracy of model predictions is inversely proportional to the level of generalisation of life histories within each group, directly proportional to the accuracy in predicting the magnitude of mining-related damage, and also depends on identifying feedback mechanisms between the dredge pressure and species' responses.

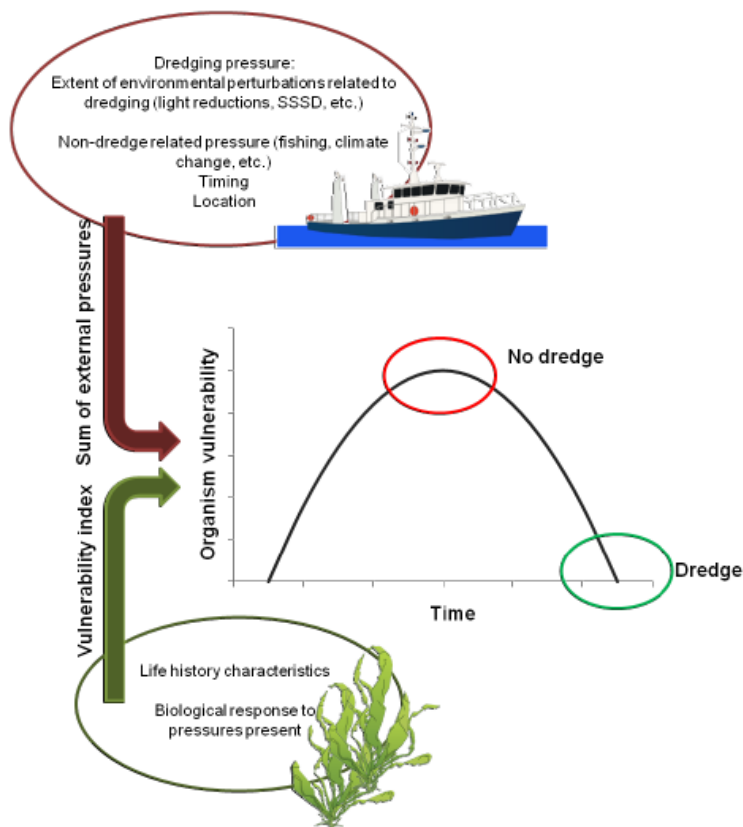


Figure 6-1 : Input requirements for environmental window modelling. Once the structure of the model is known, the life history characteristics for a particular species must be identified in order to form a vulnerability index (after Short et al 2017)

While the concept of EWs could be applied to equivalent NT marine environments, the absence of local information and species specific data (i.e. scientific uncertainty) would mean that their application to NT waters should be applied with caution.



6.3 Mitigation Measures at the Excavation Site

The most common mitigation measures implemented at the dredging site are the installation of physical barriers to prevent the spread of turbid plumes. In reality, these can be difficult to deploy in open waters that are exposed to wind and wave action. The installation of silt screens or curtains are most effective in enclosed waters or where there are only minimal currents.

6.3.1 Silt Curtains

Silt curtains are intended to allow suspended sediment at a dredging site to settle out of the water column in a controlled area, thereby minimizing the area that might be affected by the dispersal of suspended sediment from a dredging site. Localisation of spoil deposition might also be achieved, to a lesser extent, by limiting dredging to certain states of the tide.

A silt curtain is an impermeable, or permeable (to water), barrier usually constructed of flexible reinforced thermoplastic material or geofabric material with flotation material in the upper hem and ballast material in the lower hem. Silt curtains are most effective when they are not opened and closed to allow equipment access to and from the dredging or disposal area. Silt curtains are also limited to project locations with current speeds of less than 1-2 knots. Some of the key considerations and points for using a physical barrier like a silt curtain on a dredging project would include:

- Complete enclosure of the dredging plant – only possible with stationary dredgers using pipeline discharge
- Complete enclosure of the dredging zone – can be done around the dredging area of grab and backhoes while allowing barges free access alongside
- Protecting a sensitive area near the dredging area - dredge operates freely, unhindered by the barrier
- May not fully contain turbid plumes, but control dispersion by diverting them under the curtain and thereby minimising turbidity in surface layers and potential area affected by increased turbidity
- Effectiveness of screens is highly dependent on:
 - Nature of operation
 - Quantity and type of material in suspension both within and outside the screen
 - Characteristics, construction and condition of the screen
 - Area and configuration of enclosure
 - Method of mooring; and
 - Hydrodynamic conditions at site.
- In perfect conditions (no currents, waves or tide) retention of most of the mass of suspended material is possible
- With currents of greater than one knot, waves greater than 1m or a tidal range greater than 3m, retention of suspended materials will decrease considerably



- Installation and management of screen requires skill and experience
- Use of a screen may limit output level of the dredger, lengthen the execution period and increase the cost of the project;

In summation, silt curtains can be used effectively in the vicinity of sensitive areas with the appropriate equipment, however their application in NT waters where tidal currents and tidal range are relatively large, will be limited.

An alternative to a silt screen is a bubble curtain which works by preventing fine sediments from passing across the line of a perforated pipe:

- Air is continuously pumped through a perforated pipe along the seabed
- Only effective in benign sea conditions
- Use a large amount of air and result in high energy consumption.

6.3.2 Operational Controls

Operational control of dredgers can sometimes be used to good effect as an impact mitigation measure. Specific control options for mechanical, hydraulic and hopper dredgers and barges are provided below, together with the option of seasonal timing of dredging operations.

Mechanical Dredger Operational Controls

Operational controls for mechanical dredgers include:

- Increase cycle time; reduces the bucket velocity and hence the potential to wash sediment from the bucket. Reduced velocity on the way down reduces the volume that is collected
- Most sediment resuspension with a clamshell dredge occurs when the bucket hits the bottom
- Elimination of multiple bites
- Elimination of bottom stockpiling.

Hydraulic Dredger Operational Controls

Operational controls for hydraulic dredgers include:

- Reduce cutterhead rotation speed
- Reduce swing speed
- Eliminate bank undercutting.

Hopper Dredgers and Barges Operational Controls

Operational controls for hopper dredgers and barges include:

- Eliminate or reduce hopper overflow
- Lower hopper fill level
- Recirculation system.

6.4 Mitigation Measures at the Disposal (Tailings) Site

As part of considering appropriate elimination, substitution and engineering controls at the disposal site, the following mitigation measures may be applied on a seabed mining project:

- Site selection to avoid areas with high biodiversity value and/or areas which support significant fisheries
- Installation of a silt screen (around the underwater relocation site or around the outlet)
- Utilisation of underwater diffusers to reduce turbidity at the placement site
- Application of settlement ponds at the outlet of a confined placement area
- Seasonal restrictions on placement at certain locations – these are also referred to as environmental windows and apply equally to dredging as well as disposal
- Tidal restrictions for underwater placement to avoid or reduce natural transport of suspended sediments
- Use of absorbent or impermeable liners at the bottom of confined placement areas
- Use of flocculants
- *In situ* treatment of acid sulphate soils
- For contaminated sediments, design of final repositories to minimize potential for water contamination and reduce exposure risks, and
- Rapid closure and revegetation of land-based spoils repositories.



7 Monitoring Strategies

Environmental monitoring is a key activity in the control, assessment and validation of impacts that are assessed in an EIA and outlined in an Environmental Management Plan (EMP) for a specific project. Monitoring programs can be categorised into 3 types depending on their objectives (from Bray et al. 2008):

1. BACI (or surveillance) Monitoring
2. Feedback (or adaptive) Monitoring
3. Compliance Monitoring

The regulatory authority (e.g. EPA equivalent or Government department) is often responsible for the surveillance monitoring. The Project Owner (e.g. mining company or appointed Consultant) is often responsible for feedback monitoring and the Contractor (e.g. dredging company) traditionally oversees compliance monitoring. There is flexibility in the choice of environmental indicators and the appointed roles, however it is critical that a clear definition and agreement is achieved between all parties concerned during the baseline period and prior to commencement of the dredging (or mining) project.

7.1 BACI Monitoring

Before-After-Control-Impact Monitoring is monitoring which assesses temporal and spatial changes to selected parameters between a “before” condition and the current condition. This type of monitoring is the most often used and the simplest to design, particularly so in projects undertaken in the Australian marine environment. The EIA study identifies the assets or indicators most vulnerable to change and the BACI monitoring program is implemented to validate or verify the change that has been predicted or hypothesised in the EIA. The hypotheses can be of a very different nature and can relate to:

- Oceanographic conditions at the dredging or relocation site
- Environmental background conditions at the site, or
- Operational parameters related to the dredging equipment.

Validating the predictions made in the EIA is critical for ensuring that the project is executed with minimal environmental impact or within the acceptable limits of impact. Where an unexpected impact is encountered during the first phase of the project, the execution method can be modified.

For a BACI type assessment to be effective in assessing change, multiple surveys both temporally and spatially would be required. This would include multiple baseline surveys at both impact and reference sites.



7.2 Feedback (Adaptive) Monitoring

Feedback (Europe) or adaptive (USA) monitoring is a special form of surveillance monitoring where a few fast-reacting and predictable environmental variables are forecast by modelling and then monitored continuously during the dredging (or mining) activity. The purpose of the monitoring is to ensure that the possible exceedance of environmental criteria can be forecast in such good time that mining activity can be actively managed.

Feedback monitoring is usually comprehensive and more costly whereas the compliance/BACI monitoring programs are normally smaller and less costly. Feedback monitoring is normally adopted for projects with very strict environmental criteria or where legal binding limits for impact must be observed (Bray et al. 2008).

More importantly, the monitoring program should be used as a way of collecting as much direct information as possible for the project team so that environmental performance can be actively assessed and managed. This allows for modification to working procedures so that environmental performance can be improved while the project is ongoing.

7.3 Compliance Monitoring

Compliance monitoring is used to ensure compliance with contractual restrictions and obligations. A major objective in planning a monitoring program is to ensure that the dredging process is executed in accordance with various restrictions, which are legally or contractually imposed. Restrictions usually vary between projects and will depend on the prevailing human and ecological conditions at the site and can be either physical, seasonally related or quality oriented.

7.4 Monitoring Strategy

Clear environmental objectives and careful planning are important prerequisites to a monitoring strategy. This includes reflecting on the baseline information collected during the EIS/EIA/ESHIA process and the additional targeted supplementary investigations. Extraneous factors in the environment can influence some of the measured parameters during monitoring. It is therefore critical that all potential activities are considered and that adequate reference sites are established prior to dredging (or mining) so that natural changes in the environment, unrelated to dredging, can be considered and distinctions made between impacts from dredging and naturally occurring fluctuations in environmental conditions.

Water quality monitoring programs sometimes require at least 12 months of baseline data (and up to 24 months) to adequately reflect normally-occurring seasonal and interannual variation. The monitoring is usually conducted at multiple impact and reference sites. The numerical modelling developed during the EIS/ ESHIA assessment phase can be used to determine the locations of greatest potential environmental impact, which in turn, informs the process of selecting the appropriate impact and reference locations (Section 5.3).

For large or complex projects or where environmental regulations are comprehensive, feedback monitoring programs are usually recommended. Feedback monitoring ensures that possible exceedance of environmental criteria can be forecast in good time so that mining activities can be adjusted accordingly.

Based on the monitoring data, the monitoring program can be adjusted to:

- Reduce the level of monitoring (where no effect is observed);
- Continue with existing monitoring to gain further clarification of response; or
- Expand the monitoring program by increasing frequency or including additional surveys or sites.

7.5 Monitoring Variables

Careful selection of monitoring variables would be required prior to commencement of a seabed mining project. Key considerations include the environmental parameters chosen, the ability to detect change from the selected parameters and the response time required to feed back into the mining program. Typically, the monitoring programs usually involve a combination of variables. Physicochemical variables are often preferred as they are much easier to measure (compared to biological parameters) and are a useful surrogate given it is often not possible to measure all available variables, all of the time. Biological parameters such as abundance and diversity of marine species can be highly variable and it may be difficult to distinguish between natural and human disturbance. Irrespective, a monitoring program should involve a number of variables and parameters to ensure potential impact can be inferred through a number of impact pathways and lines of evidence.

The types of environmental variables chosen for monitoring may include:

- Depth
- Suspended sediment concentration (usually inferred by correlation with turbidity or more recently by use of ADCP)
- Spilled sediment accumulation
- Hydrographic parameters
- Chemical and biological parameters, and/or
- Social or economic variables.

In order to be effective for monitoring, environmental parameters should be:

- Measurable
- Relevant with respect to environment and predicted impact
- Able to provide quantifiable results
- Predictable in response to impact by using numerical tools



- Measurable response within reasonable time and cost.

Well planned and well executed monitoring programs are a standard feature of large scale and complex dredging (and therefore mining) programs. Monitoring is targeted according to the issue identified in the assessment process and is a way of verifying the predictions made during the impact assessment.

Sampling, analysis, quality control and reporting must be carried out in a professional manner and should be an important consideration in selecting sub-contractors. Monitoring of large projects can require establishment of large databases to allow for readily accessible data and the monitoring requirements are usually outlined in the Environmental Management Plan.

7.6 Environmental Monitoring Plan

The environmental monitoring plan for a project usually contains similar content to the dredge management plan but will provide more detail of what is monitored, the methods of monitoring and the reporting mechanisms. An example of the structure of an environmental monitoring plan is as follows:

- An introduction to the project, which contains the overall purpose of the monitoring program, the scope and the description of works
- A description of the existing environment, flora and fauna species and associated communities
- Roles and responsibilities, including the roles of the monitoring team, the dredge contractor and management staff and the decision-making process from project kick-off to the final review workshop
- Dredging and disposal methods and environmental management measures to be adopted
- Water quality monitoring and management
- Biological and ecological monitoring and management
- Waste and hazardous substances management
- Emergency planning and response, including performance indicators, management measures and emergency contacts
- Documentation and reporting, including data ownership and logging of data
- Training requirements of monitoring staff, dredge personnel and management, and
- References of the guidelines and documents referenced through the text.

The environmental monitoring plan specifically addresses the Terms of Reference for the project Environmental Management Plan issued by the regulator, and more broadly, requirements under other applicable legislation and regulatory approvals as well any relevant port operational policies and procedures. The performance objectives, actions, and procedures detailed in the environmental monitoring plan may be amended during the course of the project on the basis of new information and experience, subject to approval by the relevant authorities.

8 Conclusions and Recommendations

Much of the current debate around seabed mining relates to exploitation of deep sea minerals. In New Zealand for example, where there has been much ongoing debate of regulation and management, these apply to the massive sulphides in the Kermadec and Colville Ridges (located in 1000-2000m depth), phosphate nodules from Chatham Rise (in 300 to 400m water depth) and mining of iron sands and other placer deposits in the 100-150m depth range.

It is important to note that while the potential impact pathways are similar between deep sea and shallow seabed mining, there are some fundamental differences in the sensitivities of marine and benthic species that require consideration as part of an environmental assessment. Deep sea marine ecosystems for example are not well studied or understood and there is much uncertainty around the potential impacts from mining. In contrast, the potential impacts of small and large scale mining on the coastal environment are much better understood and can be better predicted due to the experience obtained from large scale dredging projects.

Nevertheless, there are many uncertainties associated with shallow seabed mining that would require to be addressed as part of any proposed mining development. As mentioned in Section 5.6, the largest and most significant uncertainty is often related to the lack of information about the marine environment itself. Without a detailed understanding of the area of interest, it is not possible to undertake an assessment of impact. This is of particular relevance to the NT marine environment as large areas of the territorial sea remain poorly studied or described.

Based on the outcomes of this review, the following guiding principles should be adopted as a starting point in assessing any future proposals involving seabed mining in the NT marine environment:

- Prohibit mining in areas containing sensitive or unique marine benthic habitats or areas that are of important spawning and feeding significance to commercially important and environmentally significant marine species;
- Complete a comprehensive characterization of the proposed seabed mining site and its resources as part of the environmental assessment;
- In the absence of adequate baseline physico-chemical and biological data, a minimum of 24 months of baseline data should be collected to inform the assessment process and to focus on areas of high scientific uncertainty;
- The optimum dimensions of the seabed mining site should be determined so that the effects on the greater marine environment can be minimised;
- The use of conceptual models can be effective in estimating the effect of massive and/or long-term mining on the surrounding marine environment including the seabed, water column and coastal habitats;
- Assess the effect of massive and/or long-term seabed mining on the ecological structure of the seabed including a thorough assessment of the spatial and temporal distribution of potential impact;



- Assess the effect of noise from mining operations on the feeding, reproduction, and migratory behaviour of marine mammals and finfish;
- Employ sediment dispersion models to characterize sediment resuspension and dispersion during mining operations. Use model outputs to design mining operations, including “at sea” processing, to limit impacts of suspended sediment and turbidity on fishery resources and minimize the area affected;
- Address the cumulative impacts of past, present, and foreseeable future development activities on aquatic habitats including assessment of impacts from other non-mining related activities that may contribute to cumulative impacts;
- Use seasonal restrictions (and environmental windows) when appropriate to avoid temporary impacts to habitat during species critical life history stages (e.g., spawning, and egg, embryo, and juvenile development);
- Although a new industry with few comparable operations, use of best management should always be applied by applying practices used by other relevant industries, e.g. dredging, where relevant;
- The use of discharge controls and environmental limits can be effective in managing and minimising adverse effects (where the key species and ecological processes are well understood);
- The use of a Technical Reference or Review Group (TRG) should also be used to oversee specific technical matters or high risk activities; and
- A project specific Environmental Monitoring and Management Plan (EMMP) will be required as part of an approval that covers all aspects of the mining activity, including the operation and the post extractive period.
- Additional species or activity specific management plans may be required as part of the EMMP subject to outcomes of the environmental assessment for the mining project.



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