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Scientific Considerations for the Assessment and Management of Mine Tailings Disposal in the Deep Sea

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Deep-sea tailings disposal (DSTD) and its shallow water counterpart, submarine tailings disposal (STD), are practiced in many areas of the world, whereby mining industries discharge processed mud- and rock-waste slurries (tailings) directly into the marine environment. Pipeline discharges and other land-based sources of marine pollution fall beyond the regulatory scope of the London Convention and the London Protocols (LC/LP). However, guidelines have been developed in Papua New Guinea (PNG) to improve tailings waste management frameworks in which mining companies can operate. DSTD can impact ocean ecosystems in addition to other sources of stress, such as from fishing, pollution, energy extraction, tourism, eutrophication, climate change and, potentially in the future, from deep-seabed mining. Environmental management of DSTD may be most effective when placed in a broader context, drawing expertise, data and lessons from multiple sectors (academia, government, society, industry, and regulators) and engaging with international deep-ocean observing programs, databases and stewardship consortia. Here, the challenges associated with DSTD are identified, along with possible solutions, based on the results of a number of robust scientific studies. Also highlighted are the key issues, trends of improved practice and techniques that could be used if considering DSTD (such as increased precaution if considering submarine canyon locations), likely cumulative impacts, and research needed to address current knowledge gaps.

Keywords: deep-sea tailings disposal (DSTD), improved practice, challenges, environmental management, stakeholders

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MINE TAILINGS DISPOSAL: AN ISSUE OF GROWING CONCERN

A rise in world population together with increased rates of economic growth in low and middle income countries over the last century has dramatically increased the demand for metals and minerals (Dold, 2014). In addition, over the span of the twentieth century, the demand for metals and minerals in high income countries has grown exponentially. For example, demand in the USA grew from a little over 160 million tons to ∼3.3 billion tons (Morse and Glover, 2000). According to the United Nations Environment Programme (UNEP), the amount of minerals, ores, fossil fuels, and biomass consumed globally per year could triple between current day and 2050 (NCIR, 2013).

Mining is defined as the acquisition of non-renewable resources from the environment, mostly accomplished by either open-pit surface mines or underground mines on land. Mining, as with other extractive industries, will always have a social and environmental impact. A major challenge for the industry is how to minimize these impacts during all stages of mining from exploration, through operations and on to mine closure. As demand for metals and minerals grows, environmental sustainability is a growing concern. The subject explored here is the disposal of mine tailings into the marine environment. Tailings are the fine-particle waste produced after extracting the desired metal from the ore. A typical copper ore is less than 1% copper and, therefore, 99% is produced as tailings (Vogt, 2012). Similarly, in gold production a typical ore contains less than 0.1% gold with 99.9% of the processed material classified as mine tailings. As high-grade ores become rarer, and technology and consumer need demands the exploitation of low-grade ores, the issue of mine waste disposal is set to increase.

Historically a relatively small number of mines have discharged tailings and mining waste into the marine environment. Originally this waste was discharged into surface waters, for example at Chañaral, Chile from 1938 to 1975 (Dold, 2014) and Jordan River Copper Mine, Canada from 1972 to 1974 (Shimmield et al., 2010). Over time, this approach has evolved to more sophisticated methods such as piped discharges, with the final depth of discharge varying from a few tens of meters to several hundred meters under the sea surface. This process is referred to as either submarine tailings disposal (STD) in shallow/intermediate waters (shallow, 0–200 m, intermediate depths 200–1,000 m) and deep-sea tailings placement (DSTP) at depths below 1000 m (GESAMP WG42). Here we advocate for a change to the current term DSTP, to a more precise terminology Deep-Sea Tailings Disposal (DSTD). The use of the term "disposal" is considered to be more accurate, as tailings are discarded at depth (where a density/turbidity current transports the tailings to a resting place on the deep seafloor) rather than the tailings being placed in a specific contained area at the outfall or pipe mouth.

Shallow STD has been utilized in a number of coastal mining sites around the world to date. However, whilst most early operations were poorly regulated, and tailings were discharged into the sea as a matter of convenience, modern STD operations now commonly follow an environmental impact assessment (EIA), with baseline studies undertaken to improve understanding of risks. However, shallow STD often leads to severe environmental damage and adverse effects on local biota (Castilla and Nealler, 1978; Loring and Asmund, 1989).

Since the early 1970s, detailed engineering has been incorporated in the practice of most marine tailings disposal operations. At the same time, both the tailings outfall and the final tailings deposition area have been progressively designed to be located deeper (below sea level) in an attempt to minimize environmental impacts (Shimmield et al., 2010; Ramirez-Llodra et al., 2015), the process termed DSTD. The general features of DSTD have been outlined by Apte and Kwong (2004), Shimmield et al. (2010), Reichelt-Brushett (2012), Vogt (2012), Ramirez-Llodra et al. (2015), and Morello et al. (2016).

The concept of DSTD is based on discharge at the edge (usually 100–300 m depth) of an extended drop-off, to a final deposition depth of 1,000 m or more, and at a depth below the euphotic surface mixing zone (Ellis and Ellis, 1994). To achieve deeper deposition, the discharge must be at a location where the tailings slurry from the pipeline will form a density/turbidity current flowing coherently with minimal plume dispersal until it reaches the base of the drop-off. Frequently submarine canyons have been considered suitable by DSTD practitioners, including those beyond fringing reefs in tropical sites (Reichelt-Brushett, 2012). DSTD systems should be designed to prevent tailings reaching surface waters and approval of the EIA will be dependent on the DSTD design achieving this constraint. Any subsurface plumes or upwelling of tailings back into shallow waters will expose coastal environments to increased physical (e.g., suspended solids) or chemical stressors (e.g., metals/metalloids), where toxic components may enter the food chain and have detrimental effects on a wide range of marine organisms. It is important that tailings do not enter the mixed layer as coral species in tropical regions have been found to be particularly sensitive to increased concentrations of suspended solids (Flores et al., 2012; Jones et al., 2015).

In 2015, only 16 of the current 2,500 large industrialized mines worldwide utilize STD/DSTD (GESAMP, 2016). These are restricted to a few countries, namely Norway, Papua New Guinea (PNG), Philippines, Indonesia, France, Turkey, and Chile (Dold, 2014). However, an additional 15–20 mines are already considering STD/DSTD as a future disposal option (GESAMP, 2016). **Figure 1** displays DSTD operations (PNG, Greece, Indonesia, France and Turkey), both active and closed, reporting the average annual tailing disposal in million tons. A number of case studies from DSTD operations can be found in the Supplementary Material.

This paper builds on previous reviews and case studies to consider (1) tailings disposal options (2) impacts and recovery potential from tailings disposal in the deep sea, and (3) the role of scientific information in analysis of case studies. With a focus on DSTD the following aspects are discussed; societal and environmental impacts, approaches to reducing environmental impact, future prospects, legal controls and constraints, science gaps and DSTD relevance to broader environmental management in the deep ocean.

ASSESSING MINE TAILINGS DISPOSAL OPTIONS

Mine tailings management and/or disposal options are important considerations for government regulators during the EIA approval process associated with new mining activities. Each site will have a different set of constraints, which will influence the decision on the tailings disposal options. The main mine

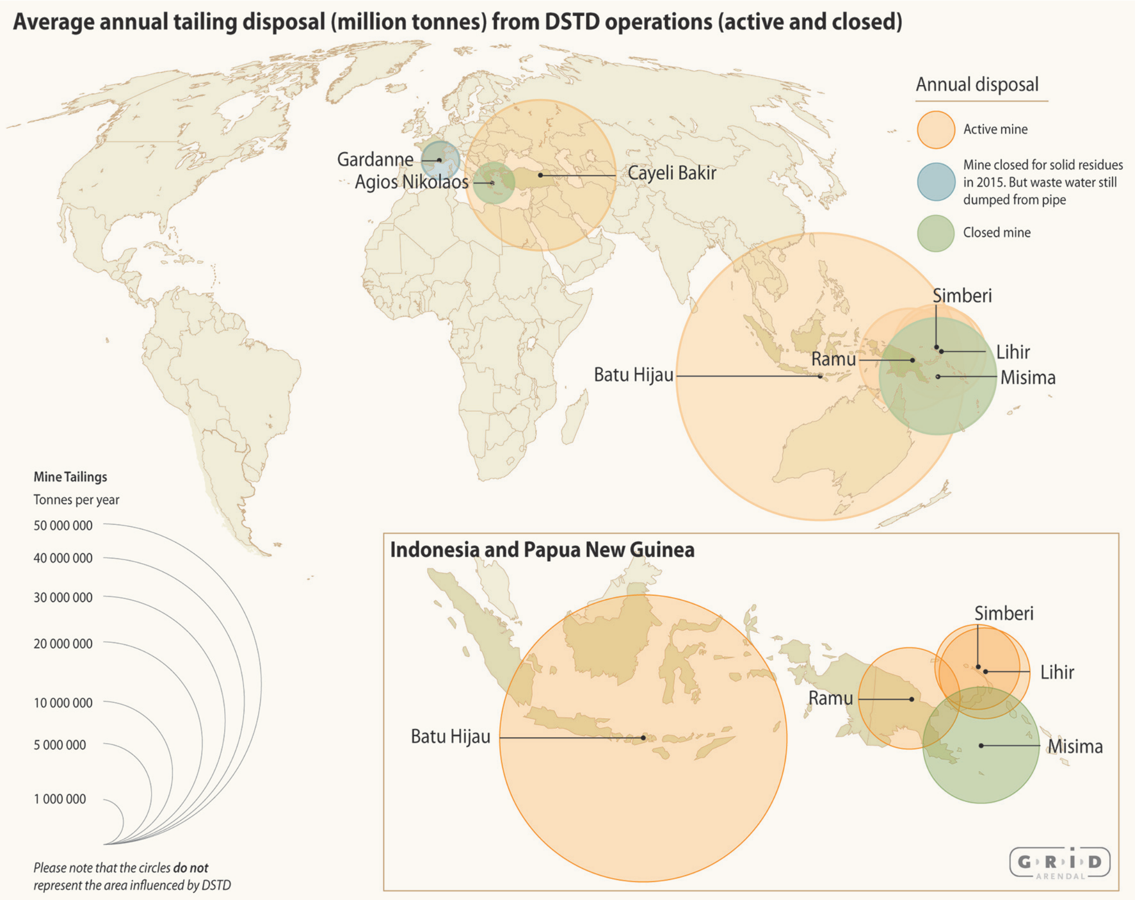

FIGURE 1 | Average annual tailing disposal (million tons) from DSTD operations (active and closed).

tailings disposal options include: tailing storage facilities (TSFs) in the form of dams or ponds (holding wet tailings or partially dewatered pastes) and underground tailings disposal on land, riverine tailings disposal and marine tailings disposal (STD/DSTD); however, some of these are not permitted in some countries. The disposal of mine tailings both on-land or in the marine environment presents unique challenges. The choice of disposal option is dependent on the physical and chemical nature of the tailings, the mine topography, climatic conditions, along with socio-economic considerations; each of the different disposal options has advantages and disadvantages.

In a large number of cases land-based storage in terrestrial impoundments or tailings dams is the norm for the constraint of mine tailings. At least 3,500 mine tailings dams/impoundments exist worldwide, but they are not without environment and public safety issues. The main issues include: the size of the footprint and loss of land that could be used for other activities, potential contamination of surface waters and groundwater, and the short- and long- term safety and integrity of the engineered facilities. There have been 138 significant recorded failures of mine tailings storage dams (Vogt, 2012).

On land, management options frequently include a combination of disposal and storage techniques and include reuse and backfilling as priorities. Geochemical developments that aim to change the character of tailings are making substantial headway in the industry, and processes to better manage acid generation and development of tailings pastes (Palkovits, 2007) are proving potential risk reduction tools. Further to this, appropriate storage of mine tailings may enable reprocessing of old tailings wastes when new technologies become available to further extract the target metal or when minerals other than those originally sought become valuable. Hence many mine tailings may have a series of potential uses immediately or in the future, but only if they are stored in a manner that enables reprocessing.

The success of land disposal is often dependent on climatic conditions and seismic activity. In areas of frequent tectonic activity and high rainfall, there is an increased risk of dam failure and concerns over contaminated mine water that may influence the water quality in local waters bodies. In some countries such as Indonesia and PNG, on-land storage facilities are considered difficult and potentially unstable due to the mountainous terrain, the high risk of earthquake events and rainfall up to 3 m per year. Combined with social demands on the customary lands, this has led to mines in these countries choosing marine disposal. Similarly, in Norway, suitable land for disposal of mine tailings near the fjords is not readily available. However, it is also important to realize that underwater earthquakes, tsunamis, currents and upwelling are also risk factors associated with STD/DSTD in many countries.

The tailings disposal option is considered one of the most important decisions during the mining feasibility assessment, each disposal method having environmental implications that will require management. Therefore, mining waste should be managed with respect to what is an acceptable environmental impact which may be determined by the various processes involved with EIA and include community engagement in the process, as well as considering the technical and economic feasibility of the disposal method. When considering disposal options, historically, short-term profit was of higher interest than long-term solutions to tailings containment and management (e.g., Palkovits, 2007). Economic, environmental and social considerations (Vanclay, 2004) are now part of the reporting process for many mining operations. However, in countries where environmental legislative frameworks and enforcement are poor, considerations of longer-term environmental and social impacts/threats may not be fully recognized and their level of importance may be lower in the decision-making process.

Regulators currently rely on information provided by EIAs, which undergo an independent review process, to inform decisions and, like many industries, it is recognized that there is a lack of information on potential long-term impacts of DSTD. Work to fill knowledge gaps is often limited by finances, along with sampling capabilities on temporal and spatial scales, and a lack of background understanding of ecosystems likely to be impacted. In this context the understanding of the consequences of DSTD on ecosystems and ecosystem processes is currently limited. Robust scientific knowledge is even scarcer for bathyal and abyssal ecosystems, including sedimentary slopes, submarine canyons, seamounts, or habitat-building fauna such as cold-water corals that may be impacted by DSTD (Ramirez-Llodra et al., 2011). In many cases, the physical, geochemical and biological information available from areas considered for DSTDs is lacking, and knowledge is based only on baseline studies commissioned in the framework of the mining activity. The main limitation to such studies is the remoteness of deep-sea ecosystems, the study of which comes with high financial and technological costs; hence, conducting the necessary research for a robust EIA and monitoring of DSTDs and their impacts, before, during and post-operation is challenging.

There has been some development of numerical assessment frameworks for selecting the most suitable tailings waste management options (e.g., Kizil and Muller, 2011). Some frameworks enable details on other options such as TSF in assessments that can potentially feed into a more complex options analysis that considers environmental and social values.

Comparisons of disposal options require a supporting framework that acknowledges/assesses constraints, impacts, and risks across terrestrial and oceanic boundaries and which go some way to applying ecosystem values to these systems. Here, an outline framework for considering disposal options has been developed, highlighting the complexity that exists at the site level and the need to incorporate social and environmental linkages (**Figure 2**). Such a framework may help resolve conflicting views on how the receiving environment is valued. It has been developed as a decision tree approach to risk assessment (e.g., ANZECC/ARMCANZ, 2000; Reichelt-Brushett, 2012), and incorporates social values that are informed by stakeholder engagement activities identifying risk assessment tools and approaches for deep-sea mining and STD/DSTD (e.g., Reichelt-Brushett et al., 2016). Consider for example, what may be deemed "acceptable" in terms of the movement and distribution of tailings in a terrestrial setting as compared to an oceanic setting.

There is increased pressure and a growing trend in leading mining companies to improve the sustainability of tailings disposal methods (ESMAP/World Bank/ICMM., 2005). The concept of mine tailings "management" is a longer-term construct that may extend for several decades after mine closure. Such a process is becoming more common practice at mine sites throughout the world and some countries require substantial bonds to be paid to ensure that longer-term management processes can be afforded in what is a relatively volatile industry. This approach is partially in response to abandoned mine sites and processing operations that have resulted in serious health risks to communities and are requiring multimillion dollar clean-up operations with complex liabilities (e.g., Barth and McNichols, 1994; Hanrahan et al., 2016).

DSTD IMPACTS AND ECOSYSTEM RECOVERY POTENTIAL

Historically, the deep seabed has been regarded as an almost unlimited repository for waste (liquid and solid) (Ramirez-Llodra et al., 2011). This misconception has changed in the latter half of the twentieth century, in part as a function of stakeholder scrutiny of the increasing interest in deep-sea living and non-living resources (Mengerink et al., 2014). Increasingly, the deep seabed is being targeted as a provider of mineral resources. Both the extraction of minerals from the seabed and disposal of mining waste products into the deep sea from either land-based or seabed mining are of growing concern due to the disturbances that can be caused to both littoral and deep-sea benthic ecosystems (e.g., Lee and Correa, 2005; Hughes et al., 2015; Levin et al., 2016).

The waste materials from DSTD consist of a slurry of predominantly finely-crushed rock materials, formed after the mineralized material has been processed. This material contains mud, silt and sand, water, low concentrations of targeted minerals (e.g., gold, copper, and silver), and measureable concentrations of other metals such as arsenic, cobalt, nickel, mercury, lead, zinc, as well as processing wastes such as sodium cyanide, lime and other acids (Ramirez-Llodra et al., 2015). DSTD EIAs require rigorous studies of hydrology and local topography to determine feasibility of site selection, together with comprehensive studies aimed to create baselines of abiotic (environmental) and biotic (ecology and function) properties of the actual seabed and overlying ecosystems in order to assess potential environmental impacts. However, although these discharged tailings are expected to be permanently deposited in a deep-water environment, the potential for plume dispersal and tailing resuspension, and the

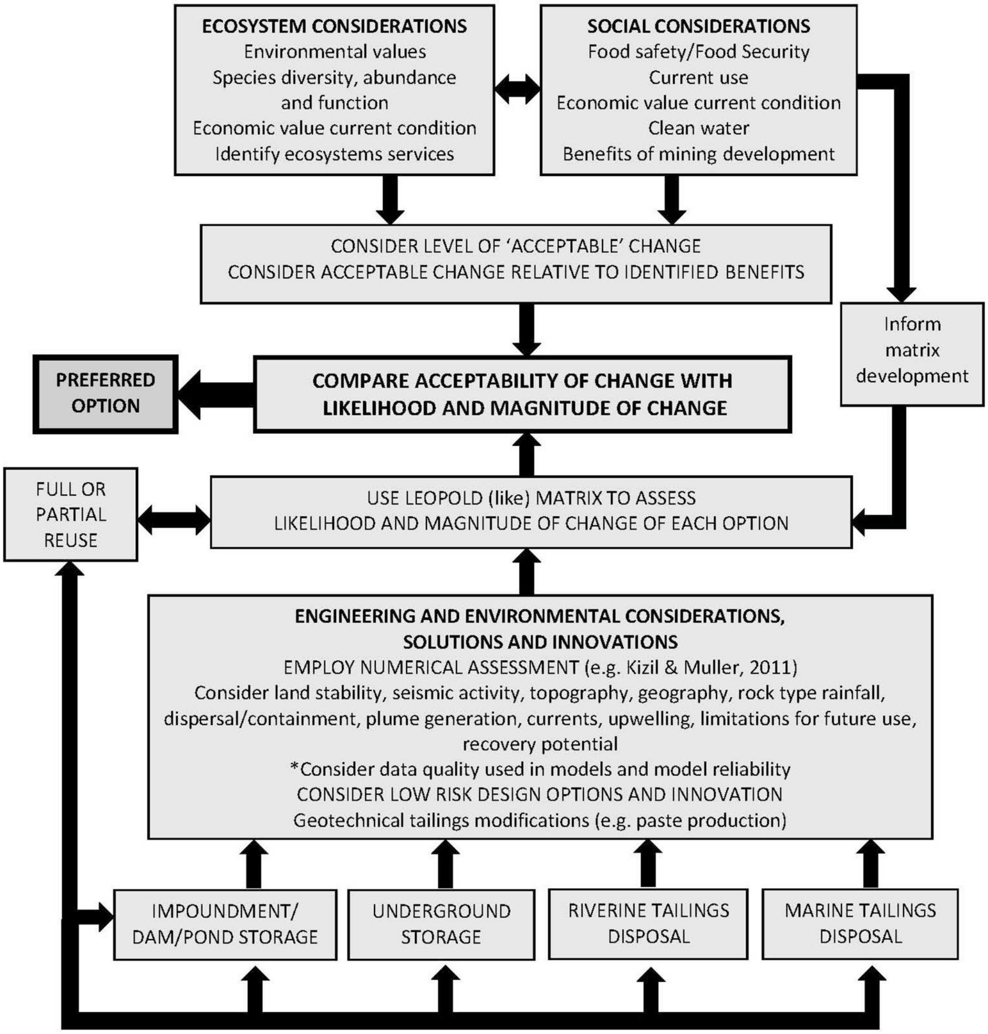

FIGURE 2 | Possible framework for considering options for tailings disposal.

consequences for the marine ecosystem in the water column and on the seafloor are uncertain over long time scales.

The post-depositional fate and behavior of tailings disposed in the deep sea will have major effects on the sedimentary and geochemical environment, and on the integrity and recovery of faunal communities. Submerged tailings constitute both a potential source of remobilized dissolved metals and metalloids, as well as processing chemicals (e.g., acids, flocculants and floatation agents) to pore water and overlying water (Perner et al., 2010; Shimmield et al., 2010; Angel et al., 2013; Simpson and Spadaro, 2016) by bacterially-mediated, sediment diagenetic processes associated with the remineralization of organic matter (Middelburg and Levin, 2009; Bourgeois et al., 2017). For instance, increased copper concentrations can affect microbial biomass and metabolic activity leading to reduction of their assimilative capacity, and hence impaired crucial ecosystem services such as carbon and nutrient cycling (Jonas, 1989; Almeida et al., 2007). Under high rates of sedimentation associated with tailings disposal, crucial reactions involved in the oxidation of organic matter may be altered, disrupting biogeochemical zonation of electron acceptor sequencing. For instance, O_2 and NO_3 could be depleted more rapidly than they can be restored (Pedersen, 1984); this would be exacerbated if bioturbating fauna were reduced. In shallow waters, limed tailings have been observed to diminish benthic phosphorous regeneration relative to natural sediments, with potential depletion of productivity (Pedersen and Losher, 1988).

The degree of physical and possible toxic impact of a mine tailings discharge will depend on several abiotic and biotic factors. These factors include the location of the outfall pipe, the volume disposed and the physical, chemical and hydrodynamic conditions of the targeted area, together with the degree of tolerance/sensitivity of the organisms in the local site and adjacent areas that may be affected by increased water turbidity and concentrations of metals or metalloids due to tailing plumes (Mineral Policy Institute, 1999; McKinnon, 2002). Most of the known effects of tailings on marine environments that are described in the literature have been based on tailings disposal in littoral, nearshore shallow waters and coastal fjords (e.g., Kathman et al., 1983; Burd, 2002; Lee and Correa, 2005; Kvassnes and Iversen, 2013; Mevenkamp et al., 2017). Studies published in the open literature specific to the impacts of DSTD are fewer (e.g., Hughes et al., 2015), and those in the gray literature are often not readily accessible (e.g., Shimmield et al., 2010; LIPI, 2014; Simpson and Angel, 2015).

Deep-sea organisms may differ from those in shallow water in having slower growth rates, greater longevity, and less exposure to disturbance or variable environmental conditions, increasing their vulnerability to the changes related to tailings disposal. The benthic marine fauna can be impacted in a number of different ways by discharged tailings. The deposited material can kill the organisms directly through smothering and asphyxiation, through contact or poisoning via ingestion or exposure to water-dissolved substances. Mortality can also occur through the destruction of sensitive juveniles and through the killing of prey organisms (Brewer et al., 2007; Shimmield et al., 2010; Reichelt-Brushett, 2012). Benthic fauna at the disposal site and in the vicinity of the plume may bioaccumulate metals from tailings porewater and ingestion of sediment (Rainbow, 2007; Casado-Martinez et al., 2010; Campana et al., 2012). These impacts can lead to ecosystem-level changes. In shallow water situations, such as marinas, elevated sediment copper levels alter faunal composition and reduce macrofaunal biodiversity, total biomass and individual body size, as compared to sites with lower sediment copper concentrations (Neira et al., 2011, 2014).

Where mine tailings have been disposed, shifts in meiofaunal and macrofaunal community structure have been observed, with reduced biodiversity (e.g., Lee and Correa, 2005; Shimmield et al., 2010; Hughes et al., 2015); these effects on the environment can last several years after tailings disposal has ceased. Monitoring around the Island Copper Mine located on Vancouver Island (British Columbia) showed lower benthos diversity and abundance in tailings depositional areas, but no consistent reduction in crab catch associated with tailings discharge. Extensive benthic repopulation was observed in areas that had not undergone tailings deposition for 12 months, although assemblages of recolonizing organisms differed from reference sites (Poling et al., 1993). This shift in composition after recovery seems to be a common pattern and could result in shifting ecosystem functions (Gollner et al., 2017).

A large disturbance such as tailings disposal may result in changes in the flux, species composition of settling larvae and colonists, which could be facilitated by the disappearance of former residents (Mullineaux et al., 2010). Characterization of regional water masses and their seasonality is of great relevance, as these waters can transport different assemblages of colonizing larvae at different times of the year (e.g., Calderon-Aguilera et al., 2003; Adams et al., 2011). Benthic animals tend to recolonize disturbed areas at a slower rate with increasing water depth (Smith and Hessler, 1987). Tailings deposition, with increased metal mobilization and altered biogeochemistry, could create "no settlement zones" for larvae (Marinelli and Woodin, 2002). Shallow-water field experiments using defaunated sediments, with contrasting natural metal loading, spiked sediments, translocation and replacement showed changes in recolonizing infaunal and hard-substrate fauna composition with reduced biodiversity and lower structural complexity (Hill et al., 2013; Neira et al., 2015). As the same high-level taxa are present and the same principles govern sediment assemblages in shallow and deep waters, analogous responses could be expected for mine tailings disposed in the deep sea, but at a much larger spatial scale, covering a wider range of habitats and ecosystems. In deep continental margin settings, sediment recolonizes are facilitated by strong currents (Levin and DiBacco, 1995), and in chemosynthetic systems, by hydrogen sulfide (Levin et al., 2006). Thus, the order of species arrival, and the rate at which the faunal community will regenerate after tailings disposal has ceased will be site-specific, and also species-specific (Hughes et al., 2015). However, even higher-taxon level identification is sufficient to detect large-scale tailings impact in shallow water environments (Lee and Correa, 2005) and deep-sea sediments (Montagna et al., 2013; Hughes et al., 2015; Mevenkamp et al., 2017).

In most places where DSTD are conducted or proposed, knowledge of the deep-sea ecosystem, including the faunal composition, the functions, and the services provided are poorly known. Such knowledge becomes particularly important at the regional scale beyond the impact area, as larger areas may be impacted by resuspension, slope failure, plume sheering, etc., as well as by changes in connectivity affecting source populations for recovery.

SOCIETAL IMPACT AND MITIGATION

Reducing the Risk of Environmental Impacts

General "good practice" for all proposed mining operations commences with background and feasibility studies, including an EIA. These practices are intended to engage and inform communities, government and the industry proponents of the benefits and risks posed for triple bottom line outcomes (social, environmental and economic). All stakeholders are then provided a period of time to consider the studies, and comment, before potential revisions are made and approvals are sought for the options providing the best outcomes.

The EIA should clearly articulate the pertinent issues and uncertainties surrounding the short- and long-term impacts, including cumulative impacts. Within the EIA, tailings management will also consider options for on-land tailing impoundments (TSF, dams, ponds). The attributes of DSTD that are reported as favorable by mining companies over TSFs frequently include:

- Less impact to land (e.g., clearing of forest, loss of agricultural land)
- Less impact to communities that may be displaced or have land-based livelihoods altered
- Lower risks associated with natural events (high rainfall or earthquakes; floods)

- Avoidance of many of the problems of acid rock drainage, and its long term management

Within the vicinity of the DSTD outfall, significant adverse impacts on the marine ecosystem are predicted in the form of (i) reduction in seawater quality due to elevated turbidity and dissolved metal/metalloid concentrations (ii) burial of benthic organisms, and (iii) potential reduced habitat for demersal fish. When the DSTD ceases operation at completion of ore processing, the intent would be that the seawater quality improves within days to weeks and the benthic ecosystem recovers during a period of years (e.g., within 2–10 years), but the timescales for the recovery of ecosystems are poorly understood. Other frequently-cited positive environmental attributes of DSTD which are uncertain include:

- The ability to engineer the DSTD such that the tailings remain below the surface mixed layer and photic zone (beneath the pycnocline), flow rapidly to great depths (slopes >12°), and deposit within predictable regions of the sea floor.
- The depositional area can be chosen to be a zone of relatively low biological productivity (i.e., not impacting a vulnerable ecosystem), and maintain low chemical reactivity and low physical disturbance of the tailings in the future.
- The availability of locations that have an absence of upwelling events or seasonal mixing, or that are truly low productivity environments.

Considerable evidence exists that tailing plumes cannot be avoided, and the fate of plumes of fine tailings particles within the water column has been difficult to predict (Boschen et al., 2013; Hughes et al., 2015). Tailing plumes contribute to sometimes large amounts of tailings depositing outside predicted areas (e.g., 15–80% outside), requiring the need to develop better 3D modeling techniques that are capable of more accurately predicting tailings deposition areas (footprints). Demonstrating that no tailings resurface and disperse within the euphotic zone as a dissolved or particulate plume requires extensive monitoring which will be required as part of the Operational Environmental Management Plan (OEMP) of the mine, without which the mine cannot operate. The ecological risk assessment should include evaluating the risk of adverse effects to aquatic organisms both within the water column (pelagic organisms) and sediment environment (benthic organisms).

The first requirement for good practice concerns the completion of comprehensive, high-quality baseline studies that provide information on the receiving environment: detailed bathymetry and physical oceanography (e.g., local and seasonal information on frequency and intensity of currents and current-shearing, upwelling and downwelling, storms), sedimentology, and ecosystem (e.g., coastal and deep-sea community structure, function, connectivity, and resilience). To achieve suitable levels of background information on the dynamics of the abiotic and biotic systems, studies will generally need to be conducted over many years. A limited number of studies have addressed interactions and connections among abiotic and biotic systems, and new methods and approaches are needed. There are many knowledge gaps, for example, how the daily vertical migrators and benthopelagic coupling of living organisms are influenced by tailing plumes (Morello et al., 2016).

The second stage of good practice involves the engineering and modeling. Engineering aspects include: design, quality, and operational management for the life of the DSTD. These elements will include the de-aeration of the tailings and other conditioning to achieve the desired density and rheology, the tailing pipe network, materials, maintenance, stability of the optimal outfall depth, and quality controls for detection of leakage. A suitable level of understanding and monitoring of residual process chemicals (e.g., xanthates for floatation, lime for pH control, or specialized flocculants) is also needed. The engineering may need frequent adaptions to cope with changes in ore processing that may influence the environment downstream of the DSTD. The behavior of the tailings is modeled and takes into account oceanographic conditions, tailings volume and composition to predict the behavior of the discharge (direction, rates of transport and deposition) in relation to the bathymetry to estimate the tailings footprint. Based on this, ecological models could be developed outlining the estimated main and potential areas of impact to the water column and benthos. Tailings depositions outside initial model-predicted areas of DSTDs are estimated as ∼15% at Batu Hijau, Indonesia (LIPI, 2014; Simpson and Angel, 2015).

Due to the uncertainty surrounding predicted environmental risks of DSTDs, there is a need to undertake extensive monitoring and ongoing review of operations (stage three of good practice). The monitoring program should be designed to provide transparent evidence that the environmental management objectives are being met, e.g., demonstration of minimal impacts to the biologically productive surface waters (e.g., the surface mixed layer and photic zone), of no tailings deposition in near-shore coastal environments, and for impacts from the deep-sea deposition to be occurring in the predicted area. The tailings management systems and monitoring programs should span the processes from the mine to the sea, starting with the upstream management of ores and mine water on site and in the processing plant, then proceeding to tailings management via controls relating to engineering (e.g., tailings rheology, integrity of land seabed pipes) and tailing quality (e.g., quantities of oxidized forms), possible treatment options (e.g., re-sulfidization), and finally to monitoring of tailings disposal impacts within the marine environment. The management and monitoring programs should provide information that enables issues to be rapidly identified (e.g., extremes such as pipe breakages or surfacing tailings), and allows continuous improvement to the management and monitoring programs. This is detailed in the OEMP and is a requirement of the environment permit.

Routine monitoring should include a network of stations both within and beyond the predicted DSTD impact zone (encompassing the full water depth range of the receiving environment). The monitoring should include: the volume, physical and chemical characteristics of the tailings prior to discharge (e.g., crucial parameters monitored daily, and other parameters weekly to monthly); the coastal environment surrounding the DSTD (e.g., CTD profiles, seawater quality, total suspended solids, sediment quality, and various components of the marine ecosystem (Shimmield et al., 2010; Simpson and Angel, 2015).

Further technical studies may be necessary to support, validate or investigate aspects of the operations or potential impacts that cannot be adequately evaluated from routine monitoring data (LIPI, 2014), or to modify and optimize closure plans (stage four of good practice). Specialist studies may be undertaken upstream of the DSTD (e.g., relating to changes in processing techniques that influence tailings properties) or downstream (e.g., hydrodynamics, plume behavior, chemistry or ecological impacts). These studies should take advantage of new technologies to optimize data acquisition in remote deep-sea habitats for example, higher resolution bathymetry mapping, advanced autonomous underwater vehicles (AUVs) and advances in eco-genomics-based methods to provide more holistic information on impacts to ecological community structure, functions connectivity and resilience. These studies will often be vital to improving monitoring programs, and adapting management practices for positive outcomes during the mine life and post-closure (ongoing good practice).

For all assessments, there is a need to consider multiple lines of evidence (LOE) in order to adequately evaluate risks posed by mine tailings to the environment (Simpson and Batley, 2016; Mestre et al., 2017). For many deep-sea assessments, the desired LOE may not be readily assessed using existing tools, resulting in greater uncertainty. Prime examples include the lack of species that are representative of deep-sea environments that can be utilized for aquatic toxicity testing (Mestre et al., 2014; Brown et al., 2017), and the inadequate knowledge of deep-sea ecosystem structures, functions and connectivity to enable informed ecological assessments. Consequently, there will be a need to develop new tools to provide new LOE and to validate these for the specific assessment purposes. The assessments, approvals and monitoring will continue to improve as new science-based tools are developed to cover all aspects of chemistry, ecotoxicology, ecology, physical oceanography, and topography. This development is necessary to enable the most informed and robust management decisions associated with DSTD.

Although some of these aspects of reducing the risk of environmental impacts from DSTD outlined above are considered and incorporated within the approval and permitting processes, it is important that all are fully addressed. Owing to the numerous unavoidable uncertainties, it is recommended that permits are not issued for the entire mine life, but instead are for limited terms (e.g., 3–5 years) with thorough scrutiny of compliance and all operating procedures that may influence the environmental management objectives. This review process should enable clarification or improvements to be made to the objectives and permit requirements.

THE FUTURE OF DSTD

Potential Locations for Future DSTDs

For future DSTDs on the continental margin, a major consideration is that the slope is steeper than 12 degrees, to facilitate the formation of a turbidity current that will transport the tailings to a deep (below 1,000 m), and stable soft-sediment seafloor area (Shimmield et al., 2010; Ramirez-Llodra et al., 2015). Physically, it is essential that the selected site is not affected by upwelling events and is located in a low-energy environment (Shimmield et al., 2010). Ecologically, the transport and deposition areas should have low-productivity ecosystems, and avoid biodiversity hotspots and vulnerable communities (such as cold-water coral and sponge reefs, seamounts or cold seeps, amongst others). Submarine canyons are geomorphological structures that form deep incisions in most shelves and slopes around the world (Fernandez-Arcaya et al., 2017). Due to their particular geomorphological features, canyons modify the local hydrography and act as enhanced transport pathways of material from the productive coastal and shelf zones to the deep basins (Puig et al., 2014; Fernandez-Arcaya et al., 2017) influencing coastal upwelling process (Sobarzo and Djurfeldt, 2004). This enhanced downslope transport of particles has been used to justify certain canyons as preferential target sites for DSTD initiatives. Submarine canyons have been used for DSTDs in PNG (Basamuk canyon; ongoing), Indonesia (Senunu canyon, Sumbawa; ongoing) and France (Cassidaigne canyon; ongoing). The increased amount of scientific data available from canyons suggests that these complex geomorphological features can serve as essential habitats to marine communities, supporting an enhanced productivity and biodiversity, and are used as hatching, nursery or refuge areas. Canyons with steep rocky walls often support communities of sessile filter feeders such as corals (Roberts et al., 2009) or sponges (Schlacher et al., 2007, 2010), and the soft-sediment axis provides habitat to a variety of mobile fauna, including commercial species (Company et al., 2012). Furthermore, many canyons support complex marine food webs that include benthic species, but also pelagic decapods, fish, sharks and mammals attracted by the increased productivity and habitat heterogeneity (Vetter et al., 2010; van Oevelen et al., 2011; Moors-Murphy, 2014). The vulnerability of some of these biological communities, together with the limited knowledge of the processes and their temporal variability that drive canyon ecosystems, call for precaution and thorough long-term studies when considering a canyon as a DSTD site. In addition, the composition, diversity and ecosystem functions supported by canyons differ amongst regions, and thus it is essential to conduct local studies where canyons are proposed sites for DSTD activities.

Cumulative Impacts

Current trends for the disposing of mine tailings are increasingly considering deep-sea areas as final waste-deposit sites. At the same time, an increasing number of industries are targeting deep-sea resources, both mineral and biological (Ramirez-Llodra et al., 2011), contaminants are spreading to the deepest parts of the ocean (Jamieson et al., 2017), and ocean warming, acidification and deoxygenation affect ocean ecosystems globally, with impacts reaching the deep ocean (Levin and Le Bris, 2015). The synergies of different direct and indirect stressors on an ecosystem may result in a magnified effect with often poorly understood consequences, particularly in the deep sea where empirical studies are lacking (Ramirez-Llodra et al., 2011).

Below we describe key impacts that may coincide in space and time with DSTD operations, highlighting the need for strategic environmental assessments that will take into account multiple activities within a region.

Fishing

The slopes of many continental margins, including some canyon habitats, are fished intensively, with depth of operations gradually increasing as technologies develop (Morato et al., 2006). Those DSTD impacts that affect the water column and sediments (plumes, contaminants) could affect the diel vertical migrators and resident mesopelagic species as well as the benthic and demersal fish in ways that ultimately influence fisheries. Some studies, although not all, report an increase of metal content in fish tissues (Mol et al., 2001; Powell and Powell, 2001; Scroggins et al., 2002) and suggest that tailing plumes can disrupt migration routes and displace populations of commercially-important species (Sheaves, 2001; Brewer et al., 2007).

Confrontation between the fishing and mining industries have taken place in New Zealand and Namibia, where seabed phosphate mining licenses were applied for in regions that were closed to fishing (New Zealand) or that support a strong fishing industry (Namibia) (Levin et al., 2016). In Chile, mining is concentrated in the northern part of the country, and environmental impacts associated with this activity have a long history, among them, deposition of hundreds of millions of tons of tailings from copper mining into coastal beaches over the years (e.g., Vásquez et al., 1999; Lancellotti and Stotz, 2004). In 1990 disposal of untreated tailings was banned and further regulations and environmental requirements have been put in place (Advanced Conservation Strategies, 2011) in part because of concerns about fisheries. Potential conflicts of interest are possible if mining activity and future DSTD installations are near Benthic Resources Fisheries and Management Areas, and other types of Marine Protected Areas. In this regard, and in order to investigate the gaps related to the use of DSTP a number of mining companies formed an independent Consortium to conduct an impartial evaluation of DSTD (The DSTP Initiative, 2014). Mining projects have been rejected by the Chilean government arguing that mining close to MPAs could affect marine resources such as fisheries, as well as biodiversity, and other environmental services. An example is the case of Dominga Mining Company (Andes Iron) project, which was finally rejected in August 2017 by the Environmental Assessment Commission (http://www.bbc.com/news/world-latin-america-41007462?SThisFB).

Deep-Seabed Mining

Although the exploitation of deep-sea minerals has yet to begin, the interest in deep-seabed mining and the number of exploration licenses continue to grow internationally (Van Dover, 2011; Mengerink et al., 2014). The first *in situ* test mining for seafloor massive sulfides (SMS) took place in Japanese waters in 2017. It is likely to be several years before any commercial exploitation begins. In addition to potential cumulative risk and impacts, DSTDs and deep-sea mining activities have several processes in common (e.g., dispersal of sediment plumes; smothering of benthic fauna) and thus there is great benefit to sharing new knowledge, new technologies, methodologies and experience for the development of good practices and impact minimization.

Marine Genetic Resources

Among the ecosystem services provided by deep ocean ecosystems, there is increasing recognition of the value of genetic resources, including genes, proteins and natural products (Harden-Davies, 2017). New pharmaceuticals, industrial agents and biomaterials are originating from the deep sea, however, because so little of the deep ocean has been explored, most genetic resources have yet to be discovered. Disturbance to the sea floor from DSTD as well as extractive industries runs the risk of causing loss of these resources before they known.

Climate Change, Ocean Acidification, and Ocean Deoxygenation

The ocean environment below 200 m is changing rapidly as a result of heat uptake from the atmosphere (ocean warming), CO_2 uptake from the atmosphere (ocean acidification), and oxygen loss from thermal effects on oxygen solubility, stratification and ventilation (ocean deoxygenation) as well as from enhanced nutrient input (Levin and Le Bris, 2015; Sweetman et al., 2017). Deep continental margin ecosystems are highly vulnerable to these climate-related stressors, which act cumulatively with direct human disturbance from bottom trawling, oil and gas extraction and spills, pollution, and potentially seabed mining (Levin and Le Bris, 2015). The inorganic and organic particles including contaminants within the plumes from DSTD have the potential to further alter biogeochemistry at the seafloor and in the water column, compounding climate-related temperature, oxygen and pH stress. These cumulative effects are likely to be widespread, altering habitat properties, many ecological functions, (e.g., biodiversity, calcification by habitat-forming species like cold water corals) and ecosystem services (e.g., carbon sequestration, nutrient cycling, and fisheries production).

LEGISLATION AND RESEARCH GAPS

Current Legislation

When considering DSTD, it is important to consider the international legislation framework. The "Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter 1972," the "London Convention" is one of the first global conventions to protect the marine environment from human activities and has been in force since 1975. Its objective is to promote the effective control of all sources of marine pollution and to take all practicable steps to prevent pollution of the sea by dumping of wastes and other matter. Currently, 87 States are Parties to this Convention.

In 1996, the "London Protocol" was agreed to further modernize the Convention and, eventually, replace it. The Protocol entered into force on 24 March 2006 and there are currently 45 Parties to the Protocol. Under the Protocol all dumping is prohibited, except for wastes on the "reverse list." Rather than state which materials may not be dumped, the

1996 Protocol restricts all dumping except from a permitted list of eight major categories, including "Inert, inorganic geological material," under which tailings may fall.

Although the national waters of a State are excluded from both the Convention and Protocol, Parties to the Protocol have the option to apply its rules to their waters if they wish (Article 7).

It is important to note that the London Convention and Protocol (LC/LP) do not cover discharges from land-based sources such as pipes and outfalls, wastes generated incidental to normal operation of vessels, or placement of materials for the purposes other than mere disposal, providing such disposal is not contrary to aims of the Convention. Therefore, the LC/LP do not directly apply to DSTD. In addition the LC/LP allows the dumping of "inert, geological material", and mining organizations argue that as mine tailings are geological in origin, they are also "inert," and therefore do not contravene the LC/LP.

In October 2008 (under increasing pressure from non-government organizations, particularly Greenpeace) the governing bodies under the LC/LP agreed for a more detailed assessment of mine tailings, in order that effective control of sub-sea tailings discharges may be considered and communicated to relevant bodies, including the United Nations Environment Program (UNEP) Global Programme of Action (GPA) for Protection of the Marine Environment from Land-Based Activities. The GPA is unique in that it directly addresses the connectivity between terrestrial, freshwater, coastal and marine ecosystems. GPA targets major threats to the health, productivity, and biodiversity of marine and coastal environments resulting from human activities on land. Importantly the GPA is not binding, but provides a framework for governments in close partnership with all stakeholders to address some of the most significant threats to marine ecosystems.

In 2009, the International Maritime Organization (IMO) submitted a paper entitled "Initial proposals for co-operation between the London Convention and Protocol and the UNEP Global Programme of Action for Protection of the Marine Environment from Land-based Activities (GPA)," (Annex 4), which considers coastal management issues and investigates options for co-operation between the LC/LP and the UNEP-GPA and the UNEP Regional Seas programme to deal with coastal management issues. This collaborative policy response is still being considered.

Although pipeline discharges and other land-based sources of marine pollution fall beyond the regulatory scope of the LC/LP, it is noted by LC/LP Scientific Groups that the discharge of such tailings frequently falls beyond the scope of any effective international regulatory control. The LC/LP has been interested in riverine and submarine disposal of tailings and associated wastes, including cooperation of the LC/LP Secretariat at the IMO with UNEP cooperation of the LC/LP GPA, in gathering information on the issue. The LC/LP Secretariat commissioned a report on the issue, which was submitted to the LC/LP Scientific Groups and discussed at the LC/LP meetings in November 2013. LC/LP Scientific Groups agreed there is a need for international guidance and/or codes of conduct to be developed but, as GESAMP noted, there is a governance gap and it is not clear which international body should take the lead. The LC/LP agreed to establish an intersessional correspondence group.

Further international concern led to an international workshop held in June 2015 in Peru led by IMO-GESAMP, and co-organized by the MITE-DEEP project (funded by the Norwegian Research Council and INDEEP) and the Chilean DSTD initiative. The final report of the Workshop has been published by the GESAMP Office and the LC/LP Secretariat (GESAMP, 2016). In addition, there is a working group in the process of developing guidelines; the group is led by Perú. Most recently in 2017 a new working group (WG42) has been established by GESAMP (sponsored by IMO and UNEP) on the impacts of wastes and other matter in the marine environment from mining operations, including marine minerals mining. Finally, as reported in their Annual Report, OSPAR are also considering the need for Guidance on the deep-sea disposal of mine tailings (OSPAR Commission, 2017).

In 2014, the European Commission started a process to review and adapt the first (2009) Reference Document on Best Available Techniques for Management of Tailings and Waste-Rock in Mining Activities, to include all new knowledge and available techniques. The release of this reviewed "Best Available Techniques Reference Document for the Management of Waste from Extractive Industries" is expected at the end of 2017. However, the draft version of this document focuses mostly on land-based processes of extractive waste and discusses only briefly issues related to submarine tailing disposal.

Research Priorities to Address Current Knowledge Gaps

A thorough review of the pros and cons of DSTDs against a land-based dam for the management of mine tailings under different scenarios (e.g., existing and potential DSTDs) would be a useful exercise. The main limitation to conducting this essential exercise before making the decision to dispose tailings in the marine or land systems is the limited knowledge of many marine community structures and processes, including the value of the services provided. Here, we highlight some of the major gaps in knowledge that need to be addressed, before a rigorous science-based evaluation of the advantages and disadvantages of marine vs. land tailings management can take place. The priority topics for future research have been selected based on discussions during the GESAMP/MITE-DEEP/INDEEP workshop and post-workshop deliberations and include expert comments from a variety of scientific disciplines, industry and policy makers from broad geographical regions. Detailed consideration should be given to the following issues:

- *Tailings dispersal* in the water column: there is a need for better spatio-temporal physical oceanographic data to feed into more accurate models (e.g., data on shearing currents and plume generation) and ground-truthing of models.
- Engineering developments to change *tailings behavior* (turbidity currents, plume generation) to reduce impacts.
- *Post-deposition behavior* of tailings: what are the potential physical (resuspension, slope failures) and chemical (reactions

in water column and sediments) processes affecting deposited tailings and their components?
- Inventories of the *materials being deposited*: what is the nature of processing chemicals and what is their behavior in different environments?
- Detailed *faunal community* studies: what is the composition and structure of benthic and pelagic fauna (using morphological and molecular tools)?
- *Ecosystem function*: what are the trophic relationships in the ecosystem? What bentho-pelagic coupling processes are found? What are the reproductive patterns of the key species and are they affected by tailings deposition? What microbial processes take place in the sediment? What ecosystem services derive from these functions and how can they be impacted by DSTDs?
- Effects on *pelagic early life-history* stages: what is the effect of plumes (particles and toxicity) on eggs, larvae and juveniles?
- Effect on *recolonization and settlement*: what are the different effects of varying grain size, organic matter content and grain shape/sharpness? What are the sensitivities of early life stages to mine-derived chemical contaminants?
- *Ecotoxicity*: although most EIAs include ecotoxicity tests, these are conducted on standard, shallow-water species, and the results may not be relevant to deep-sea species. There is an urgent need to develop similar analyses for deep-sea species, but the difficulty of collecting and maintaining deep-sea fauna alive in laboratory conditions continues to limit these studies.
- *Evaluate cumulative impacts* from different direct and indirect stressors.
- *Identify thresholds* to evaluate serious harm.
- *Assess long-term fate* of tailings in deep-sea ecosystems.
- *Value deep-sea ecosystem services* to provide the necessary information to managers assessing the cost-benefit of DSTD applications and deciding upon required compensation for lost services.
- *Circular economy*: further research is necessary in the reprocessing of tailings to extract value and minimize waste volume.
- *Legislation and standard rules/guidelines for Good Practice*: further rollout of guidelines for the use of DSTD.

DSTD IN A BROADER CONTEXT

Although DSTD is a very local practice associated with terrestrial mining, the activities and impacts associated with disposal of terrestrial tailings in the deep sea are likely to affect a broader spectrum of stakeholders. While most of these human activities occur within a national jurisdiction, they may impact transboundary organisms that exhibit ontogenetic or migratory movements into international waters, or the national waters of other countries.

As suggested above, the monitoring and research conducted for DSTD may also inform management of other activities. For example, the substrate modifications, sediment plumes, sediment deposition, toxic compounds/heavy metals and, to a lesser extent, noise are also features of deep-seabed minerals mining. Long term studies of plume behavior, benthic impacts, and faunal recovery rates as described for the Batu Hijau project in Indonesia (LIPI, 2014; Simpson and Angel, 2015) or the Lihir gold mine in PNG could inform regulation or even decision making about seabed mining, which has been proposed within Exclusive Economic Zones (EEZs) of many island nations in the Western Pacific Ocean (SPC, 2013), as well as in international waters (Levin et al., 2016). Although the seabed mineral resources differ (massive sulfides, polymetallic nodules, and cobalt crusts), their extraction will resuspend sediments and yield many similar physical impacts to DSTD. The emerging regulations and science of deep-seabed mining have much to gain by utilizing the scientific studies carried out on DSTD, and by learning from the past regulatory successes and failures of DSTD.

At the international level, there are a number of deep-focused oceanographic data networking programs that may be able to help inform plume modeling, risk assessment and post-disposal recovery. Among these are the nascent Deep Ocean Observing Strategy (www.deepoceanobserving.org), a GOOS project which will help coordinate accessibility of deep-sea biological and hydrographic data, and the Ocean Biogeographic Data System (OBIS), which is initiating a deep-sea node for biological/biodiversity data. The Deep Ocean Stewardship Initiative (DOSI, http://dosi-project.org/) and the International Network for Scientific Investigations of Deep-Sea Ecosystems (INDEEP, http://www.indeep-project.org/) both facilitate capacity building and can offer scientific and policy expertise on various aspects of anthropogenic impact in the deep ocean, including DSTD. Engagement with these entities, and with other international activities such as the UN Sustainable Developmental Goal (SDG) 14, which advocates for sustainable oceans, may be helpful to regulators in developing countries where much of the current DSTD occurs, as these often do not have their own long-term, deep-ocean monitoring programs, or scientists and policy experts familiar with the deep ocean.

There are now numerous organizations associated with UNCLOS that regulate or are responsible for various aspects of biodiversity in the ocean (Ardron and Warner, 2015): the Convention on Biological Diversity (CBD) in national waters, the International Seabed Authority for the Area (international seafloor), the Food and Agriculture Organization (FAO) for fish in international waters, the International Maritime Organization (IMO) for contaminants in international waters. Each of these organizations has identified some form of protected area—Ecologically or Biologically Significant Marine Areas (EBSAs) by the CBD, Vulnerable Marine Ecosystems (VMEs) by the FAO, Areas of Particular Environmental Interest (APEIs) by the International Seabed Authority (ISA), and Particularly Sensitive Sea Areas (PSSAs) by the IMO (Diz et al., 2017). There is an understanding that we need to identify, monitor, and minimize DSTD sediment plumes, contaminants or other ecosystem alterations that are transported from the margins into national Exclusive Economic Zones (EEZs) and international waters, especially those that may affect protected areas. The new UN treaty being negotiated to protect biodiversity in international waters is addressing impact assessment, marine protected areas, genetic resources and capacity building (Blasiak et al., 2016), all issues relevant to DSTD.

AUTHOR CONTRIBUTIONS

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SUPPLEMENTARY MATERIAL

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REFERENCES

Adams, D. K., McGillicuddy, D. J., Zamudio, L., Thurnherr, A. M., Liang, X., Rouxel, O., et al. (2011). Surface-generated mesoscale eddies transport deep-sea products from hydrothermal vents. *Science* 332, 580–583. doi: 10.1126/science.1201066

Advanced Conservation Strategies (2011). *A Coastal-Marine Assessment of Chile*. A report prepared for the The David and Lucile Packard Foundation.

Almeida, A., Cuhna, A., Fernandes, S., Sobral, P., and Alcantara, F. (2007). Copper effects on bacterial activity of estuarine silty sediments. *Estuar. Coast. Shelf Sci.* 73, 743–752. doi: 10.1016/j.ecss.2007.03.013

Angel, B. M., Simpson, S. L., Jarolimek, C. V., Jung, R., Waworuntu, J., and Batterham, G. (2013). Trace metals associated with deep-sea tailings placement at the Batu Hijau copper-gold mine, Sumbawa, Indonesia. *Mar. Pollut. Bull.* 73, 306–313. doi: 10.1016/j.marpolbul.2013.04.013

ANZECC/ARMCANZ (2000). *Australian and New Zealand Guidelines for Fresh and Marine Water Quality*. Canberra, ACT: Australian and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand.

Apte, S. C., and Kwong, Y. T. J. (2004). *Deep Sea Tailings Placement: Critical Review of Environmental Issues*. Review Undertaken by CSIRO Australia and CANMET Canada.

Ardron, J., and Warner, R. M. (2015). "International marine governance and protection of biodiversity," in *Routledge Handbook of Ocean Resources and Management*, eds H. D. Smith, J. de Vivero, and T. S. Agardy, 55–72. Available online at: https://www.routledge.com/products/9780415531757

Barth, M. E., and McNichols, M. F. (1994). Estimation and market valuation of environmental liabilities relating to superfund sites. *J. Account. Res.* 32, 177–209. doi: 10.2307/2491446

Blasiak, R., Pittman, J., Yagi, N., and Sugino, H. (2016). Negotiating the use of biodiversity in marine areas beyond national jurisdiction. *Front. Mar. Sci.* 3:224. doi: 10.3389/fmars.2016.00224

Boschen, R. E., Rowden, A. A., Clark, M. R., and Gardner, J. P. A. (2013). Mining of deep-sea seafloor massive sulfides: a review of the deposits, their benthic communities, impacts from mining, regulatory frameworks and management strategies. *Ocean Coast. Manage.* 84, 54–67. doi: 10.1016/j.ocecoaman.2013.07.005

Bourgeois, S., Archambault, P., and Witte, U. (2017). Organic matter remineralization in marine sediments: a Pan-Arctic synthesis. *Glob. Biogeochem. Cycles* 31, 190–213. doi: 10.1002/2016GB005378

Brewer, D. T., Milton, D. A., Fry, G. C., Dennis, D. M., Heales, D. S., and Venables, W. N. (2007). Impacts of gold mine waste disposal on deepwater fish in a pristine tropical marine system. *Mar. Pollut. Bull.* 54, 309–321. doi: 10.1016/j.marpolbul.2006.10.011

Brown, A., Thatje, S., and Hauton, C. (2017). The effects of temperature and hydrostatic pressure on metal toxicity: insights into toxicity in the deep sea. *Environ. Sci. Technol.* 51, 10222–10231. doi: 10.1021/acs.est.7b02988

Burd, B. J. (2002). Evaluation of mine tailings effects on a benthic marine infaunal community over 29 years. *Mar. Environ. Res.* 53, 481–519. doi: 10.1016/S0141-1136(02)00092-2

Calderon-Aguilera, L. E., Marinone, S. G., and Aragon-Noriega, E. A. (2003). Influence of oceanographic processes on the early life stages of the blue shrimp (Litopenaeus stylirostris) in the Upper Gulf of California. *J. Mar. Syst.* 39, 117–128. doi: 10.1016/S0924-7963(02)00265-8

Campana, O., Spadaro, D. A., Blasco, J., and Simpson, S. L. (2012). Sublethal effects of copper to benthic invertebrates explained by changes in sediment properties and dietary exposure. *Environ. Sci. Technol.* 46, 6835–6842. doi: 10.1021/es2045844

Casado-Martinez, C., Smith, B. D., Luoma, S. N., and Rainbow, P. S. (2010). Metal toxicity in a sediment-dwelling polychaete: threshold body concentrations or overwhelming accumulation rates? *Environ. Pollut.* 158, 3071–3076. doi: 10.1016/j.envpol.2010.06.026

Castilla, J. C., and Nealler, E. (1978). Marine environmental impact due to mining activities of El Salvador copper mine, Chile. *Mar. Pollut. Bull.* 9, 67–70. doi: 10.1016/0025-326X(78)90451-4

Company, J. B., Ramirez-Llodra, E., Sardà, F., Puig, P., Canals, M., Calafat, A., et al. (2012). "Submarine canyons in the Catalan Sea (NW Mediterranean): megafaunal biodiversity patterns and anthropogenic threats" in *Mediterranean Submarine Canyons: Ecology and Governance*, ed W. Würtz (Gland; Malaga: IUCN), 133–144.

Diz, D., Johnson, D., Riddell, M., Rees, S., Battle, J., Gjerde, K., et al. (2017). Mainstreaming marine biodiversity into the SDGs: the role of other effective area-based conservation measures (SDG 14.5). *Mar. Policy*. doi: 10.1016/j.marpol.2017.08.019. [Epub ahead of print].

Dold, B. (2014). Submarine tailings disposal (STD) – a review. *Minerals* 4, 642–666. doi: 10.3390/min4030642

Ellis, D., and Ellis, K. (1994). Very deep STD. *Mar. Pollut. Bull.* 28, 472–476. doi: 10.1016/0025-326X(94)90519-3

ESMAP/World Bank/ICMM. (2005). "Community development toolkit—pioneering New Approaches in support of sustainable development in the extractive sector," in *Energy Sector Management Assistance Programme* (Washington, DC; London: World Bank; International Council on Mining and Metals). Available online at: www.icmm.com or www.esmap.org

Fernandez-Arcaya, U., Ramirez-Llodra, E., Aguzzi, J., Allcock, A. L., Davies, J. S., Dissanayake, A., et al. (2017). Ecological role of submarine canyons and need for canyon conservation: a review. *Front. Mar. Sci.* 4:5. doi: 10.3389/fmars.2017.00005

Flores, F., Hoogenboom, M. O., Smith, L. D., Cooper, T. F., Abrego, D., and Negri, A. P. (2012). Chronic exposure of corals to fine sediments: lethal and sub-lethal impacts. *PLoS ONE* 7:e37795. doi: 10.1371/journal.pone.0037795

GESAMP (2016). *Proceedings of the GESAMP International Workshop on the Impacts of Mine Tailings in the Marine Environment*. IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/UNEP/UNDP Joint Group of Experts on the

Scientific Aspects of Marine Environmental Protection. Reports and Studies GESAMP No. 94.

Gollner, S., Kaiser, S., Menzel, L., Jones, D. O. B., Brown, A., Mestre, N. C., et al. (2017). Resilience of benthic deep-sea fauna to mining activities. *Mar. Environ. Res.* 129, 76–101. doi: 10.1016/j.marenvres.2017.04.010

Hanrahan, D., Ericson, B., and Caravanos, J. (2016). Protecting communities by remediating polluted sites worldwide. *Proc. Institut. Civil Engineers* 169, 33–40. doi: 10.1680/jcien.15.00034

Harden-Davies, H. (2017). Deep-sea genetic resources: new frontiers for science and stewardship in areas beyond national jurisdiction. *Deep-Sea Res.* Part 11 *Trop. Oceanogr.* 137, 504–513. doi: 10.1016/j.dsr2.2016.05.005

Hill, N. A., Simpson, S. L., and Johnston, E. L. (2013). Beyond the bed: effects of metal contamination on recruitment to bedded sediments and overlying substrata. *Environ. Pollut.* 173, 182–191. doi: 10.1016/j.envpol.2012.09.029

Hughes, D. J., Shimmield, T. M., Black, K. D., and Howe, J. A. (2015). Ecological impacts of large-scale disposal of mining waste in the deep sea. *Sci. Rep.* 5:9985. doi: 10.1038/srep09985

Jamieson, A. J., Malkocs, T., Piertney, S. B., Fujii, T., and Zhang, Z. (2017). Bioaccumulation of persistent organic contaminants in the deepest ocean fauna. *Nat. Ecol. Evol.* 1, 1–4. doi: 10.1038/s41559-016-0051

Jonas, R. B. (1989). Acute copper and cupric ion toxicity in an estuarine microbial community. *Appl. Environ. Microbiol.* 55, 43–49.

Jones, R., Ricardo, G. F., and Negri, A. P. (2015). Effects of sediments on the reproductive cycle of corals. *Mar. Pollut. Bull.* 100, 13–33. doi: 10.1016/j.marpolbul.2015.08.021

Kathman, R. D., Brinkhurst, R. O., Woods, R. E., and Jefries, D. C. (1983). *Benthic Studies in Alice Arm and Hastings Arm, B.C., in Relation To Mine Tailings Dispersal.* Canadian Technical Report of Hydrography and Ocean Sciences, No. 22.

Kizil, M. S., and Muller, B. (2011). "A numerical approach for prefeasibility analysis of tailings disposal options," *35th APCOM Symposium* (Woolongong, NSW).

Kvassnes, A. J. S., and Iversen, E. (2013). Waste sites from mines in Norwegian Fjords. *Mineral Produksjon* 3, A27–A38.

Lancellotti, D. A., and Stotz, W. B. (2004). Effects of shoreline discharge iron tailings on a marine soft-bottom community in northern Chile. *Mar. Pollut. Bull.* 48, 303–312. doi: 10.1016/j.marpolbul.2003.08.005

Lee, M. R., and Correa, J. A. (2005). Effects of copper mine tailings disposal on littoral meiofaunal assemblages in the Atacama region of northern Chile. *Mar. Environ. Res.* 59, 1–18. doi: 10.1016/j.marenvres.2004.01.002

Levin, L. A., and DiBacco, C. (1995). Influence of sediment transport on short-term recolonization by seamount infauna. *Mar. Ecol. Prog. Ser.* 123, 163–175. doi: 10.3354/meps123163

Levin, L. A., Mengerink, K., Gjerde, K. M., Rowden, A. A., Van Dover, C. L., Clark, M. R., et al. (2016). Defining "Serious Harm" to the marine environment in the context of deep-seabed mining. *Mar. Policy* 74, 245–259 doi: 10.1016/j.marpol.2016.09.032

Levin, L. A., Ziebis, W., Mendoza, G. F., Growney-Cannon, V., and Walther, S. (2006). Recruitment response of methane-seep macrofauna to sulfide and surrounding habitat. *J. Exp. Mar. Biol. Ecol.* 330, 132–150. doi: 10.1016/j.jembe.2005.12.022

Levin, L. A., and Le Bris, N. (2015). The deep ocean under climate change. *Science* 350, 766–768. doi: 10.1126/science.aad0126

LIPI (2014). *Batu Hijau Deep Sea Tailings Placement Deep Sea Study, November 2013.* Join Research PT Newmont Nusa Tenggara and Research Centre for Oceanography, Indonesian Institute of Sciences (RCO-LIPI).

Loring, D. H., and Asmund, G. (1989). Heavy metal contamination of a Greenland fjord system by mine wastes. *Environ. Geol. Water Sci.* 14, 61–71. doi: 10.1007/BF01740586

Marinelli, R. L., and Woodin, S. A. (2002). Experimental evidence for linkages between infaunal recruitment, disturbance, and sediment surface chemistry. *Limnol. Oceanogr.* 47, 221–229. doi: 10.4319/lo.2002.47.1.0221

McKinnon, E. (2002). The environmental effects of mining waste disposal at Lihir Gold Mine, Papua New Guinea. *J. Rural Remote Environ. Health* 1, 40–50.

Mengerink, K. J., Van Dover, C. L., Ardron, J., Baker, M. C., Escobar-Briones, E., Gjerde, K., et al. (2014). A call for deep-ocean stewardship. *Science* 344, 696–698. doi: 10.1126/science.1251458

Mestre, N. C., Calado, R., and Soares, A. M. (2014). Exploitation of deep-sea resources: the urgent need to understand the role of high pressure in the toxicity of chemical pollutants to deep-sea organisms. *Environ. Pollut.* 185, 369–371. doi: 10.1016/j.envpol.2013.10.021

Mestre, N. C., Rocha, T. L., Canals, M., Cardoso, C., Danovaro, R., Dell'Anno, A., et al. (2017). Environmental hazard assessment of a marine mine tailings deposit site and potential implications for deep-sea mining. *Environ. Pollut.* 228, 169–178. doi: 10.1016/j.envpol.2017.05.027

Mevenkamp, L., Stratmann, T., Guilini, K., Moodley, L., van Oevelen, D., Vanreusel, A., et al. (2017). Impaired short-term functioning of a benthic community from a deep norwegian fjord following deposition of mine tailings and sediments. *Front. Mar. Sci.* 4:169. doi: 10.3389/fmars.2017.00169

Middelburg, J. J., and Levin, L. A. (2009). Coastal hypoxia and sediment biogeochemistry. *Biogeosciences* 6, 1273–1293. doi: 10.5194/bg-6-1273-2009

Mineral Policy Institute (1999). *Environmental Risks Associated with Submarine Tailings Discharge in Astrolable Bay, Madang Province, Papua New Guinea.* Sydney, NSW: Marine Policy Institute.

Mol, J. H., Ramlal, J. S., Lietar, C., and Verloo, M. (2001). Mercury contamination in freshwater, estuarine, and marine fishes in relation to small-scale gold mining in Suriname, South America. *Environ. Res.* 86A, 183–197. doi: 10.1006/enrs.2001.4256

Montagna, P. A., Baguley, J. G., Cooksey, C., Hartwell, I., Hyde, L. J., Hyland, J. L., et al. (2013). Deep-sea benthic footprint of the Deepwater Horizon blowout. *PLoS ONE.* 8:e70540. doi: 10.1371/journal.pone.0070540

Moors-Murphy, H. B. (2014). Submarine canyons as important habitat for cetaceans, with special reference to the Gully: a review. *Deep-Sea Res. II.* 104, 6–19. doi: 10.1016/j.dsr2.2013.12.016

Morato, T., Watson, R., Pitcher, T. J., and Pauly, D. (2006). Fishing down the deep. *Fish Fish.* 7, 24–34. doi: 10.1111/j.1467-2979.2006.00205.x

Morello, E. B., Haywood, M. D. E., Brewer, D. T., Apte, S. C., Asmund, G., Kwong, Y. T. J., et al. (2016). The ecological impacts of submarine tailings placement. *Oceanogr. Mar. Biol.* 54, 315–366. doi: 10.1201/9781315368597-7

Morse, D. E., and Glover, A. N. (2000). *Minerals and Materials in the 20th Century: A Review. U.S. Geological Survey Minerals Yearbook.* U.S. Geological Survey (USGS).

Mullineaux, L. S., Adams, D. K., Mills, S. W., and Beaulieu, S. E. (2010). Larvae from afar colonize deep-sea hydrothermal vents after a catastrophic eruption. *Proc. Natl. Acad. Sci. U.S.A.* 107, 7829–7834. doi: 10.1073/pnas.0913187107

NCIR (National Intelligence Council Report) (2013). *Natural Resources in 2012, 2030, and 2040: Implications for the United States.* NCIR 2013-05, 25 July 2013. Available online at: https://www.dni.gov/files/documents/NICR%202013-05%20US%20Nat%20Resources%202020,%202030%202040.pdf

Neira, C., Levin, L. A., Mendoza, G., and Zirino, A. (2014). Alteration of benthic communities associated with copper contamination linked to boat moorings. *Mar. Ecol.* 35, 46–66. doi: 10.1111/maec.12054

Neira, C., Mendoza, G., Levin, L. A., Zirino, A., Delgadillo-Hinojosa, F., and Porrachia, M. (2011). Macrobenthic community response to copper in Shelter Island Yacht Basin, San Diego Bay, California. *Mar. Pollut. Bull.* 62, 701–717. doi: 10.1016/j.marpolbul.2011.01.027

Neira, C., Mendoza, G., Porrachia, M., Stransky, C., and Levin, L. A. (2015). Macrofaunal recolonization of copper-contaminated sediments in San Diego Bay. *Mar. Pollut. Bull.* 101, 794–804. doi: 10.1016/j.marpolbul.2015.09.023

OSPAR Commission (2017). *Annual Report 2016-2017.* OSPAR Secretariat, London. Available online at: https://www.ospar.org/documents?v=38007

Palkovits, F. (2007). Thickened tailings offer effective disposal alternatives. *Engineer. Mining J.* 208, 62–67.

Pedersen, T. A. (1984). Interstitial water metabolite chemistry in a marine mine tailings deposit, Rupert Inlet, B.C. Canadian *J. Earth Sci.* 21, 1–9. doi: 10.1139/e84-001

Pedersen, T. A., and Losher, A. J. (1988). "Diagenetic processes in aquatic mine tailings deposits in British Columbia," in *Chemistry and Biology of Solid Waste*, eds W. Salomons and U. Forstner (Berlin: Springer-Verlag), 238–258.

Perner, K., Leipe, T.h., Dellwig, O., Kuijpers, A., Mikkelsen, N., Andersen, T. J., et al. (2010). Contamination of Arctic Fjord sediments by Pb-Zn mining at Maarmorilik in central West Greenland. *Mar. Pollut. Bull.* 60, 1065–1073. doi: 10.1016/j.marpolbul.2010.01.019

Poling, G. W., Ellis, D. V., Pelletier, C. A., Pedersen, T. F., and Hesse, C. A. (1993). "Case studies of submarine tailings disposal," in *Vol. I – North American*

Examples, eds G. W. Poling and D. V. Ellis, U.S. Department of the Interior, Bureau of Mines Open-File Report, 89–93.

Powell, J. H., and Powell, R. E. (2001). Trace elements in fish overlying subaqueous tailings in the tropical west Pacific. *Water Air Soil Pollut.* 125, 81–104. doi: 10.1023/A:1005211832691

Puig, P., Palanques, A., and Martín, J. (2014). Contemporary sediment-transport processes in submarine canyons. *Ann. Rev. Mar. Sci.* 6, 53–77 doi: 10.1146/annurev-marine-010213-135037

Rainbow, P. S. (2007). Trace metal bioaccumulation: models, metabolic availability and toxicity. *Environ. Int.* 33, 576–582. doi: 10.1016/j.envint.2006.05.007

Ramirez-Llodra, E., Trannum, H. C., Evenset, A., Levin, L. A., Andersson, M., Finne, T. E., et al. (2015). Submarine and deep-sea mine tailing placements: a review of current practices, environmental issues, natural analogs and knowledge gaps in Norway and internationally. *Mar. Pollut. Bull.* 97, 13–35. doi: 10.1016/j.marpolbul.2015.05.062

Ramirez-Llodra, E., Tyler, P. A., Baker, M. C., Bergstad, O. A., Clark, M. R., Escobar, E., et al. (2011). Man and the last great wilderness: human impact on the deep sea. *PLoS ONE.* 6:e22588. doi: 10.1371/journal.pone.0022588

Reichelt-Brushett, A. (2012). Risk assessment and ecotoxicology. Limitations and recommendations for ocean disposal of mine waste in the Coral Triangle. *Oceanography* 25, 40–51. doi: 10.5670/oceanog.2012.66

Reichelt-Brushett, A., Hewitt, J., Lynch, P., Kaiser, S., Wood, R., and Kim, R. (2016). *Report on the PACE Net Seed Funding Project: Identifying Ecological Risk Assessment Tools and Approaches for Deep Sea Mining Activities to Support Communities, Government, and Developing Industries in the South Pacific.* Southern Cross University, Australia.

Roberts, J. M., Wheeler, A., Freiwald, A., and Cairns, S. (2009). *Cold-Water Corals, The Biology and Geology of Deep-Sea Coral Habitat.* Cambridge: Cambridge University Press.

Schlacher, T. A., Schlacher-Hoenlinger, M. A., Williams, A., Althaus, F., Hooper, J. N. A., and Kloser, R. (2007). Richness and distribution of sponge megabenthos in continental margin canyons off southeastern Australia. *Mar. Ecol. Prog. Ser.* 340, 73–88. doi: 10.3354/meps340073

Schlacher, T., Williams, A., Althaus, F., and Bryce, M. (2010). High resolution seabed imagery as a tool for conservation planning on continental margins. *Mar. Ecol.* 31, 200–221. doi: 10.1111/j.1439-0485.2009.00286.x

Scroggins, R., Van Aggelen, G., and Schroeder, J. (2002). Monitoring sublethal toxicity in effluent under the metal mining EEM program. *Water Qual. Res. J. Canada* 37, 279–294. Available online at: https://www.cawq.ca/journal/temp/article/168.pdf

Sheaves, M. (2001). "Chapter 5: an analysis of the ecology of Astrolabe Bay in relation to the Ramu Nickel Cobalt Mine," in *A Review of the Risks Presented by the Ramu Nickel Project to the Ecology of Astrolabe Bay in Papua New Guinea,* ed P. Shearman (Sydney: Mineral Policy Institute), 62–83.

Shimmield, T. M., Black, K. D., Howe, J. A., Hughes, D. J., and Sherwin, T. (2010). *Final Report: Independent Evaluation of Deep-Sea Mine Tailings Placement (DSTP) in PNG.* Oban: SAMS.

Simpson, S. L., and Angel, B. M. (2015). *Review of PTNNT Studies Relating to the Tailings Footprint in the Sea Floor. CSIRO Land and Water Report EP158344*, 15.

Simpson, S. L., and Batley, G. E. (2016). *Sediment Quality Assessment: A Practical Guide.* Victoria: CSIRO Publishing.

Simpson, S. L., and Spadaro, D. A. (2016). Bioavailability and chronic toxicity of metal sulfide minerals to benthic marine invertebrates: implications for deep sea exploration, mining and tailings disposal. *Environ. Sci. Technol.* 50, 4061–4070. doi: 10.1021/acs.est.6b00203

Smith, C. R., and Hessler, R. R. (1987). Colonization and succession in deep-sea ecosystems. *Trends Ecol. Evol.* 2, 359–363. doi: 10.1016/0169-5347(87)90136-4

Sobarzo, M., and Djurfeldt, L. (2004). Coastal upwelling process on a continental shelf limited by submarine canyons, Concepción, central Chile. *J. Geophys. Res.* 109:C12012. doi: 10.1029/2004JC002350

SPC (2013). "Deep sea minerals: deep sea minerals and the green economy," in *Secretariat of the Pacific Community*, Vol. 2, eds E. Baker and Y. Beaudoin, PP122. Available online at: http://dsm.gsd.spc.int/public/files/meetings/TrainingWorkshop4/UNEP_vol2.pdf

Sweetman, A. K., Thurber, A. R., Smith, C. R., Levin, L. A., Mora, C., Wei, C. L., et al. and Roberts, J.M. (2017). Major impacts of climate change on deep seafloor ecosystems. *Elem. Sci. Anth.* 5:4. doi: 10.1525/elementa.203

The DSTP Initiative (2014). *The DSTP Initiative: 2014 Knowledge Workshop Report. Final Report May.*

Vanclay, F. (2004). The triple bottom line and impact assessment: how do TBL, EIA, SIA, SEA, and EMS relate to each other? *J. Environ. Assess. Policy Manage.* 6 265–288. doi: 10.1142/S1464333204001729

Van Dover, C. L. (2011). Tighten regulations on deep-sea mining. *Nature* 470, 31–33. doi: 10.1038/470031a

van Oevelen, D., Soetaert, K., Garcia, R., de Stigter, H. C., Cunha, M. R., Pusceddu, A., et al. (2011). Canyon conditions impact carbon flows in food webs of three sections of the Nazaré canyon. *Deep Sea Research Part II* 58, 2461–2476. doi: 10.1016/j.dsr2.2011.04.009

Vásquez, J. A., Vega, J. M. A., Matsuhiro, B., and Urzúa, C. (1999). The ecological effects of mining discharges on subtidal habitats dominated by macroalgae in northern Chile: population and community levels studies. *Hydrobiologia* 398, 217–229. doi: 10.1023/A:1017054517382

Vetter, E. W., Smith, C. R., and De Leo, F. C. (2010). Hawaiian hotspots: enhanced megafaunal abundance and diversity in submarine canyons on the oceanic islands of Hawaii. *Mar. Ecol.* 31, 183–199. doi: 10.1111/j.1439-0485.2009.00351.x

Vogt, C. (2012). *International Assessment of Marine and Riverine Disposal of Mine Tailings.* London: Secretariat, London Convention/London Protocol International Maritime Organization London, England; United Nations Environment Programme—Global Program of Action.

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