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Environmental Impacts of the Deep-Water Oil and Gas Industry: A Review to Guide Management Strategies

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The industrialization of the deep sea is expanding worldwide. Increasing oil and gas exploration activities in the absence of sufficient baseline data in deep-sea ecosystems has made environmental management challenging. Here, we review the types of activities that are associated with global offshore oil and gas development in water depths over 200 m, the typical impacts of these activities, some of the more extreme impacts of accidental oil and gas releases, and the current state of management in the major regions of offshore industrial activity including 18 exclusive economic zones. Direct impacts of infrastructure installation, including sediment resuspension and burial by seafloor anchors and pipelines, are typically restricted to a radius of ~100 m on from the installation on the seafloor. Discharges of water-based and low-toxicity oil-based drilling muds and produced water can extend over 2 km, while the ecological impacts at the population and community levels on the seafloor are most commonly on the order of 200–300 m from their source. These impacts may persist in the deep sea for many years and likely longer for its more fragile ecosystems, such as cold-water corals. This synthesis of information provides the basis for a series of recommendations for the management of offshore oil and gas development. An effective management strategy, aimed at minimizing risk of significant environmental harm, will typically encompass regulations of the activity itself (e.g., discharge practices, materials used), combined with spatial (e.g., avoidance rules and marine protected areas), and temporal measures (e.g., restricted activities during peak reproductive periods). Spatial management measures that encompass representatives of all of the regional deep-sea community types is important in this context. Implementation of these management strategies should consider minimum buffer zones to displace industrial activity beyond the range of typical impacts: at least 2 km from any discharge points and surface infrastructure and 200 m from seafloor infrastructure with no expected discharges. Although managing natural resources is, arguably, more challenging in deep-water environments, inclusion of these proven conservation tools contributes to robust environmental management strategies for oil and gas extraction in the deep sea.

Keywords: offshore drilling, deep sea, environmental impacts, benthic communities, cold-water corals, chemosynthetic ecosystems, environmental policy, marine spatial planning

INTRODUCTION

Exploration of oil and gas deposits is now a global industrial activity in the deep ocean. As easily accessible oil and gas resources became depleted, and technology improved, the oil and gas industry expanded into deeper waters in recent decades (**Figure 1**). However, this deep-water expansion has not always been matched by legislation that reflects modern practices of environmental conservation. There is a clear need to bring together current knowledge of deep-sea ecology, known human impacts on deep-water ecosystems, and the scattered environmental protection measures that exist to date.

Numerous and varied regulations related to the management of the hydrocarbon industry exist in different maritime jurisdictions and for areas beyond national jurisdiction (ABNJ or the "Area"; Mazor et al., 2014; Katsanevakis et al., 2015). Individual nation states may manage activities within their exclusive economic zones (EEZs), complemented by the United Nations Convention on the Law of the Sea (UNCLOS; note that the U.S.A. has not ratified the Convention) considering mineral extraction activities outside EEZs. Such regulations may, for example, set out the framework for environmental assessment and monitoring, define particular habitats, and/or species that should be afforded particular protection, and define the boundaries of areas designated for spatial management. However, there has not yet been a significant effort to standardize regulations across EEZs or to develop regional management organizations as exist for high-seas fisheries management.

Application of management strategies in the deep sea is complicated by the unique ecological proscenium on which they play out (Jumars and Gallagher, 1982). Biological systems in the deep sea operate at a notably slower pace than in shallow waters (Smith, 1994). Many deep-sea species typically have low metabolic rates, slow growth rates, late maturity, low levels of recruitment, and long life spans (McClain and Schlacher, 2015). Many deep-sea habitats also harbor diverse faunal assemblages that are composed of a relatively large proportion and number of rare species at low abundances (Glover et al., 2002). In some habitats (e.g., hydrothermal vents) species can re-colonize relatively rapidly after disturbance (Van Dover, 2014), but in most other deep-sea ecosystems, recovery can be very slow (Williams et al., 2010; Vanreusel et al., 2016). These attributes make deep-sea species and assemblages sensitive to anthropogenic stressors, with low resilience to disturbances from human activities (Schlacher et al., 2014; Clark et al., 2016).

Here, we seek to synthesize current information on typical impacts from offshore oil and gas operations and review existing management strategies and regulations in order to provide the basis for a set of recommendations for a generalized management strategy to limit environmental impacts attributable to the deep-water (>200 m) oil and gas industry. Protective measures can include spatial management (i.e., spatial restrictions, marine protected areas), activity management (i.e., restrictions to industry methods), and temporal management (i.e., temporary or seasonal restrictions). These forms of management have been implemented and enforced with varying degrees of success in a number of jurisdictions. Given the highly variable nature of local management regulations, some individual deep-water oil and gas industry operators have adopted in-house best practice approaches and/or imported operating constraints from other jurisdictions to limit their liability in regions with little or no management system in place. However, there remains no standard set of best practice approaches that has broad-based support.

DEEP-WATER OIL AND GAS INDUSTRY

Industrial exploitation of oil and gas reserves has occurred in shallow marine areas since 1897, when the wells drilled at sea from piers in Summerland, California, first produced

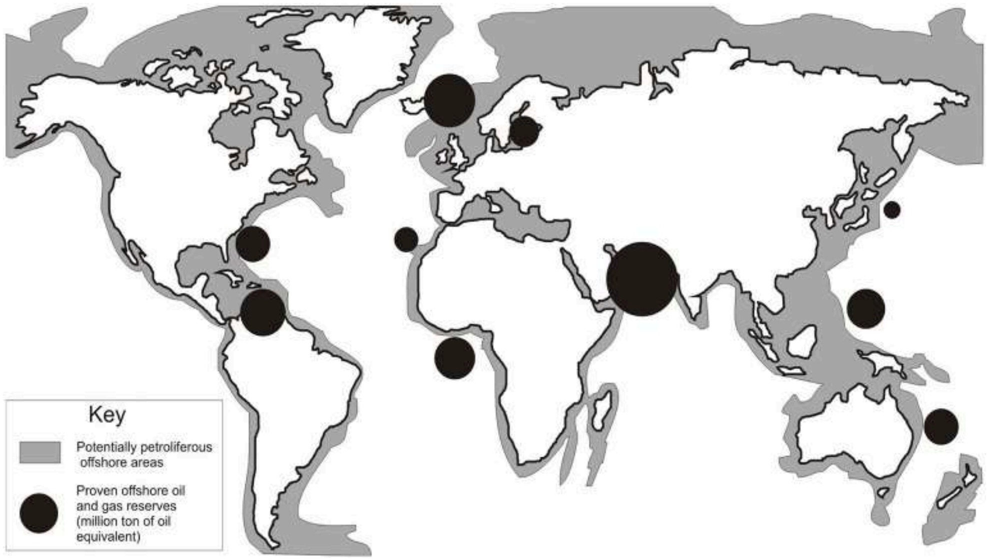

FIGURE 1 | Potentially petroliferous offshore zones and regional distribution of proven offshore oil and gas reserves. Adapted from Pinder (2001).

oil (Hyne, 2001). By the 1960s, this drilling had moved into deeper offshore areas as easily accessible resources declined, technology for offshore drilling improved, and large reserves of hydrocarbons were discovered. Currently, drilling for oil and gas is routine in all offshore environments, with major deep-water (>200 m) production in areas such as the Arctic, northern North Atlantic Ocean (UK and Norwegian waters), East and West Africa, Gulf of Mexico, South America, India, Southeast Asia, and Australia (**Figure 1**). Ultra-deep-water (>1000 m) production is still in its early stages and is likely to increase in the coming years, with the most active development in the Gulf of Mexico, where major reserves are being accessed in waters as deep as 3000 m. Gas-hydrate extraction is still in the development phase, and while many of the conclusions and recommendations included here could be applied to that nascent industry, we do not explicitly consider those activities here.

Deep-water exploration involves multiple steps (Kark et al., 2015), typically starting with acoustic remote sensing (seismic surveys) to understand the subsurface geology and potential hydrocarbon reservoir architecture (Gausland, 2003). If suitable targets are detected, one or more exploration wells are drilled to ground-truth the interpretation of the acoustic data and determine the nature of the reservoir. If economically recoverable hydrocarbon reserves are located, the site may advance to production (Hyne, 2001). This typically involves the drilling of one or more appraisal wells followed by several production wells and the installation of various surface (e.g., floating production, storage, and offloading vessels) and subsea infrastructure (e.g., manifolds, control cables, and export lines). An example of a large deep-water operation is the BP Greater Plutonio field off Angola, which covers an area of 140 km^2 and consists of 43 wells in water depths of 1200–1500 m. Once a field is operational (this may take several years to complete), hydrocarbons are exported via pipelines and/or tankers. Additional drilling may be required as the field develops, either to expand the field or to enhance oil or gas recovery (Boesch and Rabalais, 1987).

In deep-water settings, drilling is typically from semi-submersible rigs or drill ships that hold station by anchors or dynamic positioning (**Figure 2**). In a production field, the various wells are connected together with a series of pipes and control cables (Hyne, 2001). Individual wells may be 1 m in diameter, and are often several kilometers in length. Drilling an individual well may take between 1 and 3 months. The drilling process involves the use of fluids that perform a number of different functions (e.g., providing hydrostatic pressure, cooling, and cleaning the drill, carrying drill cuttings, limiting corrosion, lubrication). The fluid may be seawater or a combination of chemicals often referred to as drilling mud (see Sections below). A steel pipe, known as the casing, is pushed into the well behind the drill and eventually cemented in place (Hyne, 2001). Typically, for the first section of the well, which may extend 600 m into the sediment, there is no retention of the drill cuttings (the fragments of rock that have been drilled) and these are pushed to the seafloor surface through the casing with the drilling fluid, and form a "cuttings pile" (Jones et al., 2006). Once this first section (the "tophole") is completed and cemented in place, a blow-out preventer (BOP) is installed at the seabed (Hyne, 2001). The BOP contains a series of valves controlling the well, and once it is in place, the well is effectively sealed and the drilling fluids and cuttings can be recirculated to the rig for processing and recycling. Following processing to reduce or eliminate oil content and stabilize and/or solidify the waste, drill cuttings can be discharged overboard, may be shipped to shore for further processing and disposal, or re-injected into the seabed (Boesch and Rabalais, 1987; Ball et al., 2012).

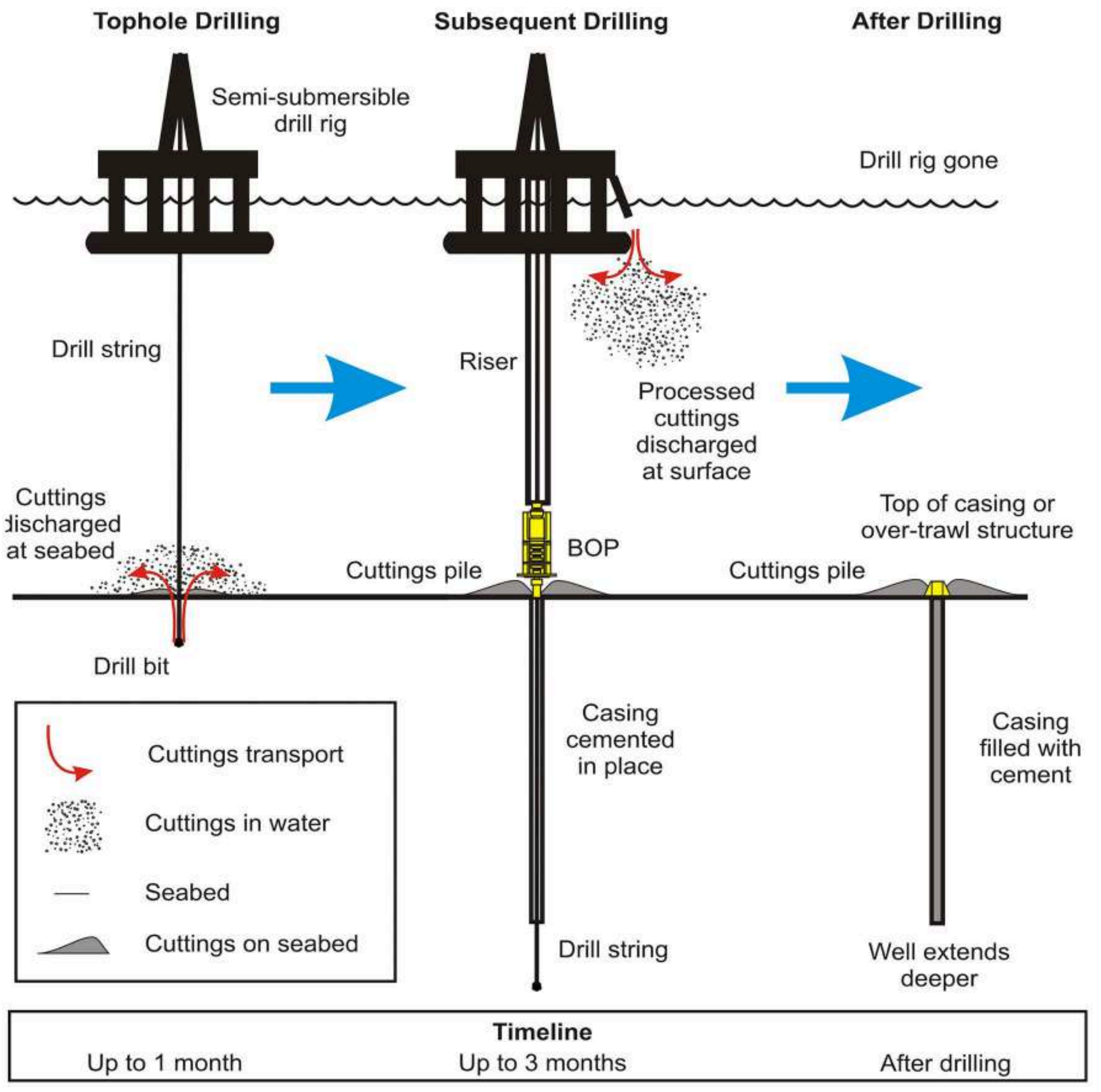

FIGURE 2 | Primary sediment discharges made during exploration drilling activity in deepwater. These effects are nearly identical whether a semi-submersible rig (as shown) or a drillship is used for drilling.

ASSESSMENT OF ENVIRONMENTAL IMPACTS

Environmental impacts of oil and gas operations may influence species, populations, assemblages, or ecosystems by modifying a variety of ecological parameters (e.g., biodiversity, biomass, productivity, etc.). At the project level, potential impacts are generally assessed through some type of formal process, termed an environmental impact assessment (EIA). These typically involve the identification, prediction, evaluation, and mitigation of impacts prior to the start of a project. Key standard components of an EIA include: (i) description of the proposed development, including information about the size, location, and duration of the project, (ii) baseline description of the environment, (iii) description of potential impacts on the environment, (iv) proposed mitigation of impacts, and (v) identification of knowledge gaps. Mitigation in current oil and gas projects is recommended to follow the mitigation hierarchy: avoid, minimize, restore, and offset (World Bank, 2012). Environmental management strategies, particularly those to avoid and minimize the environmental impacts of projects, are set during the EIA process and may become conditions of operation. As a result, this element of the EIA process is particularly important in preemptively avoiding serious impacts to the marine environment (Beanlands and Duinker, 1984). Establishing appropriate baseline data and control reference sites are critical to both an effective EIA development and subsequent assessment and monitoring of EIA predictions.

EIAs include predictions of how an ecological "baseline" condition may change in response to development and activities. Regulatory bodies generally offer advice on the appropriate assessment of potential impacts on ecological parameters such as biodiversity. For example, the UK Department for Environment, Food and Rural Affairs (DEFRA) suggests consideration of: (i) gains or losses in the variety of species, (ii) gains or losses in the variety and abundance within species, (iii) gains or losses in the amount of space for ecosystems and habitats, (iv) gains or losses in the physical connectedness of ecosystems and habitats, and (v) environmental changes within ecosystems and habitats. The DEFRA advice notes that the assessment of biodiversity will necessarily require some baseline knowledge against which to assess a proposed development and any potential impact that may result.

The reliability of EIA predictions depends largely on the quality of existing ecological data (e.g., spatial and temporal coverage, measures of natural variation, taxonomic resolution, types of fauna observed, and collected, etc.) and empirical data or model predictions of how ecological features react to human stressors. Even in the best-known deep-sea environments, the need for planned, coherent, and consistent ecological data to inform EIAs may necessitate substantial new survey operations. For example, within the UK EEZ, the Faroe-Shetland Channel has been the subject of extensive oceanographic investigations since the late 1800s (e.g., Thomson, 1873). Nevertheless, the oil industry and the UK's regulatory bodies considered it appropriate to undertake a major regional-scale survey of the deep-water environment at the onset of industry activity (Mordue, 2001). In the Gulf of Mexico, region-wide assessments of deep-sea community structure are available for different groups of fauna (e.g., Rowe and Menzel, 1971; Cordes et al., 2006, 2008; Rowe and Kennicutt, 2008; Demopoulos et al., 2014; Quattrini et al., 2014). However, following the Deepwater Horizon incident, baseline data were still found to be lacking in the immediate vicinity of the impacts, and for many key components of the ecosystem, including microbial communities and processes (Joye et al., 2016). This is reflected in the primary recommendation of a recent review (Turrell et al., 2014) that assessed the science needed to respond to a UK deep-water oil spill, which highlighted the need for the development of robust "physical, chemical, and biological baselines" in deep-water oil and gas production areas.

Testing EIA predictions and the effectiveness of implemented mitigation measures with well-designed and consistent environmental monitoring is a critical next step. Generally, some form of "before-after/control-impact" (BACI) monitoring approach is appropriate (Underwood, 1994), as this will enable the detection of accidental impacts in addition to impacts anticipated from typical operations (Wiens and Parker, 1995; Iversen et al., 2011). However, this often receives less attention and resources than the EIA itself, and most jurisdictions have minimal requirements for monitoring programs (**Table 1**). Long-term monitoring in the deep sea is generally rare (e.g., Hartman et al., 2012), and long-term environmental monitoring of deep-water oil and gas developments is extremely limited. A significant exception is found in the two observatory systems that were installed in deep waters off Angola to record long-term natural and anthropogenic changes in the physical, chemical, and biological environment and to allow an understanding of the pace of recovery from unforeseen impacts (Vardaro et al., 2013). Monitoring should also be carried out after production has ceased and throughout de-commissioning. For example, in Norway such monitoring is required at 3-year intervals during the production phase and following the cessation of production (Iversen et al., 2011).

Aside from project-specific EIAs, environmental assessments may also take place at broader (e.g., regional or national) levels, for example in the form of Strategic Environmental Assessments (SEAs). Such broad assessments may cover a single industrial sector or multiple sectors, and may involve broad analyses of environmental and socio-economic impacts of development plans. These assessments are typically aimed at assisting regulatory bodies with identifying development options that can achieve both sustainable use and national and international conservation goals (Noble, 2000; Jay, 2010). Despite the recognized benefit of integrating strategic/regional assessments into the planning and management process, their application in offshore activity planning is still relatively limited (Noble et al., 2013). Examples of regional assessments for offshore oil and gas development are known from Canadian Atlantic waters (e.g., LGL Ltd., 2003), the Norwegian Barents Sea (Hasle et al., 2009), the UK offshore area (e.g., Geotek Ltd. and Hartley Anderson Ltd., 2003), and the Gulf of Mexico (e.g., Minerals Management Service, 2003). Assessment procedures (e.g., in terms of legal mandate, objectives, process, level of detail) applied by these countries vary, but the assessments typically included the compilation of regional baseline data, identification of environmental sensitivities, and determination of where future hydrocarbon exploration could take place or should be avoided (Fidler and Noble, 2012).

EFFECTS OF ROUTINE ACTIVITIES

Routine oil and gas activities can have detrimental environmental effects during each of the main phases of exploration, production, and decommissioning (**Figure 3**). During the exploration phase, impacts can result from indirect (sound and traffic) and direct physical (anchor chains, drill cuttings, and drilling fluids) disturbance. Additional direct physical impacts occur in the production phase as pipelines are laid and the volume of discharged produced water increases. Lastly, decommissioning can result in a series of direct impacts on the sea floor and can re-introduce contaminants to the environment. It is critical that all of the potential impacts of routine operations are accounted for when designing management strategies, whether local or regional, for offshore oil and gas activities.

Impacts from deep-water oil and gas development activities begin during seismic surveys that are used to reveal the subsurface geology and locate potential reservoirs. These impacts include underwater sound and light emissions and increased vessel activity. Sound levels produced during seismic surveys vary in intensity, but in some cases, soundwaves from these surveys have been detected almost 4000 km away from the survey vessel (Nieukirk et al., 2012). Impact

TABLE 1 | Summary of some examples of regulations pertaining to the protection of marine habitats and species in various EEZs around the world.

Jurisdiction	What is protected	Implementation of protection	Status of oil and gas activities	Assessment and monitoring
Australia	Sensitive features and values of the environment, particularly the presence of threatened species	Site-specific environmental plans developed by operators and vetted by commonwealth authority	Possibly, following EIA	Each activity requires an environment plan approved by legislator, details not prescribed
Barbados	Some coral reefs and fisheries that fit conservation priorities	MPAs, small MPAs in place in coastal habitats	Possibly allowed within existing MPAs, following EIA	EIAs required, monitoring for emissions, discharges, biological indicators. 5 year review cycle
Brazil	Cold-water corals	Designation as conservation unit	"Sustainable use" allowed if deep-water corals are avoided	Monitoring of water, sediments, and biota required but methods not stipulated
Canada	Listed species, cold-water corals, unique/diverse/productive habitats	MPA designation, Areas of Interest, Sensitive Benthic Areas, Fishery closures, Marine Parks, Species-at-risk	Requires EIA and public comment period	Monitoring encouraged for exploration, mitigation plans and monitoring required for production
Colombia	Coastal and marine areas that fit conservation objectives	National Natural Parks System, regional Districts of Integrated Management, Regional Natural Parks	"Sustainable use" allowed following EIA evaluation	EIA required, monitoring required, but methods not stipulated
Grenada	Coastal reefs, offshore fisheries, pollution of offshore areas prohibited	Benthic Protection Areas (fisheries), MPAs (coastal habitats)	Possibly, following EIA	Required but not described
Israel	Unique habitats, high species richness, rare species, archeological sites	Proposal for establishment of MPA system, considering 600 m set-back distance	Possibly, following EIA	Strategic environmental survey required within 2 km, sediment sampling throughout, 8 video surveys within 500 m
Jamaica	Coastal coral reefs, some offshore fisheries, discharge of "poisonous, noxious, or polluting matter" is prohibited	MPAs, Marine Parks, some in place in shallow waters	Possibly, following EIA	Baseline surveys completed, but not explicitly required
Malaysia	Fisheries and habitat quality, CITES listed species		Possibly, following EIA and public comment period	EIA carried out by registered consultants, evaluation of impacts in accordance with international standards
Mozambique	No specific protections outlined. Rules for avoiding impacts and preventing deposition of toxic substances in the ocean			EIA is required
New Zealand	Sensitive environments and threatened species	MPA system in development, currently avoidance or mitigation	Possibly, following EIA	Baseline surveys for EIA only
Nigeria		No specific marine protections, but signatory on various international agreements	Possibly, following EIA	
Norway	Valuable and vulnerable areas, fisheries, sensitive species (e.g., corals)	Currently defining a framework for oil and gas activities within Norwegian Climate and Pollution Agency	EIA is required prior to drilling	Baseline surveys required, monitoring required after production, monitoring includes fish condition, and benthic habitat condition assessments every 3 years

(Continued)

TABLE 1 | Continued

Jurisdiction	What is protected	Implementation of protection	Status of oil and gas activities	Assessment and monitoring
Portugal	Habitats and Species listed in EU Habitats Directive	System of MPAs. Existing leases in all mainland EEZ, no exploitation yet	Possibly, following EIA	During exploration phase, all measures should be taken to prevent pollution; EIA is only required for the exploitation phase
Tanzania		Legislation stipulating that "Environmental protections should follow best practices of industry"		
Trinidad and Tobago	Sensitive areas and sensitive species	MPA system being developed, one currently for shallow water reefs	Possibly, following EIA	Baseline surveys for EIA, monitoring endorsed but not required
UK	Habitats and Species listed in EU Birds and Habitats Directive, OSPAR Convention, and other national conservation legislation	Network of MPAs with designation as Special Area of Conservation, Nature Conservation MPAs, and Marine Conservation Zones	Possibly, following evaluation of EIA	Baseline surveys for EIA, monitoring endorsed but not required
US	High density biological communities	BOEM Notice to lessees, National Monuments, National Marine Sanctuaries	NTL excludes O&G activities excluded within 2000 feet (surface) or 500 feet (seafloor), others based on EIA evaluation	Mitigation areas determined from seismic anomalies. Visual surveys only required if near known high density communities. No monitoring required

There are a number of sub-Sahara African countries for which no records of governmental regulations exist, including Cameroon, Equatorial Guinea, Gabon, and Ghana (Ackah-Baidoo, 2012).

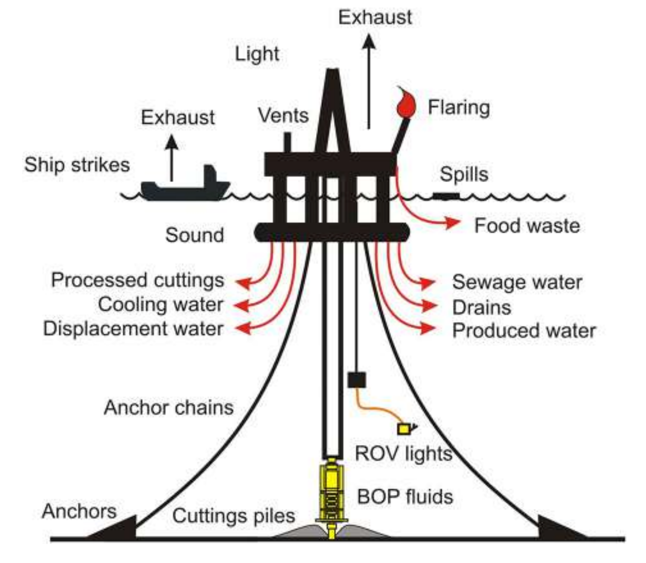

FIGURE 3 | Diagram of impacts from typical deep-sea drilling activity.

assessments of acoustic disturbance have primarily focused on marine mammals. Reported effects include disruption of behavior (e.g., feeding, breeding, resting, migration), masking of sounds used for communication and navigation, localized displacement, physiological stress, as well as physical injury including temporary or permanent hearing damage (Gordon et al., 2004; Southall et al., 2008; Moore et al., 2012). Marine mammal exposure experiments and noise propagation modeling suggest that hearing damage may occur within a few 100 m to km from the sound source, with avoidance behaviors more variable but generally detected over greater distances (Southall et al., 2008). In contrast, the potential effects of sound on fish and invertebrates remain poorly understood, but may be significant (Hawkins et al., 2014). For example, significant developmental delays and body malformations have been recorded in scallop larvae exposed to seismic pulses (de Soto et al., 2013). Exposure to underwater broadband sound fields that resemble offshore shipping and construction activity can also influence the activity and behavior of key bioturbating species in sediments (Solan et al., 2016).

Operations at oil fields introduce considerable amounts of artificial light (e.g., electric lighting, gas flares) that can potentially affect ecological processes in the upper ocean, such as diel vertical migration of plankton (Moore et al., 2000). Artificial night light also attracts numerous species, including squid, large predatory fishes, and birds (Longcore and Rich, 2004). Underwater lighting, such as used on remotely operated vehicles, is likely to be of comparatively modest impact, though it may be significant in the case of species with extremely sensitive visual systems (Herring et al., 1999).

Once the installation of infrastructure commences, direct impacts on habitats and associated fauna increase (**Table 2**). Placement of infrastructure on the seafloor, such as anchors and pipelines, will directly disturb the seabed and cause a transient increase in local sedimentation. Typically, 8–12 anchors are used

TABLE 2 | Types of impacts from offshore oil and gas activities.

	Concern	Nature	Extent	Environmental issues	References
	Drilling discharges (cuttings, drilling fluids, cement, chemicals)	Physical (excess sedimentation); Chemical (toxic effects; enrichment effects)	100–500 m (solids) "Local"	Smothering; clogging of feeding and gas exchange structures; direct toxicity; altered electrochemical environment; changes in nutrient availability, decreased species abundance, altered community structure	Reed and Hetland, 2002; Breuer et al., 2004; Jones et al., 2007; Netto et al., 2009; Pivel et al., 2009; Jones and Gates, 2010; Bakke et al., 2013; Larsson et al., 2013
	Produced water	Chemical (toxic effect)	1–2 km (produced water and dissolved components) "Widespread"	Direct toxicity; food-web contamination; potential food-chain; and trophic amplification	Bakke et al., 2013
Routine	Anchors	Physical (direct damage; hard substratum)	"Local"	Direct physical impact at emplacement, potentially continuing impact through tidally induced motions; provision of hard substratum for colonization by sessile epifauna and associates	
	Flow and control lines, umbilicals	Physical (direct damage; hard substratum)	"Local"	Direct physical impact at emplacement; increased sedimentation; provision of hard substratum for colonization by sessile epifauna and associates	
	Export pipelines	Physical (direct damage; hard substratum)	"Widespread"	Potentially extensive direct physical impact at emplacement; provision of hard substratum for colonization by sessile epifauna and associates	
	Risers	Physical (hard substratum in water column)	"Local"	Provision of hard substratum for colonization by sessile epifauna and associates	
	Anchors and pipelines	Direct physical disturbance	15 m (direct impacts), 50–100 m (indirect impacts)	Mortality and burial of benthic fauna; fragmentation of corals; increased sedimentation; pipelines can corrode; and increase toxicity	Ulfsnes et al., 2013
	Surface structures and vessels	Restricted movement of vessels	Right-of-way for working vessels; 1–2 km for surface infrastructure	Restricted industrial and scientific activity	
	Seabed infrastructure	Artificial habitat	Direct for sessile species, ~500 m for pelagic species, potentially altering distribution over large areas	Altered distribution; may increase species connectivity (including invasive species)	Doray et al., 2006; Gass and Roberts, 2006; Atchison et al., 2008; Larcom et al., 2014
	Artificial light	Physical (energy, electromagnetic spectrum)	100 s of m	Surface light attracts some mobile species and repels others; subsurface light impacts are largely unknown	Herring et al., 1999

(Continued)

TABLE 2 | Continued

	Concern	Nature	Extent	Environmental issues	References
	Acoustic energy	Physical (energy, hydrostatic pressure)	200–300 m (high intensity); up to 4000 km (lower intensity); highly variable	Localized auditory damage (100s of m), disruption of marine mammal behavior, and physiological stress; impacts to fish unknown; invertebrate larval impacts	Moore et al., 2012; Nieukirk et al., 2012; de Soto et al., 2013; Hawkins et al., 2014; Solan et al., 2016.
Accidental	Mass hydrocarbon release (atmosphere, seasurface, water column seafloor)	Release of oil and gas; potential deposition of equipment; potential additional affects of mitigation efforts (dispersants, burning, etc.)	Localized deposition of gear; pelagic and benthic impacts depend on size of event, ranging from 100 s of m to 100 s of km in diameter	Increased toxicity; altered benthic, pelagic, and infaunal communities; mortality of corals	Camilli et al., 2010; White et al., 2012; Montagna et al., 2013; Chanton et al., 2014; Fisher et al., 2014a,b; Valentine et al., 2014; Baguley et al., 2015
	Mass dispersant release	Chemical (toxic effects), synergistic with oil exposure	Variable, depending on size of event	Increased toxicity; Changes in microbial community; reduced settlement of larvae; mortality of corals	Epstein et al., 2000; Goodbody-Gringley et al., 2013; Kleindienst et al., 2015; DeLeo et al., 2016

to moor a semi-submersible drilling rig. The spatial extent of anchor impacts on the seabed varies depending on operating depth, but is typically between 1.5 and 2.5 times the water depth of the operation (Vryhof Anchors BV, 2010). As anchors are set, they are dragged along the seabed, damaging benthic organisms and leaving an anchor scar on the seafloor. The impact of anchors in the deep sea is of greatest concern in biogenic habitats, such as those formed by corals and sponges, which are fragile and have low resilience to physical forces (Hall-Spencer et al., 2002; Watling, 2014). Anchor operations have been shown to impact coral communities directly through physical disturbance and increased local sedimentation, with an estimated 100 m wide corridor of influence (Ulfsnes et al., 2013). The laying of pipelines also alters local seabed habitat conditions by adding hard substratum, which in turn may support sessile epifauna and/or attract motile benthic organisms (Lebrato and Jones, 2009). Ulfsnes et al. (2013) estimated a 50 m wide corridor of impact for pipeline installations, including dislocation of existing hard substrata. Corrosion and leakage of pipelines also poses the risk of exposing deep-sea fauna to potentially damaging pollution.

The drilling process involves the disposal of waste, including drill cuttings and excess cement, fluids (drilling mud), produced water, and other chemicals that may cause detrimental ecological effects (Gray et al., 1990). Drill cuttings are the fragments of rock that are created during the drilling process. The chemical composition of drilling muds is diverse, and has changed from the more toxic oil-based muds (currently restricted in many jurisdictions) to more modern synthetic and water-based fluids. The types of fluids most commonly used currently are generally regarded to be less toxic than oil-based fluids, but they are not without adverse biological effects (Daan and Mulder, 1996; Breuer et al., 2004; Bakhtyar and Gagnon, 2012; Gagnon and Bakhtyar, 2013; Edge et al., 2016). Produced water is contaminated water associated with oil and gas extraction process, with an estimated global production ratio of 3:1 water:oil over the lifetime of a well (Khatib and Verbeek, 2002; Neff, 2002; Fakhru'l-Razi et al., 2009). However, it should be noted that this is a global average, and these estimates vary greatly between hydrocarbon fields with the ratio of water to oil increasing over the lifetime of a single well. Produced water is primarily composed of formation water extracted during oil and gas recovery, but may also contain seawater that has previously been injected into the reservoir along with dissolved inorganic salts, dissolved and dispersed hydrocarbons, dissolved minerals, trace metals, naturally occurring radioactive substances, production chemicals, and dissolved gases (Hansen and Davies, 1994; Neff, 2002; Fakhru'l-Razi et al., 2009; Bakke et al., 2013). As a major source of contaminants from oil and gas extraction activity, produced water is typically treated in accordance with strict regulations before being discharged (e.g., OSPAR, 2001).

The spatial footprint of discharge varies with the volume of discharge, depth of discharge, local hydrography, particle size distribution, rates of settlement and floc formation, and time since discharge (Neff, 2005; Niu et al., 2009). Although volumes are likely to vary greatly depending on the local conditions during the active stage of drilling, discharges from one deep-water well at 900 m depth off the coast of Brazil were ∼270 m^3 of cuttings, 320 m^3 of water-based fluids, and 70 m^3 of non-aqueous fluids (Pivel et al., 2009). These types of discharges may produce cuttings accumulations up to 20 m in thickness within 100–500 m of the well site (Breuer et al., 2004; Jones et al., 2006; Pivel et al., 2009). Visual assessment at 10 recent deep-water well sites between 370 and 1750 m depth, drilled using current best practice in the NE Atlantic, recorded visual cuttings accumulations present over a radius of 50–150 m from the well head (Jones and Gates, 2010).

Potential impacts on seabed communities can result from both the chemical toxicants and the physical disturbance (see summary in **Table 3**, **Figure 4**). Reduction in oxygen concentration, organic enrichment, increased hydrocarbon concentrations, and increased metal abundance can alter biogeochemical processes and generate hydrogen sulfide and ammonia (Neff, 2002). At present, little information is available on the effects of these processes at the microbial level. At the metazoan level, community-level changes in the density, biomass, and diversity of protistan, meio-, macro-, and megafaunal assemblages have been recorded in several studies (Gray et al., 1990; Currie and Isaacs, 2005; Jones et al., 2007; Netto et al., 2009; Santos et al., 2009; Lanzen et al., 2016). These changes have been linked with smothering by drilling cuttings and increased concentrations of harmful metals (e.g., barium) and hydrocarbons (Holdway, 2002; Breuer et al., 2004; Santos et al., 2009; Trannum et al., 2010).

Detected ecological changes attributed to current practices have typically been found within 200–300 m of the well-head (Currie and Isaacs, 2005; Gates and Jones, 2012), but can occasionally extend to 1–2 km for sensitive species (Paine et al., 2014). Previous drilling practices, where oil-based drilling muds were used for the entire drilling process (use of such methods are currently heavily regulated in most jurisdictions), appeared to generate benthic impacts to >5 km from the discharge point (Olsgard and Gray, 1995). More recent evidence based on current drilling techniques suggests that the effects of produced water on benthic organisms will be limited to 1–2 km from the source (Bakke et al., 2013). Seafloor coverage of drill cuttings as low as 3 mm thickness can generate detectable impacts to the infauna (Schaaning et al., 2008). However, even beyond the area of observable cuttings piles, quantitative changes in meiofaunal abundance and community composition have been observed (Montagna and Harper, 1996; Netto et al., 2009). Changes in assemblage structure have also been observed beyond the areas of visually apparent seafloor disturbance as a result of increased scavenging and opportunistic feeding on dead animals (Jones et al., 2007; Hughes et al., 2010). Despite occasional observations of increased scavenger abundance in impacted areas, it has been suggested that the fauna of cuttings-contaminated sediments represent a reduced food resource for fish populations (e.g., smaller body size, loss of epifaunal species, shift from ophiuroids to polychaetes; Olsgard and Gray, 1995).

Cold-water corals (**Figure 5**) have been the focus of numerous impact studies. Discharges from typical operations have the potential to impact cold-water coral communities in deep waters through smothering and toxic effects (Lepland and Mortensen, 2008; Purser and Thomsen, 2012; Larsson et al., 2013). In laboratory studies, the reef-framework-forming stony coral *Lophelia pertusa* had significant polyp mortality following burial by 6.5 mm of drill cuttings, the maximum permissible under environmental risk assessment in Norway (Larsson and Purser, 2011). As a result, at the Morvin field in Norway, where drilling took place near a *Lophelia* reef, a novel cuttings-transport system was developed to discharge cuttings some 500 m from the well and down-current from the most significant coral reefs (Purser, 2015). The discharge location was determined to minimize impacts based on cuttings dispersion simulation modeling (Reed and Hetland, 2002). Subsequent monitoring at nine reefs between 100 m and 2 km from the discharge site suggested this mitigation measure appeared to have been generally successful. Although concentrations of drill cuttings >25 ppm were observed at several of the monitored reefs, no obvious visual impacts to the coral communities were reported (Purser, 2015). However, this concentration of drill cuttings had been shown to have a significant negative effect on *L. pertusa* growth in laboratory experiments (Larsson et al., 2013).

Impacts from oil and gas operations may be compounded in some settings by other anthropogenic disturbances, particularly as human impacts on the deep-sea environment continue to increase (e.g., Glover and Smith, 2003; Ramirez-Llodra et al., 2011; Kark et al., 2015). Climate and ocean change, including higher temperatures, expansion of oxygen minimum zones, and ocean acidification, will exacerbate the more direct impacts of the oil and gas industry through increased metabolic demand. Multiple stressors can operate as additive effects, synergistic effects, or antagonistic effects (Crain et al., 2008). While studies of the interactions between climate variables (temperature, oxygen, pH, CO_2) and drilling impacts are rare or non-existent, multiple stressors typically have antagonistic effects at the community level, but synergistic effects at the population level (Crain et al., 2008). At the most basic level, experimental work has shown that increased temperature generally increases the toxicity of petroleum hydrocarbons and other compounds (Cairns et al., 1975; Tatem et al., 1978), which suggests that the ecological impacts that have been recorded to date may expand in magnitude and distance as climate change proceeds.

Deep-water fisheries have a significant impact on deep-sea species, with detrimental effects extending to habitats and ecosystems beyond the target populations (Benn et al., 2010; Clark et al., 2016). Some authors note that the physical presence of oil and gas infrastructure may protect fished species or habitats by *de facto* creating fisheries exclusion zones (Hall, 2001; Love et al., 2006), by establishing new reef habitat (sensu Montagna et al., 2002), and by functioning as fish aggregating devices (Hinck et al., 2004). Although the value of oil and gas infrastructure in secondary production and fisheries, particularly in deep waters, is controversial (Bohnsack, 1989; Baine, 2002; Ponti, 2002; Powers et al., 2003; Fabi et al., 2004; Kaiser and Pulsipher, 2006), there is some evidence to suggest that this can occur (Claisse et al., 2015). Oil industry infrastructure may therefore have some positive effects, even in deep water (Macreadie et al., 2011), principally in terms of creating refugia from fishing impacts (e.g., Wilson et al., 2002).

TABLE 3 | Examples of the detected spatial extent ("sphere of influence") and likely recovery in the benthos attributed to spatial proximity to offshore oil and gas drilling operations on the seafloor.

Location/Site	Depth	Fauna group	Drilling fluid/Mud type[b]	Main biological metrics[a]	Spatial footprint in the benthos[c]	Recovery estimate(s)	Source
Orinoco Fan off Venezuela (1N, 60W)	543 m	Megafauna (epibenthic; >5 cm)	No direct fluid discharge	SPP, DENS, COMP	ca. 20–50 m	na	Jones et al., 2012b
Faroe–Shetland Channel (61N, 3E)	600 m	Megafauna (epibenthic; >5 cm)	?	SPP, DENS, COMP	ca. <50 m	>3–10 years for localized effects	Jones et al., 2012a
North Sea (Norwegian Sector) (58N, 2E)	114 m	Megafauna (epibenthic; >5 cm)	?	DENS	50–100 m	na	Hughes et al., 2010
Norwegian Sea (65N, 6E)	380 m	Megafauna (epibenthic; >5 cm)	WBM	SPP, DENS, COMP	ca. <100 m	>3 years	Gates and Jones, 2012
Faroe–Shetland Channel (61N, 3E)	420–509 m	Megafauna (epibenthic; >5 cm	?	SPP, DENS, COMP	ca. 100–150 m	na	Jones et al., 2007
Faroe–Shetland Channel (61N, 3E)	600 m	Megafauna (epibenthic; >5 cm	WBM	SPP, DENS, COMP	ca. 100–150 m	na	Jones et al., 2006
Australia, Bass Strait (38S, 142E)	60 m	Macrobenthos (1 mm mesh retained)	WBM	SPP, DENS, COMP	ca. 100–200 m	>11 months (composition	Currie and Isaacs, 2005
Gulf of Mexico (28N, 96W)	29–129 m	Meiofauna	WBM	SPP, DENS, COMP	ca. 100–200 m	na	Montagna and Harper, 1996
Georges Bank, NE Atlantic (41N, 69W)	80–140 m	Macrobenthos (1 mm mesh retained)	NAF (?)	SPP, DENS, COMP	ca. 200 m (?)	na	Neff et al., 1989
North Sea (Dutch Sector) (53 N, 3E)	35 m	Macrobenthos (1 mm mesh retained)	NAF-WBM	SPP, DENS	ca. 25–500 m	>8 years	Daan and Mulder, 1996
Brazil, Campos Basin (21S, 40W)	890 m	Meiofauna	SBM	SPP, DENS	ca. 500 m	~1 year	Netto et al., 2009
Brazil, Campos Basin (21S, 40W)	902 m	Macrobenthos (0.5 mm mesh retained)	WBM-NAF	SPP, DENS, COMP	ca. 500 m	>1 year (composition)	Santos et al., 2009
North Sea (61N, 2E)	120 m	Macrobenthos (1 mm mesh retained)	NAF	SPP	ca. 250–3000 m	na	Davies et al., 1984
Norwegian Shelf (60 N, 4 E)	63–380 m	Macrobenthos (1 mm mesh retained)	NAF	COMP	ca. 500–6000	na	Olsgard and Gray, 1995
Campeche Bank and Bay (20N, 92W)	12–135 m	Macrobenthos (0.5–2.0 mm mesh)	?	COMP	ca. 8000 m	na	Hernandez Arana et al., 2005
North Ionian Sea (39N, 17E)	90 m	Meiofauna	?	SPP, DENS, COMP	1000 m	na	Terlizzi et al., 2008; Fraschetti et al., 2016
Brazil, Campos Basin (21S, 40W)	215 m	Meiofauna	WBM-NAF	COMP	na	>22 months	Netto et al., 2010
Brazil, Campos Basin (21S, 40W)	170–270 m	Macrobenthos (0.5 mm mesh retained)	NAF	COMP	na	~22 months	Santos et al., 2010

[a]*SPP, Number of species or similar diversity metric; DENS, density of individuals, often at the level of community-wide abundance; COMP, composition or structure of the assemblage.*
[b]*WBM, Water-based muds; NAF, Non-aqueous fluids (Neff et al., 2000; Neff, 2005; Bakke et al., 2013).*
[c]*Reported estimates of the distance that biological effects extend outwards from drill holes or platforms have considerable uncertainty attached, largely owing to the possibility of more subtle effects not being detected, limited spatial coverage of past sampling, or the small number of reference sites in some studies. Tabulated values therefore represent conservative estimates based on currently available data, but should not be taken as implying the absence of larger-scale contamination or biological responses, that may or may not be chronic, attributable to oil and gas production in the sea.*
"?" indicates that the type of drilling mud is unknown.

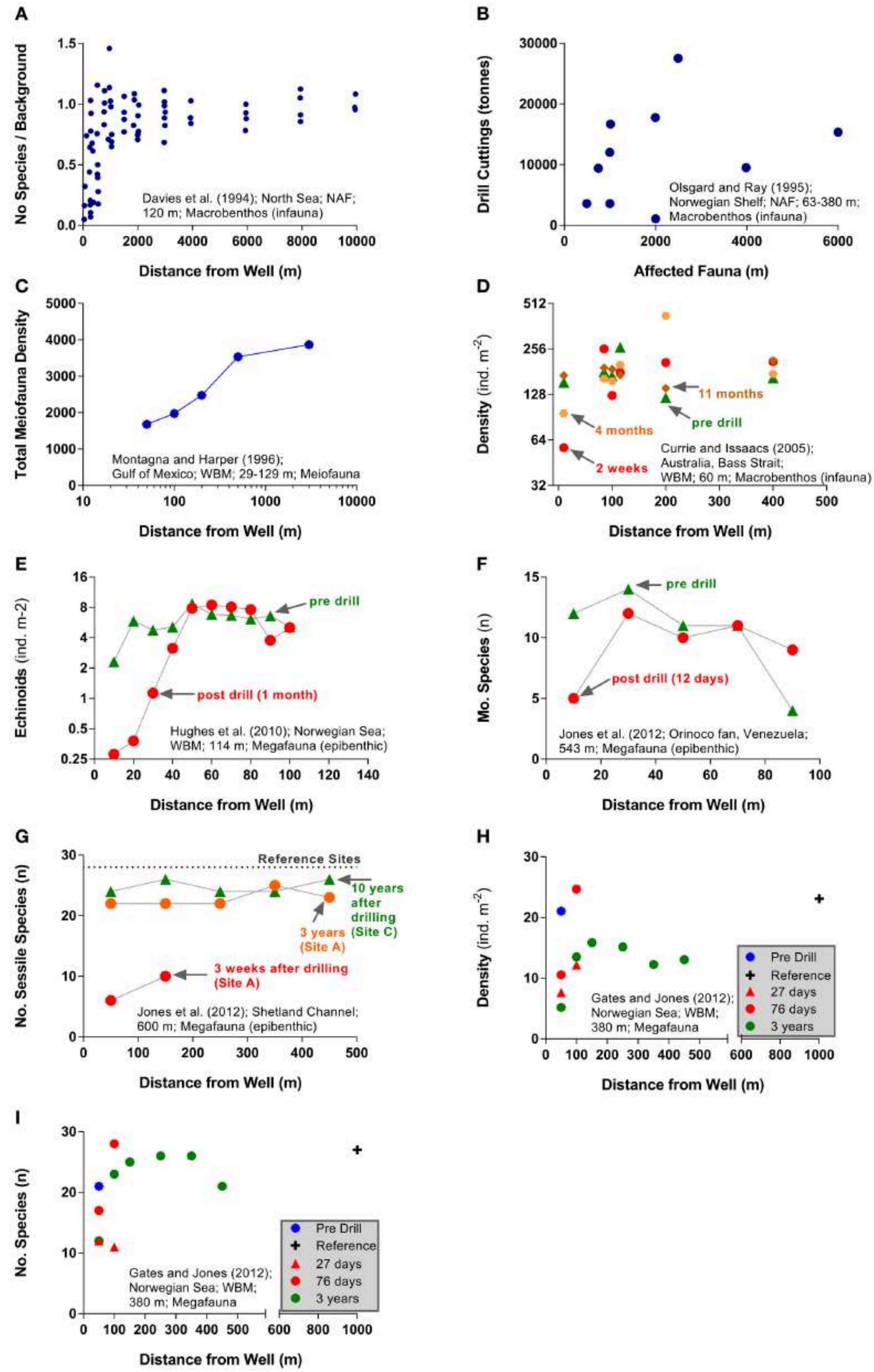

FIGURE 4 | Illustrative examples of spatial patterns in the benthos associated with exploratory and routine drilling operations (i.e., excluding large accidental spills; see Table 3 for additional information on graphed studies). Note that impacts in **(A,B)** are from oil-based drilling muds, and impacts in **(F)** are from a site where no drilling lubricant was used, while the rest of the studies **(C–E,G–I)** were from sites using water-based muds.

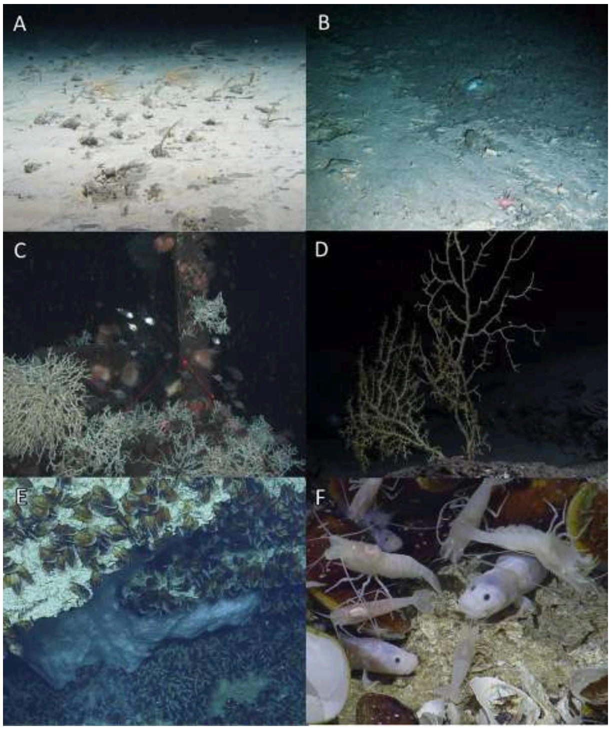

FIGURE 5 | Deep-sea communities near drilling activities. (A) Benthic communities shortly after smothering by (light colored) cuttings at the Tornado Field (1050 m depth), Faroe-Shetland Channel, UK. **(B)** Edge of cuttings pile at the Laggan field, Faroe-Shetland Channel, UK (Figure 4D from Jones et al., 2012a). **(C)** Atlantic roughy, Hoplostethus occidentalis, among *L. pertusa* around the abandoned test-pile near Zinc at 450 m depth in the Gulf of Mexico. Image courtesy of the Lophelia II program, US Bureau of Ocean Energy and Management and NOAA Office of Ocean Exploraiton and Research. **(D)** Appearance in 2013 of a *Paramuricea biscaya* colony damaged during the Deepwater Horizon oil spill in 2010. Image courtesy of ECOGIG, a GoMRI-funded research consortium and the Ocean Exploration Trust. **(E,F)**: Methane-seep communities from an area within the exclusive economic zone of Trinidad and Tobago that is targeted for future oil and gas development. The Ocean Exploration Trust is acknowledged for use of these photos from the E/V *Nautilus* 2014 Expedition.

Oil-field infrastructure can also provide hard substratum for colonization by benthic invertebrates, including scleractinian corals and octocorals (Hall, 2001; Sammarco et al., 2004; Gass and Roberts, 2006; Larcom et al., 2014). The widely-distributed coral *L. pertusa* (**Figure 5**) has been recorded on numerous oil field structures in the northern North Sea (Bell and Smith, 1999; Gass and Roberts, 2006), as well as on infrastructure in the Faroe-Shetland Channel (Hughes, 2011), and the northern Gulf of Mexico (Larcom et al., 2014). These man-made structures may enhance population connectivity (Atchison et al., 2008) and provide stepping stones for both native and potentially invasive species, which has been demonstrated for shallow-water species that may not normally be able to disperse across large expanses of open water (Page et al., 2006; Coutts and Dodgshun, 2007; Sheehy and Vik, 2010). Therefore, the increased connectivity provided by these artificial structures may be viewed both positively and negatively, and it is difficult to make predictions about the potential benefits or harm of the increased availability of deep-sea hard substrata.

EFFECTS OF ACCIDENTAL DISCHARGES

Oil and gas operations have the potential to result in accidental releases of hydrocarbons, with the likelihood of an accidental spill or blowout increasing with the depth of the operations (Muehlenbachs et al., 2013). The U.S. NOAA Office of Response and Restoration records, on average, 1–3 spills per week within the US EEZ, but most of these are relatively small and occur near the shore. On the U.S. outer continental shelf between 1971 and 2010, there were 23 large spills of more than 1000 barrels (160,000 L) of oil, or an average of one every 21 months (Anderson et al., 2012). In addition, on a global scale there were 166 spills over 1000 barrels that occurred during offshore transport of oil in the period between 1974 and 2008, or one every 2.5 months (Anderson et al., 2012). The greatest risk to the marine environment comes from an uncontrolled release of hydrocarbons from the reservoir, known as a blowout (Johansen et al., 2003). Risk modeling suggests that an event the size of the Deepwater Horizon incident can be broadly predicted to occur on an interval between 8 and 91 years, or a rough average of once every 17 years (Eckle et al., 2012). Several major offshore oil blowouts have occurred, including the IXTOC-1 well in the Bahia de Campeche, Mexico where 3.5 million barrels of oil were released at a water depth of 50 m over 9 months (Jernelov and Linden, 1981; Sun et al., 2015) and the Ekofisk blowout where 200,000 barrels (32 million liters) of oil were released at a water depth of 70 m (Law, 1978). While all of these examples represent accidental discharges, the frequency at which they occur in offshore waters suggests that they can be expected during "typical" operations.

The best-studied example of a major deep-sea blowout was at the Macondo well in the Gulf of Mexico in 2010 (Joye et al., 2016). This blowout discharged ∼5 million barrels (800 million liters) of oil at a water depth of ∼1500 m (McNutt et al., 2012). About half of the oil traveled up to the surface, while the rest of the gaseous hydrocarbons and oil suspended as microdroplets remained in a subsurface plume centered around 1100 m depth, that traveled ∼50 km from the well-head (Camilli et al., 2010). The surface oil slicks interacted with planktonic communities and mineral particles to form an emulsion of oiled marine snow (Passow et al., 2012). This material was subsequently observed as a deposited layer on the deep-sea floor that was detected in an area of ∼3200 km^2 (Chanton et al., 2014; Valentine et al., 2014). Impacts at the seabed, as revealed by elevated hydrocarbon concentrations and changes to the nematode-copepod ratio, were detected in an area of over 300 km^2, with patchy impacts observed to a radius of 45 km from the well site (Montagna et al., 2013; Baguley et al., 2015). This oiled marine snow was also implicated in impacts on mesophotic and deep-sea coral communities (White et al., 2012; Silva et al., 2015; **Figure 5**).

Deep-sea coral communities were contaminated by a layer of flocculent material that included oil fingerprinted to the Macondo well, and constituents of the chemical dispersant used in the response effort (White et al., 2012, 2014). Impacts on corals were detected at a number of sites, extending to 22 km from the well, and to water depths (1950 m) exceeding that of the well-head (Hsing et al., 2013; Fisher et al., 2014a). The severity of impact on the coral colonies appeared to be related to distance from the well, with >50% of the corals exhibiting >10% colony damage closer to the well, and less-extensive patchy damage recorded at the more distant sites (Fisher et al., 2014a). Elevated hydrocarbon concentrations and changes to infaunal communities were reported from sediment samples taken adjacent to the impacted coral sites (Fisher et al., 2014b).

Dispersants or chemical emulsifiers are applied to oil spills in an effort to disperse surface slicks. Globally, there have been over 200 documented instances of dispersant use between 1968 and 2007 (Steen, 2008). Dispersant applications typically are successful in dispersing large oil aggregations, although their effectiveness varies with oil composition, mixing dynamics, temperature, salinity, and the presence of light (Weaver, 2004; Henry, 2005; NRC, 2005; Chandrasekar et al., 2006; Kuhl et al., 2013). However, the use of dispersants creates two additional impacts: (i) a toxic effects from the dispersant itself, and (ii) a broader and/or more rapid contamination of the environment as a result of the dispersal of hydrocarbons.

Dispersant use can cause increases in environmental hydrocarbon concentrations (Pace et al., 1995) and direct toxic effects (Epstein et al., 2000). Dispersants increase the surface area for oil-water interactions (Pace et al., 1995), ostensibly increasing the biological availability of oil compounds (Couillard et al., 2005; Schein et al., 2009), potentially enhancing toxic effects (Chandrasekar et al., 2006; Goodbody-Gringley et al., 2013; DeLeo et al., 2016). However, in the case of the Deepwater Horizon accident, dispersant use was shown to impede hydrocarbon degradation by microorganisms (Kleindienst et al., 2015). Chemically-dispersed oil is known to reduce larval settlement, cause abnormal development, and produce tissue degeneration in sessile invertebrates (Epstein et al., 2000; Goodbody-Gringley et al., 2013; DeLeo et al., 2016). Dispersant exposure alone has proved toxic to shallow-water coral larvae (Goodbody-Gringley et al., 2013) and deep-sea octocorals (DeLeo et al., 2016). Some of the potentially toxic components of dispersants may persist in the marine environment for years (White et al., 2014), but there are few *in situ* or even *ex situ* studies of effects of dispersants on deep-sea organisms.

RECOVERY FROM IMPACTS

Typical impacts from drilling may persist over long time scales (years to decades) in the deep sea (**Table 3**). In deep waters, the generally low-energy hydrodynamic regime may lead to long-term persistence of discharged material, whether it be intentional or accidental (Neff, 2002; Chanton et al., 2014). Sediment contamination by hydrocarbons, particularly PAHs, is of particular concern, as these compounds can persist for decades, posing significant risk of prolonged ecotoxicological effects. Hydrocarbons from the Prestige spill, off the Galician coast, were still present in intertidal sediments 10 years post-spill (Bernabeu et al., 2013), and petroleum residues from the oil barge *Florida* were still detectable in salt marsh sediments in West Falmouth, MA, after 30 years (Reddy et al., 2002). In the Norwegian Sea (380 m depth), there was a reduction in the visible footprint of drill cuttings from a radius of over 50 m to ∼20 m over 3 years, but chemical contamination persisted over the larger area (Gates and Jones, 2012). In the Faroe-Shetland Channel (500–600 m), visible drill cuttings reduced from a radius of over 85–35 m over a 3-year period, while an adjacent 10 year-old well-site exhibited visually distinct cuttings piles at a radius of only 15–20 m (Jones et al., 2012a). Recovery of benthic habitats may take longer at sites where bottom water movements limit dispersal of cuttings (Breuer et al., 2004).

Much of the deep-sea floor is characterized by comparatively low temperatures and low food supply rates. Consequently, deep-sea communities and individuals generally exhibit a slower pace of life than their shallow-water counterparts (reviewed in Gage and Tyler, 1991; McClain and Schlacher, 2015). Deep-water corals and cold-seep communities (**Figure 5**) represent anomalous high-biomass ecosystems in the deep sea and frequently occur in areas of economic interest because of their direct (energy and carbon source) or indirect (substratum in the form of authigenic carbonate) association with oil and/or gas-rich fluids (Masson et al., 2003; Coleman et al., 2005; Schroeder et al., 2005; Cordes et al., 2008; Bernardino et al., 2012; Jones et al., 2014). Cold-seep tubeworms and deep-water corals exhibit slow growth and some of the greatest longevities among marine metazoans, typically decades to hundreds of years, but occasionally to thousands of years (Fisher et al., 1997; Bergquist et al., 2000; Andrews et al., 2002; Roark et al., 2006; Cordes et al., 2007; Watling et al., 2011). Recruitment and colonization dynamics are not well-understood for these assemblages, but recruitment appears to be slow and episodic in cold-seep tubeworms (Cordes et al., 2003), mussels (Arellano and Young, 2009), and deep-sea corals (Thresher et al., 2011; Lacharité and Metaxas, 2013; Doughty et al., 2014).

Because of the combination of slow growth, long life spans and variable recruitment, recovery from impacts can be prolonged. Based on presumed slow recolonization rates of uncontaminated deep-sea sediments (Grassle, 1977), low environmental temperatures, and consequently reduced metabolic rates (Baguley et al., 2008; Rowe and Kennicutt, 2008), Montagna et al. (2013) suggested recovery of the soft-sediment benthos from the *Deepwater Horizon* well blowout might take decades. For deep-sea corals, recovery time estimates are on the order of centuries to millennia (Fisher et al., 2014b). However, in some cases re-colonization may be relatively rapid, for example, significant macrofaunal recruitment on cuttings piles after 6 months (Trannum et al., 2011; **Table 3**). Altered benthic species composition may, nevertheless, persist for years to decades (Netto et al., 2009). Direct studies of recovery from drilling in deep water are lacking and the cumulative effects of multiple drilling wells are not well-studied.

ENVIRONMENTAL MANAGEMENT APPROACHES

Environmental management takes many forms. We focus on management activities that mitigate the adverse environmental effects of oil and gas development, specifically addressing avoidance- and minimization-type approaches (World Bank, 2012). Here, we consider three complementary strategies: (i) activity management, (ii) temporal management, and (iii) spatial management (**Table 1**).

Activity Management

In activity management, certain practices (or discharges) are restricted or banned, or certain technologies are employed to reduce the environmental impact of operations. An example of activity management is the phasing out of drilling muds that used diesel oil as their base. These drilling fluids biodegrade very slowly, have a high toxicity, and exposure to them can result in negative environmental consequences (Davies et al., 1989). In addition, many countries have introduced restrictions on the discharge of lower-toxicity organic-phase drilling muds (i.e., oil-based muds containing mineral oil or synthetic liquids) and untreated cuttings contaminated with these fluids. For example, the OSPAR Convention prohibits Contracting Parties from discharging whole organic-phase fluids and cuttings containing organic-phase muds of more than 1% by weight on dry cuttings (OSPAR Commission, 2000), and permits are typically required for the use, reinjection and discharge of chemicals including drilling muds and cuttings containing hydrocarbons from the reservoir. The elimination of these discharges has led to demonstrably reduced extents of drilling impacts (**Figure 4**), from thousands of meters around wells drilled using oil-based muds (Davies et al., 1984; Mair et al., 1987; Gray et al., 1990; Kröncke et al., 1992) to hundreds of meters for wells drilled using water-based muds (Jones et al., 2006; Gates and Jones, 2012). Restrictions are also imposed on the discharge of produced water, with produced water typically being expected to be re-injected into subsurface formations, or to be cleaned to meet national oil-in-produced water discharge limits before being disposed into the sea (Ahmadun et al., 2009).

During exploration activities, activity management may be required for seismic surveys, because the intense acoustic energy can cause ecological impacts particularly to marine mammals. In many countries, including the US, UK, Brazil, Canada, and Australia, mitigation protocols have been developed to reduce the risk of adverse impacts on marine mammals (Compton et al., 2008).These include "soft-start" or "ramp-up" rules that require air gun power to be slowly increased to allow marine mammals to vacate the area before the full power is reached, and the need for trained Marine Mammal Observers to monitor an exclusion zone around the sound source and to delay or stop operations should any marine mammals be observed within a predefined safety zone (Compton et al., 2008).

Activity management may also be applied to oil and gas industry decommissioning. In European waters, for example, OSPAR has prohibited the dumping or leaving in place of disused infrastructure (OSPAR Decision 98/3, 1998). Although some large installations are exempt, most structures must be taken onshore for disposal; however the environmental impacts caused by removing these large structures may outweigh any negative effects of leaving them in place. In many other jurisdictions, such as the US, Malaysia, Japan, and Brunei, decommissioned structures may be left in place as artificial reefs (Fjellsa, 1995; Kaiser and Pulsipher, 2005). Since 1986, the US Department of the Interior has approved over 400 "Rigs-to-Reefs" proposals (Bureau of Safety and Environmental Enforcement). To date, these rig-to-reef proposals are limited to shallow waters, where they are thought to create habitat for commercial and recreational fisheries species.

Temporal Management

Temporal management of oil and gas activities is not yet widely applied in deep-water settings. Temporal management approaches are intended to reduce impacts on the breeding, feeding, or migration of fish, marine mammals, and seabirds. Furthermore, seismic operations along marine mammal migration routes or within known feeding or breeding grounds may be restricted during aggregation or migration periods in order to reduce the probability of marine mammals being present in the area during the survey (Compton et al., 2008). In addition, soft-start procedures may only be allowed to commence during daylight hours and periods of good visibility to ensure observers can monitor the area around the air gun array and delay or stop seismic operations if necessary (Compton et al., 2008). In Norway, seismic surveys cannot commence if marine mammals or turtles are present in the immediate area and monitoring is carried out by trained observers, whose presence is required on all deep-water (>200 m depth) seismic surveys.

Temporal management has also been proposed for the cold-water coral *L. pertusa* in Norway (Norsk Olje og Gass, 2013). In the NE Atlantic, this species appears to spawn mainly between January and March (Brooke and Jarnegren, 2013) and the larvae are thought to be highly sensitive to elevated suspended sediment loads, including drill cuttings (Larsson et al., 2013; Jarnegren et al., 2016). Recommendations are to delay drilling activities near *Lophelia* reefs during main spawning periods of the corals or other ecologically and/or economically important species. Special steps to strengthen the oil spill emergency response system, including shorter response times during the spawning season have also been implemented.

Spatial Management

Spatial management prohibits particular activities from certain areas, for example where sensitive species or habitats are present. This can range from implementing exclusion zones around sensitive areas potentially affected by individual oil and gas operations to establishing formal marine protected areas through legislative processes where human activities deemed to cause environmental harm are prohibited. The use of EIAs as a tool for identifying local spatial restrictions for deep-water oil and gas operations is widely applied, and specific no-drilling zones (mitigation areas) are defined by the regulatory authority around sensitive areas known or occurring with high-probability (**Table 1**). The need for spatial restrictions to hydrocarbon development may also be identified at the strategic planning stage. In Norway, for example, regional multi-sector assessments have been undertaken to examine the environmental and socio-economic impacts of various offshore sectors and to develop a set of integrated management plans for Norway's maritime areas. The plans incorporate information on potential cumulative effects from multiple sectors, potential user conflicts and key knowledge gaps, as well as locations that should be exempt from future hydrocarbon exploration owing to their ecological value and sensitivity to potential effects from offshore drilling (Fidler and Noble, 2012; Olsen et al., 2016).

A number of approaches have been used to identify the ecological features and attributes used in setting targets for spatial management, some of which may be relevant in the deep-sea environment. For example, the term "vulnerable marine ecosystem" (VME) is commonly used in fisheries management and is defined as an ecosystem that is easily damaged as a result of its physical and/or functional fragility (e.g., Ardron et al., 2014). The VME concept was conceived under the auspices of the United Nations Food and Agricultural Organisation (FAO, 2009) to assist in the assessment and control of the impacts of demersal fisheries in areas beyond national jurisdiction (the "Area" or the 'High Seas'). Cold-seep and deep-water coral ecosystems (**Figure 5**) would be considered as VMEs under this framework. However, given that the deep-water oil and gas industry still operates, almost exclusively, within areas of national jurisdiction, and has impacts that differ in extent and character to bottom-contact fishing, the VME concept may not be the most appropriate.

A potentially more relevant framework for determining deep-water habitats to be protected is that of the "ecologically or biologically significant area" (EBSA) developed under the United Nations Convention on Biological Diversity (CBD; see e.g., Dunn et al., 2014; note that the US is not a signatory to the CBD). EBSAs are thought of as "discrete areas, which through scientific criteria, have been identified as important for the health and functioning of our oceans and the services that they provide" (UNEP-WCMC, 2014). Such criteria include: uniqueness or rarity; special importance for life-history stages of species; importance for threatened, endangered or declining species and/or habitats; vulnerability, fragility, sensitivity, or slow recovery; biological productivity; biological diversity; and naturalness. These criteria synthesize well-established regional and international guidelines for spatial planning (Dunn et al., 2014), and therefore should be highly relevant for future spatial planning in the oil and gas industry (Clark et al., 2014). Regional cooperation is encouraged in the spatial management of EBSAs, including identifying and adopting appropriate conservation measures and sustainable use, and establishing representative networks of marine protected areas (Dunn et al., 2014).

Deep-sea habitats that would be considered as VMEs and would also fit many of the EBSA criteria include cold-seep and deep-water coral communities. Both habitats are of particular significance for the management of deep-water oil and gas activities because they frequently occur in areas of oil and gas interest (**Figure 5**). These habitats attract conservation attention because they are localized (*sensu* Bergquist et al., 2003), structurally complex (Bergquist et al., 2003; Cordes et al., 2008), and contain high primary (seeps) and secondary (corals) productivity, relatively high biomass, and large-sized organisms (Sibuet and Olu, 1998; Bergquist et al., 2003; Cordes et al., 2003). The foundation species in these communities are very long-lived, even compared to other deep-sea fauna (McClain et al., 2012), and support a diverse community including some endemic species (Cordes et al., 2009; Quattrini et al., 2012). The infaunal and mobile fauna that live on the periphery of these sites are also distinct from the fauna in the background deep sea, both in terms of diversity and abundance (Demopoulos et al., 2010), and also deserve consideration for protection (Levin et al., 2016).

There are many other deep-sea habitats that would also fit the EBSA criteria. These are typically biogenic habitats, where one or several key species (ecosystem engineers) create habitat for other species. Examples of these include sponges (Klitgaard and Tendal, 2004), xenophyophores (Levin, 1991), tube-forming protists (De Leo et al., 2010), and deposit feeders that create complex burrow networks (Levin et al., 1997). Furthermore, areas of brine seepage, particularly brine basins, may not contain abundant hard substrata, but still support distinct and diverse microbial communities, as well as megafaunal communities (e.g., glass sponge gardens in the Orca Basin, Shokes et al., 1977).

For spatial management of these sensitive areas to be effective, information on the spatial distribution of features of conservation interest is essential. Mapping these features can be particularly challenging in the deep sea, but advances in technology are improving our ability to identify and locate them (e.g., multibeam swath bathymetry, sidescan sonar, seismic survey). Even modest occurrences of deep-water corals can be mapped by both low and high frequency sidescan sonar in settings with relatively low background topography (e.g., Masson et al., 2003). Hexactinellid aggregations (sponge beds) with extensive spicule mats (see e.g., Bett and Rice, 1992) may also have sufficient acoustic signature to be detectable. In some cases, seep environments can also be detected via water-column bubble plumes or surface ocean slicks (Ziervogel et al., 2014; MacDonald et al., 2015).

In the absence of direct seabed mapping, habitat suitability models have been used in attempts to predict the occurrence of species/habitats of interest. These often involve the combination of point observations and oceanographic/environmental data in a geographical context (Bryan and Metaxas, 2007; Tittensor et al., 2009; Howell et al., 2011; Georgian et al., 2014). Relevant oceanographic and environmental datasets can be obtained from local field measurements, global satellite measurements, and compilations from world ocean datasets (Georgian et al., 2014; Guinotte and Davies, 2014; Rengstorf et al., 2014; Vierod et al., 2014). Point source biological observations are best determined from direct seabed sampling and visual observation (Georgian et al., 2014; Rengstorf et al., 2014). Additional data can be derived from historical data (e.g., museums and biogeographic databases such as OBIS and GBIF) or bycatch from trawl fisheries (Ardron et al., 2014). However, these data must be interpreted with caution as they may include dead and possibly displaced organisms (i.e., coral skeletons), and the location information can be imprecise if it is based on the mid-points of trawl locations or from older records before twenty-first century improvements in global and seafloor positioning systems technology.

In most cases, implementation of spatial restrictions depends on positive confirmation of the feature/species/habitats of interest. This is often best achieved via visual imaging surveys (towed camera, autonomous underwater vehicles, ROVs, manned submersible), which are typically non-destructive and provide valuable data on both biological and environmental characteristics (Georgian et al., 2014; Morris et al., 2014; Rengstorf et al., 2014; Williams et al., 2015). Collection of reference physical specimens is also highly desirable in providing accurate taxonomic identifications of key taxa (Bullimore et al., 2013; Henry and Roberts, 2014; Howell et al., 2014), and may provide additional relevant data (e.g., life cycles, reproductive strategies, population connectivity). Together, mapping through remote sensing, habitat suitability models, and ground-truthing by seafloor observations and collections provide adequate maps of ecological features to better inform the trade-offs between conservation and economic interests in advance of exploration or extraction activities (Mariano and La Rovere, 2007).

Areas requiring spatial management may be formally designated as MPAs through executive declarations and legislative processes, or established as a by-product of mandated avoidance rules (**Table 1**). In the UK, these come in the form of Designations as Special Areas of Conservation, Nature Conservation Marine Protected Areas, or Marine Conservation Zones. In the US, these are in the form of National Monuments (Presidential executive order), National Marine Sanctuaries (congressional designation), fisheries management areas such as Habitat Areas of Particular Concern, or, in the case of the oil and gas industry, through Notices to Lessees issued by the U.S. Bureau of Ocean Energy Management (BOEM). In Canada, they are Marine Protected Areas, Marine Parks, Areas of Interest or Sensitive Benthic Areas. In Colombia, MPAs are included in the National Natural Parks System, in Regional Districts of Integrated Management, or as Regional Natural Parks. In many jurisdictions, systems of MPAs are still under development, and oil and gas exploration and development is permitted within these areas. It remains uncommon for setback distances or buffer zone requirements to be specified.

The formal designation process for MPAs varies greatly among EEZs. Fundamentally, a firm, widespread systematic conservation plan (sensu Margules and Pressey, 2000) in the deep sea will be critical in creating MPAs that are representative and effective (Kark et al., 2015). MPAs can be large "no-go" areas that comprise a broad set of representative habitat types. They can also be networks of smaller areas that may serve as stepping stones across the seascape. There have been numerous reviews of the theory behind these various designs (e.g., Hyrenbach et al., 2000; Botsford et al., 2003; Klein et al., 2008), and future work including scientists, managers, industry representatives, and other stakeholders, will be needed to arrive at the most effective scenarios that can be used both as general recommendations and on a case-by-case basis.

Even when the formal MPA designation process is followed, oil and gas industrial activity may still be permissible, although their proximity typically triggers additional scrutiny of development plans (**Table 1**). Examples of wells that have been drilled near some important marine protected areas include the Palta-1 well off the Ningaloo reef in Australia and drilling and production in the Flower Gardens National Marine Sanctuary in the U.S. Gulf of Mexico. There are also examples of marine protected areas that have been designated in regions already supporting active oil production and / or exploration (e.g., Quad 204 development in the Faroe-Shetland Channel Sponge Belt, Nature Conservation MPA).

In some cases, MPAs may not be formally declared, but sensitive habitats are explicitly avoided during field operations as part of the lease conditions. For example, in Norway, exploration drilling has occurred near the Pockmark-reefs in the Kristin oil field and the reefs of the Morvin oil field (Ofstad et al., 2000). Direct physical damage was limited by ensuring the well location and anchoring points (including chains) were not near the known coral locations. Similarly, in Brazil, impacts to deep-water corals must be avoided, and ROV surveys of proposed tracklines for anchors are typically conducted before or after installation.

Despite the requirements of many jurisdictions to avoid deep-water petroleum activities near sensitive habitats, it remains uncommon for legally mandated setback distances or buffer zone requirements to be specified. For example, there are no mandated separation distances of industry infrastructure and deep-water corals for both the Brazilian and Norwegian case studies, rather the need for spatial restrictions is evaluated on a case-by-case basis as part of the environmental impact assessment process.

Some exceptions exist, such as activities within the US EEZ, where restriction zones for oil and gas industry activities that could damage "high-density" deep-water benthic communities have been established. BOEM has taken a precautionary approach and defined mitigation areas in which oil and gas activity is prohibited. These areas are determined from interpretation of seismic survey data. Previous studies have demonstrated that these seismic data can reliably predict the presence of chemosynthetic and deep-water coral communities (Roberts et al., 2000, 2010), and can explain over 40% of the variability in *L. pertusa* distribution in the northern Gulf of Mexico (Georgian et al., 2014).

Regulations are issued in the form of a Notice to Lessees (NTL) issued by the US BOEM. The NTL for high-density deep-water (>300 m water depth) benthic communities (NTL 2009-G40) stipulates that operators have to submit maps depicting bathymetry, seafloor and shallow geological features, and potential biological areas that could be disturbed by the proposed activities, including those located outside of the operator's lease. ROV surveys of the tracklines of anchors are typically conducted, but can occur after the installation of the infrastructure if the plan is approved. However, if the well is drilled near a known high-density community or archeological site, then visual surveys are mandatory prior to installation. If the ROV surveys reveal high-density chemosynthetic or coral communities, the operator is required to report their occurrence and submit copies of the images to BOEM for review. Avoidance measures have to be undertaken for all potential and known high-density benthic communities identified during these assessments.

Beyond the borders of the BOEM mitigation areas, there are mandated set-back distances for oil and gas infrastructure in US territorial waters. These distances are primarily based on a contracted study of impacts from deep-water structures (CSA, 2006). The set-back distance for sea-surface discharges of drilling muds and cuttings was originally 305 m, corresponding to the average distance over which impacts were detected in the CSA

(2006) study. Following more recent discoveries of abundant deep-water coral communities in and near the hard-ground sites within the mitigation areas, the set-back distance was doubled to 610 m (2000 feet). The set-back distance for the placement of anchors and other seafloor infrastructure is 150 m (500 feet) from the mitigation areas, but this may be reduced to 75 m (250 feet) if a waiver is requested.

In addition to specific targets for avoidance or establishment of protected areas, the use of reference areas can also assist in spatial management, and in the testing of EIA predictions more generally. For example, Norwegian protocols require the establishment and monitoring of regional reference sites, representative of "normal" benthic conditions. Comparison of reference sites with those proximal to industry operations allows the effects of drilling and routine operations to be assessed, properly attribute any changes in the ecological communities, and further inform spatial management practice (Iversen et al., 2011). Some real-time monitoring and responsive action has also been undertaken in the benthic environment. In Norway, Statoil has monitored the potential impacts on a coral reef system at the Morvin oil field, which included sediment sampling, video observations, sensors and sediment traps (Tenningen et al., 2010; Godø et al., 2014). The sensor data were available in real time and enabled drillers to observe if selected reef sites were being impacted by drilling activities. Regardless of the structure of the monitoring program, some periodic post-development assessments, both within the development area and in appropriate reference areas, are required to evaluate the efficacy of the implemented protections.

CONCLUSIONS AND RECOMMENDATIONS

Deep-sea species, assemblages, and ecosystems have a set of biological and ecological attributes (e.g., life-history traits, spatial distribution, dispersal, and recruitment) that generally confer low resilience and recovery potential from anthropogenic disturbances, including those associated with the deep-water oil and gas industry. In general, deep-sea organisms are slower growing and more long lived than their shallow-water counterparts and their distributions, abundance, and species identity remain largely unknown at most locations. The combination of their sensitivity to disturbance and the direct threat posed by industrial activity (of any kind) should stipulate a precautionary approach to the management of deep-sea resources.

A comprehensive management plan requires accurate environmental maps of deep-sea oil and gas production areas. These maps could be more effectively generated by creating a central archive of industry-generated acoustic remote sensing data, including seismic data and bathymetry, and making these data available to managers and scientists via open-access platforms. Predictive habitat modeling can also contribute to the development of distribution maps for specific taxa. In addition, maps need ground-truthing: broad-scale baseline environmental data (biological/physical/chemical) that are acquired over a large area are required to place all EIAs in context, with continued monitoring necessary to test their predictions and account for changing baselines. Baseline surveys should be carried out first at a regional level if no historical data are available. Prior to industrial activity, comprehensive surveys should be carried out within the planning area (including along pipeline tracks) and in a comparable reference area outside of the influence of typical impacts (at least 4–5 km). Ideally, surveys should include high-resolution mapping, seafloor imagery surveys, and physical samples to characterize the faunal community and ensure proper species identifications, which should consist of a combination of classical and molecular taxonomy. We also recommend the inclusion of newer high-throughput sequencing and metabarcoding techniques for a robust assessment of biodiversity at all size classes (Pawlowski et al., 2014; Lanzen et al., 2016). International collaboration with the oil and gas industry to develop and conduct basic scientific research should be further strengthened to obtain the baseline information required for a robust understanding of the ecology of these systems and the interpretation of monitoring results, both at local and regional scales.

We recommend that representatives of all habitat types, ideally based on a strategic regional assessment, should be granted protection. Any high-density, high-biomass, high-relief, or specialized (i.e., chemosynthetic) deep-sea habitat should be identified and mapped and avoidance rules or formal MPA designations implemented to minimize adverse impacts. The definition of these significant communities will vary from region to region and will depend on national or international regulations within the region of interest, but the EBSA concept should be generally applicable. Given the likely proximity of sensitive habitats to oil and gas activities, and the potential for extremely slow (centuries to millennia) recovery from perturbation in deep waters, an integrated approach to conservation is warranted. This will include spatial management in conjunction with activity management in the form of restrictions on discharge and the use of water-based drilling fluids, and temporal management in areas where industry activity is near breeding aggregations or seasonally spawning sessile organisms.

Most countries have an in-principle commitment to conservation that typically extends to deep-water ecological features. However, it is rare that mandatory set-back distances from sensitive features or extensions of spatial protections are included to ensure that industrial activity does not impact the habitats designated for protection. This is significant because these habitats, in particular deep-sea coral and cold-seep ecosystems, consist of central, high-biomass sites surrounded by transition zones that can extend at least 100 m from the visually apparent border of the site to the background deep-sea community (Demopoulos et al., 2014; Levin et al., 2016). Considering the inherent sources of uncertainty associated with the management of deep-sea habitats, from the imprecise placement of seafloor infrastructure, to the variability in discharge impact distances, to the uncertainty in seafloor navigation and the locations of the sensitive deep-sea habitats and species, we strongly recommend that buffer zones be incorporated into spatial management plans.

TABLE 4 | Recommendations for the spatial management of deep-sea ecosystems in the vicinity of oil and gas industrial activity.

1. Establish robust baseline ecological survey data within planning area and in appropriate reference areas
2. Determine the locations, size and type of Ecological and Biological Significant areas (EBSAs) through comprehensive surveys including visual imagery
3. Establish protected areas around significant areas of representative communities
4. Establish borders of protected areas to be set-back distances based on typical distances of impacts from installations:
    - 200 m from seafloor infrastructure with no expected discharges
    - 2 km from any discharge points and surface infrastructure
5. Consider activity and temporal management to restrict impacts
6. Implement a comprehensive and robust monitoring programme that can reliably detect significant environmental changes in areas of exploration activity, areas inside the established MPAs, and reference sites outside of MPAs and activity zones

Based on what is known on distances over which impacts have been observed, we can propose a set of recommendations for appropriate buffer zones or MPA extensions from sensitive habitats (**Table 4**). Following the Deepwater Horizon spill, impacts to the deep-sea benthos were greatest within a 3 km radius with a signal detected within a 45 km radius (Montagna et al., 2013), and impacts to deep-sea coral communities were observed within a 25 km radius of the location of the Deepwater Horizon drilling rig (Fisher et al., 2014a). While distances derived from the spatial footprints of large spills might offer a solid precautionary approach in regions undergoing development for the first time, they may prove impractical in most settings. For example, a 25 km buffer around each of the BOEM mitigation areas in the Gulf of Mexico would exclude drilling from ∼98% of the actively leased blocks of the northern Gulf of Mexico. Therefore, in regions of active leasing, the focus should be on the protection of suitably large, representative areas, while still allowing for industrial activity in the area.

The size of the buffer zones around habitats should be based on the available information on the typical distances over which impacts of standard oil and gas industry operations have been documented. Produced water travels 1–2 km on average, elevated concentrations of barium (a common component of drilling muds) are often detected for at least 1 km from the source, and cuttings and other surface disposed materials, along with changes to the benthic community are often observed on the seafloor at distances of up to 200–300 m. Considering that impacts can extend to 2 km, we recommend that surface infrastructure and any discharge sites should be at least 2 km away from known EBSAs. A more conservative approach, based on the variability in water column current structure and intensity, would be to set the distance as a function of the water depth of operations, with the 2 km extent of typical impacts observed as the minimum distance. Seafloor disturbances from direct physical impacts of anchor, anchor chain, and wire laying occur within a 100 m radius of activities. In addition, the infaunal community is significantly different between the typical deep-sea benthos and areas within ∼100 m of deep-water coral reef structures (Demopoulos et al., 2014) or cold seeps (Levin et al., 2016). Therefore, based on the combination of the typical impact distance and the transition zone to the background deep-sea community, we recommend that any seafloor infrastructure without planned discharges should be placed at least 200 m from the location of these communities. Temporal management should also be considered, particularly during discrete coral spawning events (Roberts et al., 2009).

Although these recommendations are based on a thorough review of available literature and the authors' extensive experience in several EEZs, the information on potential impact zones is still relatively sparse. As a result, processes should be implemented that allow adaptive management to be implemented as more data become available. Management plans must clearly communicate quantitative conservation targets that are measurable, the set of environmental and ecological features to be protected, the levels of acceptable change, and any remedial actions required, increasing the capacity of the industry to better cost and implement compliance measures as part of their license to operate. It is also in the best interests of scientists, managers, and industry alike to arrive at a common, global standard for deep-water environmental protection across EEZs, and it is our hope that this review represents a first step in this direction toward the integrated and comprehensive conservation of vulnerable deep-sea ecosystems.

AUTHOR CONTRIBUTIONS

EC and DJ wrote, edited and revised the text, created and edited figures and tables. TS contributed analysis and figures and edited and revised the manuscript. All authors contributed to the tables, wrote portions of the text, and edited the manuscript.

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