Endangered Species Act Biological Assessment
And
Essential Fish Habitat Evaluation

Use of Oil Spill Dispersants
And
In-Situ Burning
As Part of Response Actions Considered
By the
Caribbean Regional Response Team

Prepared by:

CRRT Response Technologies Committee
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Biological Assessment and Essential Fish Habitat Evaluation
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I. DESCRIPTION OF PROPOSED ACTIONS

The U.S. Coast Guard (USCG) and Environmental Protection Agency (EPA), as the Co-Chairs of the Caribbean Regional Response Team (CRRT), request reinitiation of consultation under Section 7 of the Endangered Species Act (ESA) with the National Marine Fisheries Service (NMFS), on the potential use of dispersants and in-situ burning in waters of the Caribbean region. Informal ESA Section 7 Consultations were completed with NMFS on June 14, 1995 for the CRRT preauthorization agreement on in-situ burning, and on March 24, 1997 for the CRRT preauthorization agreement on dispersants. NMFS concurred with the USCG's determination that implementation of the proposed dispersant and in-situ burning agreements was not likely to adversely affect listed species, with the implementation of certain measures to avoid any potential effects to these species. Since completion of those consultations on the dispersant and in-situ burning plans' effects on whales and sea turtles, new species have been listed, and critical habitats have been designated, that require reinitiation of consultation under the ESA.

The CRRT also requests the initiation of an essential fish habitat (EFH) consultation pursuant to the requirements of the Magnuson-Stevens Fishery Conservation and Management Act (Magnuson-Stevens Act; 16 USC 1801 et seq). An EFH consultation has not been completed previously for the use of dispersants and in-situ burning in the U.S. Caribbean.

Following an oil spill, response actions are needed to minimize or reduce the threat to human health and environmental impacts. While physical control and recovery techniques are the traditional response measures, other countermeasures such as dispersants and/or in-situ burning also need to be considered. The CRRT agrees that the primary method of controlling discharged oil should be the physical removal of the oil from the environment. However, the CRRT recognizes that the complete physical containment, collection, and removal of oil discharges may not be possible. The use of dispersants and in-situ burning may therefore be considered to prevent a substantial threat to the public health or welfare, or to minimize the threat of impacts to the environment. The CRRT encourages the combination of effective techniques to minimize a spill's effect.

A. Concurrence and Consultation Requirements under NCP Subpart J

Subpart J of the National Oil and Hazardous Substances Pollution Contingency Plan (NCP) provides for the RRT representatives for EPA, the affected states[1], and the DOC and DOI natural resource trustees to review and either approve, disapprove, or approve with modification preauthorization plans for the use of chemical countermeasures for oil spill response. If preauthorization is approved, the Federal On-Scene Coordinator (FOSC) may authorize the use

[1] The definition of *State* under Section 300.5 of the NCP includes the Commonwealth of Puerto Rico and the U.S. Virgin Islands

of chemical countermeasures as specified in the plan without obtaining specific concurrences from EPA, the affected states, or DOC and DOI.

For spill situations that are not addressed by preauthorization plans, the FOSC, with the concurrence of the EPA representative to the RRT and, as appropriate, the RRT representatives from the states with jurisdiction over the navigable waters threatened by the release or discharge, and in consultation with the DOC and DOI natural resource trustees, may authorize the use of chemical countermeasures, provided that the products are on the NCP Product Schedule.

B. CRRT Dispersant Preauthorization Agreements

Between 1991 and 1995, the CRRT signed two Letters of Agreement (LOAs) on Limited Use of Dispersants and Chemical Agents for Puerto Rico and the U.S. Virgin Islands. Under those LOAs, the following waters are designated as preauthorized areas for the initiation of dispersant application:

a. For Puerto Rico:
    - Waters at least 0.5 miles seaward of any shoreline; and
    - Waters at least 30 feet (ft) in depth.

b. For U.S. Virgin Islands:
    - Waters at least 1.0 miles seaward of any shoreline or at least one mile from any reef which is less than 20 feet from the water's surface.; and
    - Waters at least 60 ft in depth.

Additionally, the "Protocols" section of the LOAs states that dispersants or chemical agents shall not be used in, on, or over waters containing reefs; waters designated as marine reserves; mangrove areas; or waters in coastal wetlands; except with the prior and express concurrence of the commonwealth/territory and EPA, in consultation with DOC and DOI. Coastal wetlands were identified as including:

1. Submerged algae beds (rocky or unconsolidated bottom)
2. Submerged seagrass beds
3. Coral reefs

ESA Section 7 consultations were conducted for listed species at the time the LOAs were developed, and were completed in 1995. However, due to the listing of additional endangered species and the designation of critical habitat since 1995, reinitiation of Section 7 consultation on the preauthorization agreement is required for newly listed species and designated critical habitat, and for any species that were included in the previous consultation for which new information is available related to the potential impacts of the use of chemical countermeasures on these species or their habitats.

EFH consultation under MSFCMA was not conducted at that time because the amendment was finalized in 1996, and has since been further amended. Therefore, an EFH consultation is also being requested on the preauthorization agreement by the CRRT at this time, in conjunction with the ESA Section 7 consultation.

C. Consensus Conditional Zones for Dispersant Consideration

Between 2003 and 2008, three Ecological Risk Assessment Consensus Workshops (ERAs) were conducted within the Caribbean region. The purpose of the workshops was to provide oil spill response training and to evaluate the relative risk to natural resources from various oil spill response options (e.g. on water mechanical recovery, dispersant application and shoreline cleanup) in comparison to natural recovery. Among the recommendations from those workshops was the consensus that dispersant usage in the Caribbean should be considered in waters shallower and closer to shore than identified in the current LOAs.

Based on the results of the ERAs, as well as increased research, expertise and knowledge concerning the use of dispersants and other chemical countermeasures, the CRRT has determined that the FOSC should consider the use of chemical countermeasures to be potentially beneficial in waters with the following depth and distance from shore (known as the Consensus Conditional Zones):

a. For Puerto Rico:
    - Waters 30 ft in depth or more, regardless of distance from shoreline

b. For U.S. Virgin Islands:
    - Waters at least 1.0 mile seaward of any shoreline; and
    - Waters 30 ft in depth or more.

It is important to emphasize that this general consensus does NOT constitute preauthorization to apply chemical countermeasures. The designation of preauthorization within those parameters is not practical due to several factors, including the expansion of the Buck Island Reef National Monument off the northeast coast of St. Croix and creation of the Virgin Islands Coral Reef National Monument around St. John in 2001; and the presence of approximately 40 Marine Protected Areas in Puerto Rico and the US Virgin Islands. The intent of this designation is to identify waters where there is general consensus that dispersant usage may significantly reduce the negative short-term and long-term environmental impacts of oil spills; as well as to expedite the decision-making process to concurrently initiate the mobilization of resources for operational use, the concurrence and consultation requirements under Subpart J of the NCP, and the applicable ESA and EFH consultation procedures, as soon as possible.

In addition to the preauthorized zones identified earlier, this biological assessment addresses the potential use of dispersants within the Consensus Conditional Zones identified above.

D. CRRT Policy for the Use of In-Situ Burning

In 1996, the CRRT signed the "Caribbean Regional Response Team Policy for Use of In-Situ Burning in Ocean, Coastal and Inland Waters." The purpose of this Agreement is to provide concurrence of the USCG, EPA, DOC, DOI, and the Puerto Rico and USVI representatives to the CRRT for the pre-authorized use of in-situ burning in response to oil discharges within the jurisdiction of the CRRT. The CRRT recognizes that in some instances the physical containment and collection of oil is unfeasible or inadequate, and the effective use of in-situ burning as an oil spill response technique should be considered.

Pre-authorization within the set guidelines of this agreement allows the federal On-Scene Coordinator (OSC) to employ in-situ burning to: (1) prevent or substantially reduce a hazard to human life, (2) minimize the environmental impact of the spilled oil or, (3) reduce or eliminate economic or aesthetic losses which would otherwise presumably occur without the use of this technique. The agreement covers protocols under which in-situ burning is pre-authorized for use by the federal On-Scene Coordinator on waters off the coasts of Puerto Rico and the USVI which are within the boundaries of the CRRT region.

The term "in-situ burning" applies to operations conducted for removal of oil by burning. These operations may apply during daylight or nighttime hours. The authority to authorize the use of in-situ burning provided to the USCG OSC may not be delegated. The following four zones have been established to specify pre-authorized locations and conditions under which burning may occur:

1) <u>"A" ZONES – PRE-AUTHORIZATION FOR OPEN-WATER BURNING</u>

The "A" zone is defined as any area in the CRRT region, falling exclusively under federal jurisdiction; and not classified as a "B", "C", or "R" zone; which is at least 6 miles from any state coastline; and outside of any state waters. In the event that state jurisdiction extends beyond 6 miles from a state shoreline, pre-authorization for the "A" zone applies only to those areas outside the state jurisdiction.

Within "A" zones, the USCG, EPA, DOC, DOI, and the state(s) agree that the decision to initiate in-situ burning rests solely with the pre-designated USCG OSC, and that no further concurrence or consultation on the part of the USCG OSC with EPA, DOC, DOI, or the state(s) is required. The USCG agrees with EPA, DOC, DOI, and the state(s) that the USCG will immediately notify said agencies and state(s) of a decision to conduct burning within the "A" zone, via each agency or state's respective CRRT representative.

2) <u>"B" ZONES – PRE-AUTHORIZATION WITH FAVORABLE WIND CONDITIONS</u>

The "B" zone is defined as any areas under CRRT jurisdiction not classified as an "A", "C", or "R" zone; which is at least 3 miles from any state coastline; and outside of any state waters. In the event that state jurisdiction extends beyond 3 miles from a state shoreline, pre-authorization for the "B" zone applies only to those areas outside the state jurisdiction.

3) <u>"C" ZONES – WATERS REQUIRING CASE-BY-CASE APPROVAL</u>

"C" zones are all areas falling: 1) anywhere within state waters, 2) waters less than 30 ft in depth that contain living reefs, 3) waters designated as a marine reserve, National Marine Sanctuary, National or State Wildlife Refuge, unit of the National Park Service, proposed or designated Critical Habitats, and 4) mangrove areas, or coastal wetlands. Coastal wetlands include submerged algal beds, submerged seagrass beds, lagoons and salt ponds.

4) <u>"R" ZONES – EXCLUSION ZONES</u>

The "R" zone is that area designated by the USCG, EPA, DOC, DOI, and the state(s) as an exclusion zone. No in-situ burning operations will be conducted in the "R" zone unless; 1) in-situ burning is necessary to prevent or mitigate a risk to human health and safety; and/or 2) an emergency modification of this agreement is made on an incident-specific basis.

The CRRT currently has not designated any areas as "R" zones, but retains the right to include areas for exclusion at a future point in time if it feels this is warranted.

Under the CRRT policy, in-situ burning will be conducted in accordance with any consultations approved by the USFWS and the NMFS, under Section 7 of the ESA, and under the EFH provisions of the Magnuson-Stevens Act. Prior to beginning an in-situ burn, an on-site survey will be conducted to determine if any threatened or endangered species are present in the burn area or otherwise at risk from any burn operations, fire, or smoke. Appropriate natural resource specialists, knowledgeable about any special resource concerns in the area and representing the resource trustee, will be consulted prior to conducting any in-situ burn. Measures will be taken to prevent risk of injury to any wildlife, especially endangered or threatened species. Examples of potential protection measures may include: moving the location of the burn to an area where listed species are not present; temporary employment of auditory or visual hazing techniques to prompt wildlife to leave or avoid the location of the burn, if effective; and physical removal of individuals of listed species only under the authority of the trustee agency.

E. Description of Dispersants

Chemical dispersants are mixtures of surfactants and solvents designed to enhance the miscibility of oil with water, facilitating oil weathering and biodegradation. Consequently, the use of dispersants alters the spatial distribution, physical transport, and chemical and biological fate of spilled oil in aquatic environments. The intended purpose of dispersant application is to reduce the concentration of oil at the water surface by breaking the oil slick into smaller droplets that can be suspended and distributed (and subsequently diluted and biologically degraded) throughout the water column. Dispersant application is also a useful tool for reducing the amount of oil that may strand in shoreline habitats, when applied appropriately and in a timely manner (i.e., prior to migration of the slick into shallow waters, where oil cannot be greatly diluted, and prior to significant weathering of the oil), and is expected to substantially reduce the known long-term impacts of shoreline oiling (Peterson et al., 2003; Cross and Thomson, 1987).

Dispersing oil may reduce the damaging effects of surface slicks on birds, sea turtles, and marine mammals and help prevent oil from stranding on tourist and sea turtle nesting beaches or impacting sensitive shoreline habitats such as mangroves, marshes, or enclosed lagoons. Dispersants treat large slicks quickly, reducing the formation of emulsions, and enhance the biodegradation of oil in the water column. Dispersants are an important cleanup tool, and they offer a response method that can be deployed during weather conditions that preclude mechanical recovery methods. Dispersion is the primary response option in many countries and a secondary option in several others. Many tropical islands, atolls, and reefs are too remote to deploy mechanical protection and cleanup methods, but dispersant use may still be an option if pre-planning efforts have stockpiled dispersants in readily available locations.

Under the proper conditions, lighter fuel oils to medium crude oil can be easily dispersed, but heavier bunker oils are much more difficult to disperse. Weathering increases oil viscosity and may cause formation of water-in-oil emulsions, which are less amenable to dispersion. A moderate amount of turbulence is needed to mix dispersed oil into the surface waters.

The application of dispersants in a typical spill response involves the release of a large tank of undiluted dispersant chemical (commonly referred to as a sortie) from deployed vehicles (e.g., airplanes, boats, or helicopters) onto the surface of a spill on open water (Nuka Research, 2006). The volume released depends largely on the vehicles' carrying capacities for liquid dispersants (Nuka Research, 2006); however, the rate of application (i.e., volume per unit area) is expected to be as consistent as possible over a large area (ASTM, 2010a, 2010b, Nuka Research, 2006), resulting in a more or less uniform input of dispersant chemicals. Ideally, the dispersant droplets come into contact with the oil and mix rapidly, resulting in nearly instantaneous dispersion into the water column. Although dispersant is applied as evenly as possible, because oil slicks tend to be unevenly distributed across the ocean's surface (NRC, 2005), the true dispersant-to-oil ratio (DOR) is expected to vary spatially. The required volume of chemical dispersant is assumed to be that which is needed to coat the surface of an oil slick with minimal volume allowed for overspray (Scelfo and Tjeerdema, 1991). The typical DOR is 1:20, though ratios of 1:40 or even 1:60 could be achievable with some dispersants and some oil types. Conversely, DORs as high as 1:10 have been required with some of the more emulsified and viscous heavy oils (NRC, 2005).

Horizontal transport of dispersants and dispersed oil is largely driven by ocean currents. Both oil and dispersed oil will assumedly be carried in the direction of major currents. It has been noted that the spread of oil across the ocean's surface can rapidly increase after dispersant application (preceding dispersion into the water column) (NRC, 2005), and that dispersants sprayed at the edge of a slick can cause oil to be herded, whereby the slick area decreases somewhat (Fingas, 2008).

F. Chemical Constituents in Dispersants

The key components of chemical dispersants are one or more surface-active agents, or surfactants (e.g., Tween 80, Tween 85, Span 80, sodium dioctyl sulfosuccinate or DOSS). Surfactants contain molecules with both water-compatible (hydrophilic) and oil-compatible (lipophilic or hydrophobic) groups. The surfactant molecules reduce the oil/water interfacial surface tension, enabling the oil layer to be broken into smaller droplets with minimal mixing energy, thereby enhancing natural dispersion. Surfactants also tend to prevent coalescence of oil droplets and reduce adherence to solid particles and surfaces, such as sediments and feathers. In addition to surfactants, most dispersant formulations also contain a solvent carrier (e.g., propylene glycol, Dipropylene glycol, n-butyl ether, or DPnB) to reduce viscosity of the surfactant so that the dispersant can be sprayed uniformly. The solvent may also enhance mixing and penetration of the surfactant into more viscous oils. Though early dispersants contained agents highly toxic to marine life, manufacturers have refined formulations of more recent generations of dispersants to dramatically reduce toxicity. Modern dispersants contain solvents composed of nonaromatic hydrocarbons or water-miscible concentrates (alcohols or glycols) as well as less toxic surfactants.

The goal of dispersant application is to break the surface tension of the water-oil interface such that droplets of oil form that are small enough to remain suspended in the water column (Brandvik et al., 2010). Dispersant chemical formulations are designed to bind to non-polar substrates and crude oil specifically, so the individual chemicals in dispersants tend to move through the water column with plumes of dispersed oil (Kujawinski et al., 2011).[2] Once broken into small droplets, the oil mixes into the water column, effectively reducing the amount of oil slicks at the water surface and thus reducing the risk of exposure and fouling to surface water wildlife (e.g., seabirds, marine mammals, and turtles). Because chemical dispersion of oil enhances the partitioning of oil into the top few meters of the water column, the resulting oil concentrations are higher compared to oil physically dispersed by currents, wind and waves (NRC, 1989, 2005). As a result, entrained pelagic species (e.g., fish eggs and larvae) and slow moving organisms may be temporarily exposed to elevated oil concentrations after chemical dispersion, because typical concentrations of oil in the water column are low prior to dispersion, even just below the slick (Mackay and McAuliffe, 1988).

By dispersing oil into the water column, the spreading or dilution becomes three-dimensional. The subsurface oil concentration initially increases, but diminishes rapidly with distance and time due to physical transport processes. This is in contrast to untreated oil concentrated at the water surface, which can coalesce in surface convergence zones even after it has spread out to very low concentrations. The highest concentration of chemically dispersed oil typically occurs in the top meter of water during the first hour following treatment (Rycroft et. al., 1994). Available data suggest that concentrations of more than ten parts per million (ppm) of dispersed oil are unlikely beyond ten meters (depth) of the slick and that even within one meter depth of the slick, concentrations rarely exceed 100 ppm. The rate of oil and chemical dispersant mixing is primarily determined by the energy of the environment into which the dispersant is applied,

[2] Therefore, free dispersant in the water column is unlikely in the presence of oil; overspray into unoiled water is an exception and would result in partitioning to water.

although some additional factors contribute to effective dispersion (e.g., spill size, dispersant droplet size, penetration of spill upon impact, thickness of spill, extent of weathering, and the formation of less dispersible emulsions) (NRC, 2005). A calm sea will mix more slowly than churning waters, where waves stir the oil and dispersant together. Wind also produces turbulent mixing, facilitating dispersion (NRC, 2005). Both wave action and wind energy act on any oil, regardless of the presence of dispersants, and cause the natural dispersion of oil droplets. The continuous mixing and dilution capabilities of open water lead to uniformity and are sufficient to rapidly reduce these concentrations. Field studies show that water column concentrations decline to undetectable or background levels within several hours following application of a dispersant (SEA, 1995; Bejarano et al., 2014). Under untreated slicks, oil concentrations typically range from a few parts per million to less than 0.1 ppm, diminishing with depth and time.

The dispersed oil droplets, ranging in size from microns to a few millimeters, break down by natural processes, such as biodegradation. Microbial biodegradation of oil appears to be enhanced by chemical dispersion (Lessard and Demarco, 2000; Prince et al., 2013) because of the larger surface area available as compared to a surface slick. Dispersants also prevent formation of tarballs and oil-in-water emulsions (mousse), which tend to be resistant to biodegradation due to their low surface area. The chemical dispersants applied, like the oil droplets, are diluted by diffusion and convective mixing. Much of the solvent fraction evaporates immediately after the dispersant is applied. The surfactants are readily biodegraded under aerobic conditions (summarized in Lee, 2011). .

Gallaway et al. (2012) modeled the expected concentration of dispersant released to the environment assuming an application rate of 5 gallons of Corexit® 9500 per acre, a 10-km^2 area, and a total volume of 5,000 gallons of dispersant. The receiving waters were modeled as having a local initial value of approximately 18 parts per million (ppm) of Corexit® 9500, which was diluted rapidly over time. Within approximately one hour, the concentration of dispersant was diluted to below the species sensitivity distribution (SSD) - the Hazardous Concentration-5 (HC5) of dispersants or dispersed oil below which no expected acutely toxic effects will occur in 95% of aquatic species. The implication of this model is that the concentration of a dispersant is diluted rapidly after application to below the HC5 or concentrations below which no acute toxicity is expected relative to ESA-listed species (specific to dispersants alone). Overspray is unlikely to result in significant acute toxicity to planktonic, embryonic, or larval species of fish or invertebrates, because the duration of exposure to toxic concentrations is very short, much shorter than in controlled toxicity experiments. The rate of dispersant dilution indicated by the Gallaway et al. (2012) model is similar to that reported by Nedwed (2012), who indicated that concentrations of dispersant decreased to < 1 ppm within a matter of hours (and to the parts per billion [ppb] range within 24 hours). Modeling conducted by the National Oceanic and Atmospheric Administration (NOAA) using the General NOAA Operational Modeling Environment (GNOME) provides similar results (NOAA, 2012b): dispersion is rapid, and dilution drives concentrations of dispersants to < 1 ppm within 24 hours.[3]

[3] GNOME model inputs used to derive dispersant concentration dilution models assumed idealized conditions for dispersion, such as 100% effectiveness (NOAA, 2012b).

Similarly, McAuliffe et al. (1980, 1981) and Mackay and McAuliffe (1988) showed that dispersed oil, although highly concentrated in the water column below an oil slick immediately after dispersion, decreased to below what the authors considered to be protective levels within a matter of hours. Furthermore, the time-averaged concentration of dispersed oil was low (i.e., 0.46 ppm C_1-C_{10} hydrocarbons), even over short time periods immediately following the application of dispersant (i.e., between 10 and 30 minutes after application) (Mackay and McAuliffe, 1988). Although Mackay and McAuliffe (1988) measured only the light fraction of oil as it dispersed, it can be assumed that heavier fractions of oil (i.e., C_{11} and larger molecules) will disperse and dilute at the same rate (i.e., be transported within the same droplets of oil). That is not to say that dissolution and biodegradation of hydrocarbons into the water column from oil droplets will be equivalent, as heavier organic molecules tend to be inherently less soluble and less biodegradable than lighter fractions even in the presence of chemical dispersants (Yamada et al., 2003).

In all cases, concentrations of dispersant or dispersed oil are shown to be diluted below their respective HC5s in less than the 48- to 96-hour exposure durations used in toxicity tests. For this reason, it is expected that the chemical dispersion of oil will result in mitigated acute toxicity, even in relatively sensitive species, due to the reduction in exposure duration and concentration driven primarily by dilution. Furthermore, it is expected, based on previously published models of oil and dispersant dilution that limited acute toxicity will occur in pelagic species, such as ESA-listed scalloped hammerhead sharks or prey species of ESA-listed wildlife.

G. Toxicity of Dispersants

Toxicity values are commonly reported as the concentration of the product in the aqueous exposure media that is lethal or that causes an adverse effect to 50% of the exposed aquatic organisms (LC50 and EC50, respectively). Toxicity data provided by the manufacturers[4] on four dispersants listed on the NCP Product Schedule (Table 1) included test results with two standard test species, the inland silverside fish, *Menidia beryllina* (96 hour [h] tests), and a mysid shrimp, *Americamysis bahia* (48 h tests). The reported toxicity values, derived from constant exposures, fall within the range of what is considered slightly toxic to practically non-toxic, based on standard toxicity categories[5]. While there is considerably less toxicity data available for Nokomis® dispersants, their toxicity is generally comparable to the toxicity of the widely tested Corexits® dispersants. Similarly, toxicity data (96 h LC50 and EC50) for Corexit® 9527 and Corexit® 9500 (Figure 1) from different sources and displayed in the form of Species Sensitivity Distributions (SSDs)[6] showed that most existing data for a wide range of species fall within the slightly toxic to practically non-toxic range. While the HC5, or the concentration assumed to be protective of 95% of the species in the SSD, falls within the moderately toxic range, these concentrations are assumed to be worst case, as exposure concentrations under standard dispersant applications (ASTM, 2010a, 2010b), are not expected to remain constant in the water column. Consequently, when using the generally accepted dispersant application rate (5

[4] Table modified from data in http://www.epa.gov/osweroe1/content/ncp/product_schedule.htm
[5] LC50 and EC50 toxicity categories for aquatic organisms are as follows: very highly toxic <0.1 ppm; highly toxic 0.1-1 ppm; moderately toxic >1-10 ppm; slightly toxic >10-100 ppm; practically nontoxic >100 ppm. Source: http://www.epa.gov/oppefed1/ecorisk_ders/toera_analysis_eco.htm#Ecotox
[6] SSDs are probabilistic models that describe the relative sensitivity of species to a particular chemical. Each data point in a SSD represents the geometric mean of reported toxicity values for unique species.

gallons/acre at a prescribed 1:20 dispersant to oil ratio), dispersants are not expected to cause toxicity to most water column organisms. While it is notable that toxicity data derived from tropical species is limited in the scientific literature, it is likely that their range of sensitivity falls within the sensitivity range of species with other geographic distributions.

Table 1. Toxicity data (LC50; ppm or mg/L) of four listed dispersants for two test species as reported in the NCP Product Schedule. Values represent the mean LC50 and its associated 95% confidence interval.

Dispersant	LC50 (mg/L)	
	Menidia beryllina	*Americamysis bahia*
Corexit® 9500	25 (14–47)	32 (27–39)
Corexit® 9527	15	24
Nokomis® 3-AA	34 (29–38)	20 (17–23)
Nokomis® 3-F4	30 (24–35)	32 (28–37)

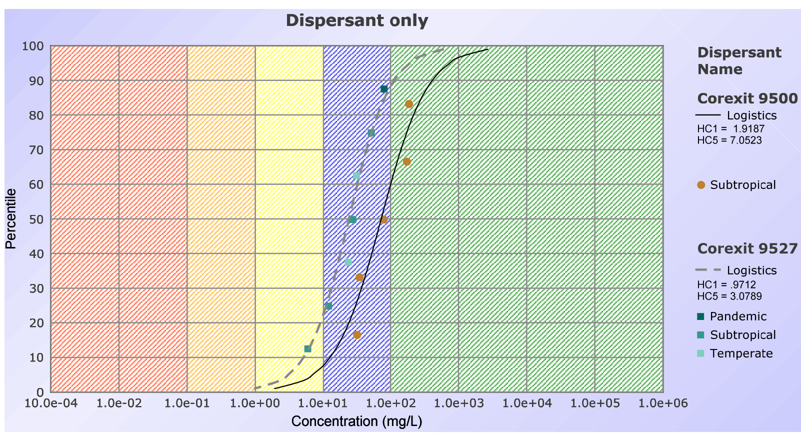

Figure 1. 95-h LC50 and EC50 data for Corexit® 9500 and Corexit® 9527 from constant exposures (worst case exposure scenario) (Bejarano and Dahlin, 2013). As a reference, the maximum dispersant concentrations under standard dispersant application rates is estimated at 5 mg/L

H. Toxicity of Chemically Dispersed Oil

The environment in which dispersants are applied is often much different than the system in which a controlled toxicology study is conducted. In an artificial test system with well-defined boundaries, oil is constrained even when dispersed, limiting dilution. In a large water body, such as an ocean or embayment, dispersed oil is less constrained. Typically, field applications are more effective in reducing surface oiling than are applications in laboratory tests, as shown by Nedwed and Coolbaugh (2008).

The exposure of species to toxic components of oil (i.e., polycyclic aromatic hydrocarbons [PAHs]) is likely to increase immediately after dispersant application (Yamada et al., 2003; Ramachandran et al., 2004; Milinkovitch et al., 2011a), and may result in increased toxicity (Barron, 2003; Barron et al., 2008). PAHs are likely to decrease rapidly in concentration as a result of natural processes (e.g., wave action, wind-driven currents and advection, photo-oxidation, and biodegradation), though toxicity may still occur (French-McCay, 2010).

Crude oil consists of a complex mixture of hydrocarbon compounds (aliphatic, aromatic, and asphaltic hydrocarbons), with different chemical and physical properties. The use of dispersants increases the oil loading and volumetric footprint of these oil constituents in the top few meters of the water column, temporarily increasing the risks to entrained organisms. However, under most field circumstances vertical and horizontal water mixing rapidly dilutes oil constituents, reducing the likelihood of long-term adverse effects.

The toxicity of both physically and chemically dispersed oil has been studied for several decades using a variety of experimental designs, which in many cases do not adequately represent open water exposures: short acute exposures to elevated oil concentrations declining rapidly over space and time due to dilution and water column mixing (reviewed in Bejarano et al., 2014). As a result, tests performed under constant static exposures tend to overestimate the acute toxicity of chemically dispersed oil when compared to spiked or flow through tests. Consistently, the latter are generally viewed to be more environmentally realistic (Aurand and Coelho, 2005; Clark et al., 2001; Singer et al., 1995). Despite these shortcomings in toxicity data, constant static exposures provide a measure of absolute worst case exposure conditions under most field exposures. When comparing toxicity data between physically and chemically dispersed oil on the basis of measured exposure concentrations (reviewed in Bejarano et al., 2014), there is not substantial evidence to suggest that chemically dispersed oil is more acutely toxic than physically dispersed oil (for example see Figure 2). Dispersants are designed to enhance the rate of partitioning of oil fractions into the water column, particularly PAHs, leading to aqueous exposures with higher hydrocarbon loading than exposures than those of physically dispersed oil. It is also important to note that, when compared to chemically dispersed oil, dispersants alone are generally less toxic than physically or chemically dispersed oil. Consequently, the majority of the toxicological effects in exposures to chemically dispersed oil are the result of exposures to oil constituents, and not to the toxicity of dispersants (Clark et al., 2001; Fuller and Bonner, 2001; NRC, 2005; Wetzel and Van Fleet, 2001).

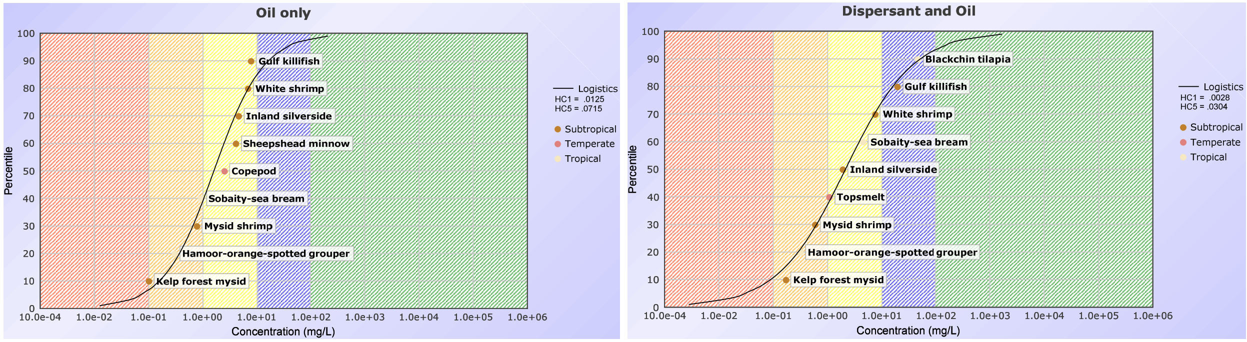

Figure 2. 95-h LC50 and EC50 constant exposure toxicity data for physically dispersed (left) and chemically dispersed (right) medium oils (worst case exposure scenario) reported on the basis of measured total hydrocarbon content. Toxicity data for chemically dispersed oil used all the available data for Corexit® 9500 and Corexit® 9527 (Bejarano and Dahlin, 2013).

While toxicity data provide valuable information that helps inform decisions, it is also important to interpret these data within the context of anticipated field exposures. Field trials in open ocean waters have provided valuable information on the fate and behavior of both physically and chemically dispersed oil (e.g., Cormack and Nichols, 1977; Lunel, 1994; McAuliffe et al., 1980; Strom-Kristiansen et al., 1997). Based on limited field trials, measured oil concentrations in the top few meters of the water column immediately following chemical dispersion of oil are generally ≤1 mg/L and typically higher (20-50 mg/L range) during the first 30 min of dispersant use (Cormack and Nichols, 1977; Lunel, 1994; McAuliffe et al., 1980; Strom-Kristiansen et al., 1997; reviewed in Bejarano et al., 2014). Similarly, oil concentrations at 1 m depth following surface dispersant use during the Deepwater Horizon (DWH) oil spill were up to 2 mg/L approximately 30 min after dispersant use (Bejarano et al., 2013). The same study reported that 96 of 102 water samples collected before and after surface dispersant use, and analyzed for total petroleum hydrocarbons (TPH), had concentrations below the conservative HC5 value for TPHs (0.81 mg/L).

I. Biodegradation of Dispersants and Dispersed Oil

Dispersants, once released into the environment, undergo physical and chemical processes much like spilled oil or other degradable substances. Neff (1988) indicated that as the volatile components of dispersants evaporate, physical processes initially control the rate of elimination of dispersants from a marine system.[7] After initial evaporation, biological processes determine the rate of removal from the environment.[8]

In a spiked laboratory exposure, Corexit® mixtures were reported to have a 107-minute half-life (i.e., time required for 50% degradation of chemical) in solution (George-Ares and Clark, 2000), indicating rapid removal from water under certain conditions. Mulkins-Phillips and Stewart (1974) also noted that dispersants are biodegradable, but that degradation occurred only after a

[7] Refer to Table 2, which indicates that current Corexit® formulations contain only one potentially volatile component, petroleum distillates.

[8] Dilution is also a major factor in determining the concentration of dispersed oil in the water column, although such redistribution of oil does not, in itself, result in removal from the environment.

microbial lag period in growth; this lag period is likely due to observed shifts in natural microbial communities in response to oil spills (Hazen et al., 2010; Lu et al., 2011; Baelum et al., 2012). A study by Okpokwasili and Odokuma (1990) observed that Corexit® 9527 biodegraded 90% or more within 16 days, and the half-life of the chemical mixture was approximately 2 to 3 days. Baelum et al. (2012) measured total Corexit® 9500 and the glycol and dioctyl sulfosuccinate sodium (DOSS) components individually in the presence of oil; the authors report rapid biodegradation of Corexit and DOSS within 5 to 20 days, but glycol components that were largely unaffected after 20 days. Mudge et al. (2011) specifically observed 1-(2-butoxy-1-methylethoxy)-2-propanol (DPnB), for which a half-life of approximately 30 days was determined.

Studies by Staples and Davis (2002), Kim and Weber (2005), EPA (2005, 2009, 2010), the Organization for Economic Cooperation and Development (OECD) (1997), and West et al. (2007) indicate that the component chemicals of Corexit® 9500 and Corexit® 9527 are marginally or readily biodegradable (as well as abiotically degradable). Table 2 provides a summary of biodegradation information for Corexit® component chemicals. The rates are given as either the half-life or percent degradation. Percent degradation is accompanied by the duration of the microbial exposure. The percent loss over time is used in determining biodegradability, such that a > 60% loss of a chemical within 28 days characterizes that chemical as readily biodegradable.

Table 2. Biodegradation information for Corexit® component chemicals

CAS No.	Chemical Name (Common Name)	Biodegradability	Half-Life (Days)	Concentration Loss (%, Duration)	Source(s)
57-55-6	1,2-propanediol (propylene glycol)	readily biodegradable	13.6	81%, 28 days	West et al. (2007); Dow AgroSciences (2012)
111-76-2	2-butoxyethanol[a]	readily biodegradable	nr	> 60%, 28 days	OECD (1997)
577-11-7	butanedioic acid, 2-sulfo-, 1,4-bis(2-ethylhexyl) ester, sodium salt (1:1) (DOSS)	readily biodegradable[b]	nr	66.4%, 28 days	EPA (2009)
		readily biodegradable	nr	91 to 97.7%, 3 to 17 days	TOXNET (2011)
1338-43-8	sorbitan, mono-(9Z)-9-octadecenoate (Span™ 80)	readily biodegradable	nr	58 to 62%, 14 to 28 days	EPA (2005, 2010)
9005-65-6	sorbitan, mono-(9Z)-9-octadecenoate, poly(oxy-1,2-ethanediyl) derivs. (Polysorbate 80)	not readily biodegradable	nr	52%, 28 days	Fisher Scientific (2010)
9005-70-3	sorbitan, tri-(9Z)-9-octadecenoate, poly(oxy-1,2-ethanediyl) derivs (Polysorbate 85)	readily biodegradable	nr	60 to 83%, 28 days[c]	EPA (2005)
29911-28-2	1-(2-butoxy-1-methylethoxy)-2-propanol (glycol ether DPnB)	readily biodegradable	10.3 – 28	> 60%, 28 days	Howard et al. (1991); Dow (1993, 1987); Staples and Davis (2002)
64742-47-8	petroleum distillates, hydro-treated, light[a]	readily biodegradable	nr	> 97%, 4.7 days	Rozkov et al. (1998)

[a] Potentially volatile component

[b] EPA states that DOSS did not biodegrade readily; however, the rate at which biodegradation occurred was greater than 60%, above the typical criterion for ready biodegradability. Therefore, it has been changed in the table to reflect the more widely accepted criterion.

[c] Value is expected based on the degradation of chemicals with similar chemical structures.

CAS – Chemical Abstracts Service

DOSS – dioctyl sulfosuccinate sodium

DPnB – dipropylene glycol n-butyl ether

EPA – US Environmental Protection Agency

nr – not reported

OECD – Organisation for Economic Cooperation and Development

Dispersants enhance the biodegradation of oil and the partitioning of hydrocarbon constituents by promoting the formation of oil droplets that are smaller in size (<70 µm diameter) than those created by physical forces (≥100 µm diameter) (NRC, 2005 and references herein). Compared to physically dispersed oil droplets, chemically dispersed oil droplets have larger surface to volume ratios providing a large surface area for the colonization of oil-biodegrading bacteria (Atlas, 1995; Atlas and Bartha, 1992; Leahy and Colwell, 1990; NRC, 2005).

The biodegradation of dispersed oil is well studied, although results vary among studies (NRC, 2005; Fingas, 2008; Bruheim et al., 1999). In general, biodegradation testing results indicate that oil dispersion increases the rate of oil elimination from the water column under a variety of conditions (Hua, 2006; Lindstrom et al., 1999; Lindstrom and Braddock, 2002; Hazen et al., 2010, as cited in Lee et al., 2011a; McFarlin et al., 2012b; Otitoloju, 2010; MacNaughton et al., 2003; Prince et al., 2003; Zahed et al., 2010; Zahed et al., 2011; Prince et al., 2013; Baelum et al., 2012). Zahed et al. (2011) reported Corexit® 9500-dispersed oil half-lives of 28, 32, 38, and 58 days at oil concentrations of 100, 500, 1,000, and 2,000 ppm, respectively; concentrations of dispersed oil have rarely exceeded 100 ppm during testing, and have not been shown to exceed 500 ppm (McAuliffe et al., 1980, 1981; Mackay and McAuliffe, 1988). These half-lives were all less than those of untreated oil: 31, 40, 50, and 75 days at the same respective oil concentrations. Baelum et al. (2012) reported that non-dispersed oil degraded only 20% within 20 days, whereas dispersed oil degraded by 60%, an increase of 40% caused by the addition of Corexit® 9500. Prince et al. (2013) reported half-lives for oil and Corexit® 9500-dispersed oil of 13.8 days and 11 days, respectively, corroborating previous results (Zahed et al., 2011; Baelum et al., 2012). It is important to note that the test conditions applied by Prince et al. (2013) and Baelum et al. (2012) were more relevant to Alaskan waters (i.e., water temperatures of 8 and 5°C, respectively) whereas those applied by Zahed et al. (2011) (i.e., water temperature of 27.5°C) are more representative of water temperatures around the U.S. Caribbean.

Increased biodegradation in the presence of dispersant chemicals is significant, but often incomplete. Biodegradation processes are limited largely to the lighter components of oil, and the addition of dispersants appears to facilitate the mineralization of oil only somewhat (McFarlin et al., 2012b). Studies investigating individual components of oil over time found that heavy components within degraded oil made up a larger proportion of the whole volume (Lindstrom and Braddock, 2002; Lindstrom et al., 1999). This has been shown to be true in field observations as well (Hazen et al., 2010; Atlas and Hazen, 2011). Heavier organic components of oil become enriched over time for both oil and dispersed oil (Lindstrom et al., 1999), so this phenomenon does not constitute a negative long-term impact on the degradation of oil relative to baseline conditions. Reductions in the biodegradation of some hydrocarbons due to the addition of chemical dispersant may be linked to selective inhibition of hydrocarbon-degrading bacteria in the marine environment (Hamdan and Fulmer, 2011). The results of such tests are not relevant to field conditions, considering the rapid community-level shifts that occur under natural conditions when oil and dispersant are introduced to a diverse microbial community (Hazen et al., 2010; Lu et al., 2011).

Lyman et al. (1990) indicate that components of Corexit® 9500 are not expected to be susceptible to photolysis, although hydrolytic degradation may occur in the absence of microbial action. The half-lives indicated for individual components range from 77 days for Tween 85® to 7.7 years for Span® 80 (TOXNET, 2011). Rates of hydrolytic degradation vary greatly based on pH. For example, DOSS has a half-life of 240 days at pH 8, but a half-life of 6.7 years at pH 7, in the absence of microbial degradation (TOXNET, 2011). Because these chemicals have much shorter half-lives for biodegradation than under abiotic conditions, (George-Ares and Clark, 2000; Baelum et al., 2012), it is not expected that abiotic degradation pathways play a major role in initial degradation of Corexit® dispersants in the field.

Similarly, it is expected that abiotic degradation is limited relative to biodegradation (and physical effects) in decreasing the dispersed oil in an aquatic system over an extended period of time. However, physical weathering is known to have a marked impact on the initial concentration of oil, primarily since evaporation from the ocean's surface can result in the loss of approximately 20–50% of an oil spill within 24 hours (Mackay and McAuliffe, 1988; Suchanek, 1993). Similarly, many components of oil (e.g., PAHs) are susceptible to photolysis (Shemer and Linden, 2007).

J. Description of In-Situ Burning

In-situ burning is an oil spill response technique which, when used under appropriate conditions, quickly and efficiently removes large quantities of oil from the water surface with minimal logistical support. A typical in-situ burn employs boats towing fire resistant boom in a U-shaped configuration, in which oil is collected, towed away from the main slick and ignited. The configuration is slowly towed during the burn in order to maintain the oil toward the back end of the boom at the minimum thickness necessary to sustain the burn. After the boomed oil is burned, the process is repeated. In-situ burning can be used simultaneously with other offshore oil spill response techniques or can be conducted when and where other techniques are insufficient or impossible.

Perhaps the biggest advantage of in-situ burning is that it can achieve a burn efficiency of up to 99 % of the oil contained in the boom, a substantially higher removal efficiency than is achieved with mechanical removal or dispersants. When conditions are optimal for an effective and safe ignition, burning can eliminate spilled oil at approximately 100 gallons/day/square foot. This elimination rate means that a single 500 foot fire boom positioned in a U-configuration to intercept an ongoing spill could provide enough burn area to sustain an elimination rate of 15,000 barrels per day (Allen and Ferek, 1993, Fingas *et al.*, 1994). A major operational advantage of in-situ burning is the lack of dependence on skimming, transfer, and storage equipment for recovered oil and water. Burning also reduces the amount of wastes for transport and disposal. Burning can be conducted at night. Operational monitoring can determine effectiveness and when to halt a burn, which can be accomplished easily by releasing the containment boom.

As with any response technique, effective use of in-situ burning requires a specific set of operational, environmental, and oil slick conditions (Fingas and Punt, 2000). Most crude and refined oils will burn on water if the oil layer is at least several millimeters (mm) thick (minimum of 1 mm for fresh crude oils; 2-5 mm for weathered crude oil; 10 mm for heavy fuel oils), the ignition area sufficiently large, and the temperature high enough to vaporize the oil for continued combustion. Emulsification, evaporation of lighter volatiles, and the thinning of spilled oil layers can significantly reduce the successful use of controlled burning. Stable emulsions of >25% water will not ignite. Consequently, burning at sea is most effective early in a spill response. Due to containment requirements for ignition, relatively calm wind (<15-18 knots [kt] for ignition and <15-25 kt to sustain a burn) and sea conditions (<3.5 ft) are also necessary.

Typically 97% to 98% of the heat produced during a burn is directed upward and outward so that any heat absorbed by the underlying water is generally negligible. This is particularly true where currents continuously cause an exchange of water below the burning oil. At mesoscale burn tests conducted in the Mobile, Alabama in 1992, researchers found that temperature did not increase in the static water layer at depths greater than four centimeters below the surface (Shigenaka and Barnea, 1993).

In-situ burning rapidly coverts the oil into its primary combustion products (carbon dioxide and water), a small amount of other gases such as carbon monoxide, nitrogen dioxide, and sulfur dioxide, a small percentage of smoke particulates and residue byproducts. Soot yield ranges from 1 to 15% based on oil type burned (Fingas, 2011; Ferek et al., 1997). The smoke particulates and other products of combustion produce a visible smoke plume. The heat generated by the burning oil in the boom causes the smoke to rise several hundred to several thousand feet and to be carried away by the prevailing winds. Larger particles usually only remain in the air for a few minutes to hours and settle near the source; smaller particles (10 microns and smaller) may remain in the air from several days to weeks and be spread by winds over wide areas or long distances from the original source (Middlebrook et al, 2011). These particulates are generally removed from the atmosphere by wet precipitation or when they come in contact with other surfaces. Only the smallest particles tend to remain in the air for long periods; however, as distance from the burn increases, the dilution of the components in the plume increases. Laboratory and field experiments indicate concentrations of the gases and fine particulate matter dissipate to background levels within less than two hundred meters downwind of the burn location. The exact distance depends on several factors, including size of the burn, wind velocity, and plume behavior (Walton, *et al*., 1993, 1994. Fingus *et al*., 1994).

A small percentage of the original oil volume remains as a taffy-like residue following an in-situ burn. Floating residue can be collected with nets and requires relatively small volumes for temporary storage. However, sinking of burn residues can occur. Studies have found a strong correlation between the densities of the burn residue versus the original oil (SL Ross, 2002). These tests suggest that medium and light crude oils, condensates, and light and intermediate refined products are likely to produce floating burn residues, while residues from heavy crude oils and heavy refined product burn are expected to sink. However, it is important to understand that these are general rules, and they cannot apply during a specific spill. For example, the light-medium crude oil (API = 37.2) released during the DWH spill did slowly cool and sink (Shigenaka et al., 2015).

Potential aquatic toxicity resulting from in-situ burning has been evaluated in laboratory studies and during the Newfoundland Oil Burn Experiment (NOBE), conducted in 1993. Results of these studies indicate that in-situ burning does not adversely affect the underlying water column beyond those effects already associated with the unburned oil. Lethal and sublethal toxicity and concentrations of petroleum hydrocarbons from the water collected in the vicinity of unburned and burned crude oil slicks in the open sea were extremely low with no significant differences found between water samples collected in both areas (Daykin et al., 1994). In Australia, bioassays using crude oil burn residues created in the laboratory showed no acute toxicity to amphipods and very low sub-lethal toxicity (burying behavior) to marine snails (Gulec and Holdway, 1999). It is important to remember that the surface area affected by in-situ burning is small relative to the total surface area and depth of a given body of water and that any adverse ecological impacts are likely to be confined to a small localized area. Based on these limited tests and the chemical composition of the burn residues, burn residues are expected to yield little or no chemical toxicity, and water quality and toxicity after a burn are expected to be the same or less as before the burn.

II. SPECIES AND HABITAT WITHIN THE ACTION AREA

A. Description of ESA-Listed Species Present

1. Sea Turtles

Loggerhead Sea Turtle (*Caretta caretta*)

Description: The loggerhead is characterized by a large head with blunt jaws. The carapace and flippers are a reddish-brown color; the plastron is yellow. The carapace has five pairs of costal scutes with the first touching the nuchal scute. There are three large inframarginal scutes on each of the bridges between the plastron and carapace. Adults grow to an average weight of about 200 pounds. This species was listed as threatened on July 28, 1978. On September 22, 2011, the Services listed 9 distinct populations segments (DPS) for loggerhead sea turtles, 4 as threatened and 5 as endangered. Loggerheads in the U.S. Caribbean are in the Northwest Atlantic Ocean DPS.

Nesting Season and Development: Nesting season extends from about May through August with nesting occurring primarily at night. Nesting is infrequent in Puerto Rico and reported only occasionally at Buck Island, St. Croix, in USVI. Loggerheads are known to nest from one to seven times within a nesting season (mean is about 4.1 nests per season) at intervals of approximately 14 days. Mean clutch size varies from about 100 to 126 along the southeastern U.S. coast. Incubation ranges from about 45 to 95 days, depending on incubation temperatures, but averages 55 to 60 days for most clutches in Florida. Hatchlings generally emerge at night. Remigration intervals of 2 to 3 years are most common in nesting loggerheads, but remigration can vary from 1 to 7 years. Age at sexual maturity is believed to be about 20 to 30 years. The species feeds on mollusks, crustaceans, fish, and other marine animals.

Distribution/Habitat: The loggerhead sea turtle can be found throughout the temperate and tropical regions of the Atlantic, Pacific, and Indian Oceans. It may be found hundreds of miles out to sea, as well as in inshore areas such as bays, lagoons, salt marshes, creeks, ship channels, and the mouths of large rivers. Coral reefs, rocky places, and ship wrecks are often used as feeding areas. Loggerheads nest on ocean beaches and occasionally on estuarine shorelines with suitable sand. Nests are typically made between the high tide line and the dune front. Most loggerhead hatchlings originating from U.S. beaches are believed to lead a pelagic existence in the North Atlantic gyre for an extended period of time, perhaps as long as 10 to 12 years, and are best known from the eastern Atlantic near the Azores and Madeira. Post-hatchlings have been found floating at sea in association with Sargassum rafts. Once they reach a certain size, these juvenile loggerheads begin recruiting to coastal areas in the western Atlantic where they become benthic feeders in lagoons, estuaries, bays, river mouths, and shallow coastal waters. These juveniles occupy coastal feeding grounds for a decade or more before maturing and making their first reproductive migration, the females returning to their natal beach to nest.

Green Sea Turtle (*Chelonia mydas*)

Description: The green sea turtle grows to a maximum size of about 4 ft and a weight of 440 pounds. It has a heart-shaped shell, small head, and single-clawed flippers. Color is variable. Hatchlings generally have a black carapace, white plastron, and white margins on the shell and limbs. The adult carapace is smooth, keelless, and light to dark brown with dark mottling; the plastron is whitish to light yellow. Adult heads are light brown with yellow markings. Identifying characteristics include four pairs of costal scutes, none of which borders the nuchal scute, and only one pair of prefrontal scales between the eyes. This species was listed under the ESA on July 28, 1978. The breeding populations in Florida and the Pacific coast of Mexico are listed as endangered; elsewhere the species is listed as threatened.

Nesting Season and Development: The nesting season varies with the locality. In Puerto Rico, it is roughly June through October. Nesting occurs nocturnally at 2, 3, or 4-year intervals. Only occasionally do females produce clutches in successive years. A female may lay as a many as nine clutches within a nesting season (overall average is about 3.3 nests per season) at about 13-day intervals. Clutch size varies from 75 to 200 eggs, with an average clutch size of 136 eggs reported for Florida. Incubation ranges from about 45 to 75 days, depending on incubation temperatures. Hatchlings generally emerge at night. Age at sexual maturity is believed to be 20 to 50 years.

Distribution/Habitat: The green turtle is globally distributed and generally found in tropical and subtropical waters along continental coasts and islands between 30° North and 30° South. In U.S. Atlantic and Gulf of Mexico waters, green turtles are found in inshore and nearshore (reefs and seagrass beds) waters from Texas to Massachusetts, the U.S. Virgin Islands, and Puerto Rico.

Critical habitat: In 1998, critical habitat for green turtles was designated in coastal waters around Culebra, extending seaward 3 nautical miles (5.6 km) from the mean high water line.

Leatherback Sea Turtle (*Dermochelys coriacea*)

Description: The leatherback is the largest, deepest diving, and most migratory and wide ranging of all sea turtles. The adult leatherback can reach 4 to 8 ft in length and 500 to 2000 pounds in weight. Its shell is composed of a mosaic of small bones covered by firm, rubbery skin with seven longitudinal ridges or keels. The skin is predominantly black with varying degrees of pale spotting; including a notable pink spot on the dorsal surface of the head in adults. A toothlike cusp is located on each side of the gray upper jaw; the lower jaw is hooked anteriorly. The paddle-like clawless limbs are black with white margins and pale spotting. Hatchlings are predominantly black with white flipper margins and keels on the carapace. Jellyfish are the main staple of its diet, but it is also known to feed on sea urchins, squid, crustaceans, tunicates, fish, blue-green algae, and floating seaweed. The leatherback turtle was listed under the ESA as endangered in 1970.

Breeding Season and Development: On Culebra, nesting occurs from about February to August with the peak occurring around April to May. Female leatherbacks nest an average of 5 to 7 times within a nesting season, with an observed maximum of 11 nests. The average interesting interval is about 9 to 10 days. The nests are constructed at night in clutches of about 70 to 80 yolked eggs. The white spherical eggs are approximately 2 inches in diameter. Typically incubation takes from 55 to 75 days, and emergence of the hatchlings occurs at night. Most leatherbacks return to their nesting beaches at 2 to 3-year intervals. Leatherbacks are believed to reach sexual maturity in 6 to 10 years. In the U.S., small nesting populations occur on the Florida east coast (35 females/year), Sandy Point, U.S. Virgin Islands (50 to 100 females/year), and Puerto Rico (30 to 90 females/year). The leatherback is the most pelagic of the sea turtles. Adult females require sandy nesting beaches backed with vegetation and sloped sufficiently so the crawl to dry sand is not too far. The preferred beaches have proximity to deep water and generally rough seas. Culebra beaches most used by the species are Flamenco, Brava, Resaca and Soni Beach.

Distribution/Habitat: The leatherback turtle is distributed worldwide in tropical and temperate waters of the Atlantic, Pacific, and Indian Oceans. It is also found in small numbers as far north as British Columbia, Newfoundland, and the British Isles, and as far south as Australia, Cape of Good Hope, and Argentina.

Designated critical habitat:

Critical habitat for leatherback turtles has been designated by NMFS in waters adjacent to Sandy Point, USVI, up to and inclusive of the waters from the hundred fathom curve shoreward to the level of mean high tide with boundaries at 17°42'12"N and 64°50'00"W.

USFWS critical habitat for leatherback turtles has been designated in the USVI as:

- A strip of land 0.2 miles wide (from mean high tide inland) at Sandy Point Beach on the western end of the island of St. Croix beginning at the southwest cape to the south and running 1.2 miles northwest and then northeast along the western and northern shoreline, and from the southwest cape 0.7 miles east along the southern shoreline.

Hawksbill Sea Turtle (*Eretmochelys imbricata*)

Description: The hawksbill turtle (*Eretmochelys imbricata*) is small to medium-sized compared to other sea turtle species. Adults weigh 100 to 150 lbs (45 to 68 kg) on average, but can grow as large as 200 lbs (91 kg). Hatchlings weigh about 0.5 oz (14 g). The carapace (top shell) of an adult ranges from 25 to 35 inches (63 to 90 cm) in length and has a "tortoiseshell" coloring, ranging from dark to golden brown, with streaks of orange, red, and/or black. The shells of hatchlings are 1-2 inches (about 42 mm) long and are mostly brown and somewhat heartshaped.

The plastron (bottom shell) is clear yellow. The rear edge of the carapace is almost always serrated, except in older adults, and has overlapping scutes. The hawksbill turtle's head is elongated and tapers to a point, with a beak-like mouth that gives the species its name.

Hawksbill turtles are unique among sea turtles in that they have two pairs of prefrontal scales on the top of the head and each of the flippers usually has two claws. This species was listed under the ESA as endangered in 1970.

Nesting Season and Development: The nesting season varies with locality, nesting occurs all year long in the U.S. Caribbean with a peak from August to November. Hawksbills nest at night and, on average, about 4.5 times per season at intervals of approximately 14 days. In Florida and the U.S. Caribbean, clutch size is approximately 140 eggs, although several records exist of over 200 eggs per nest. They nest under the vegetation on the high beach and nests have been observed having the last eggs of the clutch as close as 3 inches from the sand's surface. Remigration intervals of 2 to 3 years predominate. The incubation period averages 60 days. Hawksbills recruit into the reef environment at about 35 cm in length and are believed to begin breeding about 30 years later. However, the time required to reach 35 cm in length is unknown and growth rates vary geographically. As a result, actual age at sexual maturity is not known.

Distribution/Habitat: Hawksbill turtles use different habitats at different stages of their life cycle, but are most commonly associated with healthy coral reefs. The ledges and caves of coral reefs provide shelter for resting hawksbills both during the day and at night. Hawksbills are known to inhabit the same resting spot night after night. Hawksbills are also found around rocky outcrops and high energy shoals. These areas are optimum sites for sponge growth, which certain species are the preferred food of hawksbills. They are also known to inhabit mangrove-fringed bays and estuaries, particularly along the eastern shore of continents where coral reefs are absent.

Designated Critical Habitat for Hawksbill Sea Turtles:

On September 2, 1998, NMFS published a final rule designating hawksbill sea turtle critical habitat. Within the geographical area occupied by a listed species, critical habitat consists of specific areas on which are found those physical or biological features essential to the conservation of the species and which may consider special management considerations or protection. The essential features for hawksbill sea turtle critical habitat include breeding/nesting areas, food resources, water quality and quantity, and vegetation and soil types. Areas containing these features have been identified as waters extending from the mean high water line of Mona and Monito Islands, Puerto Rico. Mona Island lies approximately 39 nautical miles west of the southwest coral of mainland Puerto Rico. The area in general is bounded north to south by 18°13' North to 18°00' North and east to west by 67°48' West and 68°01' West. Both islands are surrounded by deep waters of the Mona Passage up to 1,000 meters (m) in depth. Benthic habitats less than 30 m deep around Mona Island amount to approximately 2,174 hectares (ha) and are composed of 92% coral reef and colonized hardbottom, 5% unconsolidated sediments, and 3% submerged vegetation based on benthic mapping by NOAA's National Ocean Service Biogeography Program in 2000 (Kendall et al. 2001).

USFWS critical habitat for hawksbill turtles is designated in 50 CFR 17.95 as:

- Isla Mona: All areas of beachfront on the west, south, and east sides of the island from mean high tide inland to a point 150 m from shore. This includes all 7.2 kilometers of beaches on Isla Mona.
- Culebra Island: The following areas of beachfront on the north shore of the island from mean high tide to a point 150 m from shore: Playa Resaca, Playa Brava, and Playa Larga.
- Cayo Norte: South beach, from mean high tide inland to a point 150 m from shore.
- Island Culebrita: All beachfront areas on the southwest facing shore, east facing shore, and northwest facing shore of the island from mean high tide inland to a point 150 m from shore.

2. Marine Mammals

Humpback Whale (*Megaptera novaeangliae*)

Description: Humpback whales are well known for their long "pectoral" fins, which can be up to 15 ft (4.6 m) in length. Their scientific name, *Megaptera novaeangliae*, means "big-winged New Englander" as the New England population was the one best known to Europeans. These long fins give them increased maneuverability; they can be used to slow down or even go backwards. Similar to all baleen whales, adult females are larger than adult males, reaching lengths of up to 60 ft (18 m). Their body coloration is primarily dark grey, but individuals have a variable amount of white on their pectoral fins and belly. This variation is so distinctive that the pigmentation pattern on the undersides of their "flukes" is used to identify individual whales, similar to a humans fingerprint.

In June 1970, humpback whales were designated as "endangered" under the Endangered Species Conservation Act (ESCA). In 1973, the ESA replaced the ESCA, and continued to list humpbacks as endangered.

Behavior, Development and Diet: Humpback whales travel great distances during their seasonal migration, the farthest migration of any mammal. The longest recorded migration was 5,160 miles (8,300 km). This trek from Costa Rica to Antarctica was completed by seven animals, including a calf. One of the more closely studied routes is between Alaska and Hawaii, where humpbacks have been observed making the 3,000 mile (4,830 km) trip in as few as 36 days. During the summer months, humpbacks spend the majority of their time feeding and building up fat stores (blubber) that they will live off of during the winter. Humpbacks filter feed on tiny crustaceans (mostly krill), plankton, and small fish and can consume up to 3,000 pounds (1360 kg) of food per day. Several hunting methods involve using air bubbles to herd, corral, or disorient fish. One highly complex variant, called "bubble netting," is unique to humpbacks. This technique is often performed in groups with defined roles for distracting, scaring, and herding before whales lunge at prey corralled near the surface.

In their wintering grounds, humpback whales congregate and engage in mating activities. Humpbacks are generally "polygynous" with males exhibiting competitive behavior on wintering grounds. Aggressive and antagonistic behaviors include chasing, vocal and bubble displays, horizontal tail thrashing, and rear body thrashing. Males within these groups also make physical contact; striking or surfacing on top of one another. These bouts can cause injuries ranging from bloody scrapes to, in one recorded instance, death. Also on wintering grounds, males sing complex songs that can last up to 20 minutes and be heard 20 miles (30 km) away. A male may sing for hours, repeating the song several times. All males in a population sing the same song, but that song continually evolves over time.

Gestation lasts for about 11 months. Newborns are 13 to 16 ft (4 to 5 m) long and grow quickly from the highly nutritious milk of their mothers. Weaning occurs between 6 and 10 months after birth. Mothers are protective and affectionate towards their calves, swimming close and frequently touching them with their flippers. Males do not provide parental support for calves. Breeding usually occurs once every two years, but sometimes occurs twice in three years.

Distribution/Habitat: Humpback whales live in all major oceans from the equator to sub-polar latitudes. In the western North Atlantic ocean, humpback whales feed during spring, summer, and fall over a range that encompasses the eastern coast of the U.S. (including the Gulf of Maine), the Gulf of St. Lawrence, Newfoundland/Labrador, and western Greenland. In winter, whales from the Gulf of Maine mate and calve primarily in the West Indies. Not all whales migrate to the West Indies every winter, and significant numbers of animals are found in mid- and high-latitude regions at this time.

During migration, humpbacks stay near the surface of the ocean. While feeding and calving, humpbacks prefer shallow waters. During calving, humpbacks are usually found in the warmest waters available at that latitude. Calving grounds are commonly near offshore reef systems, islands, or continental shores. Humpback feeding grounds are in cold, productive coastal waters.

Fin or Finback Whale (*Balaenoptera physalus*)

Description: Fin or finback whales are the second-largest species of whale, with a maximum length of about 75 ft (22 m) in the Northern Hemisphere, and 85 ft (26 m) in the Southern Hemisphere. Fin whales show mild sexual "dimorphism", with females measuring longer than males by 5-10%. Adults can weigh between 80,000-160,000 lbs (40-80 tons). Fin whales have a sleek, streamlined body with a V-shaped head. They have a tall, "falcate" dorsal fin, located about two-thirds of the way back on the body, that rises at a shallow angle from the animal's back. The species has a distinctive coloration pattern: the back and sides of the body are black or dark brownish-gray, and the ventral surface is white. The unique, asymmetrical head color is dark on the left side of the lower jaw, and white on the right side. Many individuals have several light-gray, V-shaped "chevrons" behind their head, and the underside of the tail flukes is white with a gray border. Within the U.S., the fin whale is listed as endangered throughout its range under the ESA and is listed as "depleted" throughout its range under the Marine Mammal Protection Act of 1972.

Behavior, Development and Diet: Fin whales can be found in social groups of 2-7 whales and in the North Atlantic are often seen feeding in large groups that include humpback whales, minke whales, and Atlantic white-sided dolphins. Fin whales are large, fast swimmers and the killer whale (*Orcinus orca*) is their only non-human predator.

During the summer, fin whales feed on krill, small schooling fish (e.g., herring, capelin, and sand lance), and squid by lunging into schools of prey with their mouth open, using their 50-100 accordion-like throat pleats to gulp large amounts of food and water. They then filter the food particles from the water using the 260-480 "baleen" plates on each side of the mouth. Fin whales fast in the winter while they migrate to warmer waters. Little is known about the social and mating systems of fin whales. Similar to other baleen whales, long-term bonds between individuals are rare. Males become sexually mature at 6-10 years of age; females at 7-12 years of age. Physical maturity is attained at approximately 25 years for both sexes. After 11-12 months of gestation, females give birth to a single calf in tropical and subtropical areas during midwinter. Newborn calves are approximately 18 ft (6 m) long, and weigh 4,000-6,000 lb (2 tons). Fin whales can live 80-90 years.

Distribution/Habitat: Fin whales are found in deep, offshore waters of all major oceans, primarily in temperate to polar latitudes, and less commonly in the tropics. They occur year-round in a wide range of latitudes and longitudes, but the density of individuals in any one area changes seasonally.

Sei Whale (*Balaenoptera borealis*)

Description: Sei whales are members of the baleen whale family and are considered one of the "great whales" or rorquals. Two subspecies of sei whales are recognized, *B. borealis* in the Northern Hemisphere and *B. schlegellii* in the Southern Hemisphere. These large animals can reach lengths of about 40-60 ft (12-18 m) and weigh 100,000 lbs (45,000 kg). Females may be slightly longer than males. Sei whales have a long, sleek body that is dark bluish gray to black in color and pale underneath. The body is often covered in oval-shaped scars (probably caused from cookie-cutter shark and lamprey bites) and sometimes has subtle "mottling". This species has an erect "falcate", "dorsal" fin located far down (about two-thirds) the animals back. They often look similar in appearance to Bryde's whales, but can be distinguished by the presence of a single ridge located on the animal's "rostrum." Bryde's whales, unlike other rorquals, have three distinct prominent longitudinal ridges on their rostrum. They have 219-410 baleen plates that are dark in color with gray/white fine inner fringes in their enormous mouths. They also have 30-65 relatively short ventral pleats that extend from below the mouth to the naval area. The number of throat grooves and baleen plates may differ depending on geographic population.

When at the water's surface, sei whales can be sighted by a columnar or bushy blow that is about 10-13 ft (3-4 m) in height. The dorsal fin usually appears at the same time as the blowhole, when the animal surfaces to breathe. This species usually does not arch its back or raise its flukes when diving.

This species was listed under the ESA as endangered in 1970.

Behavior, Development and Diet: They are usually observed singly or in small groups of 2-5 animals, but are occasionally found in larger (30-50) loose aggregations. Sei whales are capable of diving 5-20 minutes to opportunistically feed on plankton (e.g., copepods and krill), small schooling fish, and cephalopods (e.g., squid) by both gulping and skimming. They prefer to feed at dawn and may exhibit unpredictable behavior while foraging and feeding on prey. Sometimes seabirds are associated with the feeding frenzies of these and other large whales.

Sei whales become sexually mature at 6-12 years of age when they reach about 45 ft (13 m) in length, and generally mate and give birth during the winter in lower latitudes. Females breed every 2-3 years, with a gestation period of 11-13 months. Females give birth to a single calf that is about 15 ft (4.6 m) long and weighs about 1,500 lbs (680 kg). Calves are usually nursed for 6-9 months before being weaned on the preferred feeding grounds. Sei whales have an estimated lifespan of 50-70 years.

Distribution/Habitat: Sei whales have a cosmopolitan distribution and occur in subtropical, temperate, and subpolar waters around the world. They prefer temperate waters in the midlatitudes, and can be found in the Atlantic, Indian, and Pacific Oceans. During the summer, they are commonly found in the Gulf of Maine, and on Georges Bank and Stellwagen Bank in the western North Atlantic. The entire distribution and movement patterns of this species is not well known. This species may unpredictably and randomly occur in a specific area, sometimes in large numbers. These events may occur suddenly and then not occur again for long periods of time. Populations of sei whales, like other rorquals, may seasonally migrate toward the lower latitudes during the winter and higher latitudes during the summer. They prefer subtropical to subpolar waters on the continental shelf edge and slope worldwide and they are usually observed in deeper waters of oceanic areas far from the coastline.

Sperm Whale (*Physeter macrocephalus*)

Description: Sperm whales are the largest of the odontocetes (toothed whales) and the most sexually dimorphic cetaceans, with males considerably larger than females. Adult females may grow to lengths of 36 ft (11 m) and weigh 15 tons (13607 kg). Adult males, however, reach about 52 ft (16 m) and may weigh as much as 45 tons (40823 kg). It is distinguished by its extremely large head, which takes up to 25 to 35% of its total body length. It is the only living cetacean that has a single blowhole asymmetrically situated on the left side of the head near the tip. Sperm whales have the largest brain of any animal (on average 17 pounds (7.8 kg) in mature males), however, compared to their large body size, the brain is not exceptional in size. There are between 20-26 large conical teeth in each side of the lower jaw. The teeth in the upper jaw rarely erupt and are often considered to be vestigial. It appears that teeth may not be necessary for feeding, since they do not break through the gums until puberty, if at all, and healthy sperm whales have been caught that have no teeth.

Sperm whales are mostly dark gray, but oftentimes the interior of the mouth is bright white, and some whales have white patches on the belly. Their flippers are paddle-shaped and small compared to the size of the body, and their flukes are very triangular in shape. They have small dorsal fins that are low, thick, and usually rounded.

This species was listed under the ESA as endangered in 1970.

Behavior, Development and Diet: Because sperm whales spend most of their time in deep waters, their diet consists of many larger organisms that also occupy deep waters of the ocean.
Their principle prey are large squid weighing between 3.5 ounces and 22 pounds (0.1 kg and 10 kg), but they will also eat large demersal and mesopelagic sharks, skates, and fishes. The average dive lasts about 35 minutes and is usually down 1,312 ft (400 m), however dives may last over an hour and reach depths over 3280 ft (1000 m).

Female sperm whales reach sexual maturity around 9 years of age when they are roughly 29 ft (9 m) long. At this point, growth slows and they produce a calf approximately once every five years. After a 14-16 month gestation period, a single calf about 13 ft (4 m) long is born. Although calves will eat solid food before one year of age, they continue to suckle for several years. Females are physically mature around 30 years and 35 ft (10.6 m) long, at which time they stop growing. For about the first 10 years of life, males are only slightly larger than females, but males continue to exhibit substantial growth until they are well into their 30s. Males reach physical maturity around 50 years and when they are 52 ft (16 m) long. Unlike females, puberty in males is prolonged, and may last between ages 10 to 20 years old. Even though males are sexually mature at this time, they often do not actively participate in breeding until their late twenties.

Most females will form lasting bonds with other females of their family, and on average 12 females and their young will form a family unit. While females generally stay with the same unit all their lives in and around tropical waters, young males will leave when they are between 4 and 21 years old and can be found in "bachelor schools", comprising of other males that are about the same age and size. As males get older and larger, they begin to migrate to higher latitudes (toward the poles) and slowly bachelor schools become smaller, until the largest males end up alone. Large, sexually mature males that are in their late 20s or older, will occasionally return to the tropical breeding areas to mate.

Distribution/Habitat: They inhabit all oceans of the world. They can be seen close to the edge of pack ice in both hemispheres and are also common along the equator, especially in the Pacific. Sperm whales are found throughout the world's oceans in deep waters between about 60° N and 60° S latitudes. Their distribution is dependent on their food source and suitable conditions for breeding, and varies with the sex and age composition of the group. It migrations are not as predictable or well understood as migrations of most baleen whales. In some mid-latitudes, there seems to be a general trend to migrate north and south depending on the seasons (whales move poleward in the summer). However, in tropical and temperate areas, there appears to be no obvious seasonal migration.

Sperm whales tend to inhabit areas with a water depth of 1968 ft (600 m) or more, and are uncommon in waters less than 984 ft (300 m) deep. Female sperm whales are generally found in deep waters (at least 3280 ft, or 1000 m) of low latitudes (less than 40°, except in the

North Pacific where they are found as high as 50°). These conditions generally correspond to sea surface temperatures greater than 15°C, and while female sperm whales are sometimes seen near oceanic islands, they are typically far from land. Immature males will stay with female sperm whales in tropical and subtropical waters until they begin to slowly migrate towards the poles, anywhere between ages 4 and 21 years old. Older, larger males are generally found near the edge of pack ice in both hemispheres. On occasion, however, these males will return to the warm water breeding area. No critical habitat has been designated for this species.

Blue Whale (*Balaenoptera musculus*)

Description: The blue whale is a cosmopolitan species of baleen whale. In the Northern Hemisphere, they are generally smaller than those in the Southern Ocean. Maximum body length in the North Atlantic was about 88.5 ft (27 m) and the largest blue whale reported from the North Pacific was about 88 ft (26.8 m). Adults in the Antarctic can reach a maximum body length of about 108 ft (33 m) and can weigh more than 330,000 pounds (150,000 kg). As is true of other baleen whale species, female blue whales are somewhat larger than males. Blue whales are identified by the following characteristics: a long-body and comparatively slender shape; a broad, flat "rostrum" when viewed from above; a proportionately smaller dorsal fin than other baleen whales; and a mottled gray color pattern that appears light blue when seen through the water. This species was listed under the ESA as endangered in 1970.

Behavior, Development and Diet: Scientists have yet to discern many details regarding the life history of the blue whale. The best available science suggests the gestation period is approximately 10-12 months and that blue whale calves are nursed for about 6-7 months. Most reproductive activity, including births and mating, takes place during the winter. Weaning probably occurs on, or en route to, summer feeding areas. The average calving interval is probably two to three years. The age of sexual maturity is thought to be 5-15 years. There are no known differences in the reproductive biology of blue whales in the North Pacific and North Atlantic oceans.

The primary and preferred diet of blue whales is krill (euphausiids). In the North Atlantic, blue whales feed on two main euphausiid species: *Thysanoëssa inermisand* and *Meganyctiphanes norvegica*. In addition, *T. raschiiand* and *M. norvegica* have been recorded as important food sources of blue whales in the Gulf of St. Lawrence. In the North Pacific, blue whales prey mainly on *Euphausia pacifica* and secondarily on *T. spinifera*. While other prey species, including fish and copepods, have been mentioned in the scientific literature, these are not likely to contribute significantly to the diet of blue whales.

Distribution/Habitat: They are found in oceans worldwide and are separated into populations by ocean basin in the North Atlantic, North Pacific, and Southern Hemisphere. They follow a seasonal migration pattern between summering and wintering areas, but some evidence suggests that individuals remain in certain areas year-round. The extent of knowledge concerning distribution and movement varies with area and migratory routes are not well known but, in general, distribution is driven largely by food requirements.

Blue whales inhabit sub-polar to sub-tropical latitudes. Poleward movements in spring allow the whales to take advantage of high zooplankton production in summer. Movement towards the subtropics in the fall allows blue whales to reduce their energy expenditure while fasting, avoid ice entrapment in some areas, and engage in reproductive activities in warmer waters of lower latitudes. Although the species is often found in coastal waters, blue whales are thought to occur generally more offshore than humpback whales, for example.

3. Fish

Scalloped Hammerhead Shark (*Sphyrna lewini*)

On July 3, 2014, NMFS issued a final determination to list the Central and Southwest (SW) Atlantic Distinct Population Segment (DPS) of scalloped hammerhead shark as threatened species under the Endangered Species Act (ESA). The Central & SW Atlantic DPS is bounded to the north by 28° N. lat., to the east by 30° W. long., and to the south by 36° S. lat. All waters of the Caribbean Sea are within this DPS boundary, including the Bahamas' Exclusive Economic Zone (EEZ) off the coast of Florida, the U.S. EEZ off Puerto Rico and the U.S. Virgin Islands, and Cuba's EEZ.

Description: Scalloped hammerhead sharks are moderately large sharks with a global distribution. The eight or so species of hammerhead sharks are characterized by the flat, extended head or "cephalofoil." The cephalofoil of a scalloped hammerhead shark is characterized by an indentation located centrally on the front margin of the broadly arched head. Two more indentations flank the main central indentation, giving this hammerhead a "scalloped" appearance. They feed on crustaceans, teleosts, cephalopods and rays.

Habitat: The scalloped hammerhead shark is a coastal pelagic species that can also be found in ocean waters and occurs over continental and insular shelves and adjacent to deeper water. It has been observed close inshore and even entering estuarine habitats, as well as offshore to depths of 1000m. Adult aggregations are common at seamounts, especially near the Galapagos, Malpelo, Cocos and Revillagigedo Islands and within the Gulf of California, but otherwise adults can be solitary or occur in pairs.

Distribution: Scalloped hammerhead sharks are found worldwide residing in coastal warm temperate and tropical seas in the Atlantic, Pacific, and Indian Oceans between 46°N and 36°S to depths of 1000 m.

Nassau grouper *(Epinephelus striatus)*

On September 2, 2014, NMFS, announced a 12-month finding and listing determination on a petition to list the Nassau grouper as threatened or endangered under the Endangered Species Act (ESA). After reviewing the best scientific and commercial data available, NMFS determined that the Nassau grouper meets the definition of a threatened species. While the species still occupies its historical range, spawning aggregations have been reduced in size and number due to fishing pressure. The lack of adequate management measures to protect these aggregations increases the extinction risk of Nassau grouper. Based on these considerations, NMFS concluded that the Nassau grouper is not currently in danger of extinction throughout all or a significant portion of its range, but is likely to become so within the foreseeable future. However, a final listing rule has not yet been published for this species.

Description: The Nassau grouper is a long-lived, moderate sized Serranid fish with large eyes and a robust body. The range of color is wide, but ground color is generally buff, with 5 dark brown vertical bars and a large black saddle blotch on top of caudal peduncle and a row of black spots below and behind eye. Color pattern can change within minutes from almost white to bicolored to uniformly dark brown, according to the behavioral state of the. A distinctive bicolored pattern is seen when two adults or an adult and large juvenile meet and is frequently observed in spawning aggregations. There is also a distinctive dark tuning-fork mark beginning at the front of the upper jaw, extending dorsally (on top) along the interorbital region, and then dividing into two branches on top of the head behind the eyes; another dark band from the tip of the snout through the eye and then curving upward to meet its fellow just before the dorsal-fin origin. Juveniles exhibit a color pattern similar to adults.

Maximum age has been estimated up to 29 years, based on an ageing study using sagittal otoliths. Most studies also indicate rapid growth, which has been estimated to be about 10 mm/month (total length (TL)) for small juveniles, and 8.4 to 11.7 mm/month for larger juveniles (30-270 mm TL). Maximum size is about 122 cm TL and maximum weight is about 25 kg. Generation time (the average age of parents in the population) is estimated as 9-10 years.

Distribution, Habitat and Depth: The Nassau grouper's confirmed distribution currently includes Bermuda, Florida, and throughout the Bahamas and Caribbean. The Nassau grouper is primarily a shallow-water, insular fish species that has long been valued as a major fishery resource throughout the wider Caribbean, South Florida, Bermuda and the Bahamas. The Nassau grouper is considered a reef fish, but it transitions through a series of developmental shifts in habitat. As larvae, they are planktonic. After an average of 35-40 days and at an average size of 32 mm TL, larvae recruit from an oceanic environment into demersal habitats. Following settlement, Nassau grouper juveniles are reported to inhabit macroalgae (primarily *Laurencia* spp.), coral clumps (*Porites* spp.), and seagrass beds. Recently-settled Nassau grouper have also been collected from tilefish (*Malacanthus plumieri*) and rubble mounds at 18 m depth. Post-settlement, small Nassau grouper have been reported with discarded queen conch shells (*Strombus gigas*) and other debris around turtle grass beds.

Juvenile Nassau grouper (120-150 mm TL) are relatively solitary and remain in specific areas for months. Juveniles of this size class are associated with macroalgae, and both natural and artificial reef structure. As juveniles grow, they move progressively to deeper areas and offshore reefs. Schools of 30-40 juveniles (250-350 mm TL) were observed at 8-10 m depths in the Cayman Islands. No clear distinction can be made between types of adult and juvenile habitats, although a general size segregation with depth occurs—with smaller Nassau grouper in shallow inshore waters (2 to 9 fathoms) and larger individuals more common on deeper (10 to 30 fathoms) offshore banks. Recent work by Nemeth and coworkers in the USVI (manuscript, in prep) found more overlap in home ranges of smaller juveniles compared to larger juveniles, and adults having larger home ranges with less overlap. Mean home range of adult Nassau grouper in the Bahamas was 18,305m2+/−5,806 (SD) with larger ranges at less structurally complex reefs (Bolden 2001). The availability of habitat and prey was found to significantly influence home range of adults.

Adult Nassau grouper tend to be relatively sedentary and are generally associated with high relief coral reefs or rocky substrate in clear waters to depths of 130 m. Generally adults are most common at depths less than 100 m except when at spawning aggregations where they are known to descend to depths of 255m.

Diet and Feeding: Adult Nassau grouper are unspecialized, bottom-dwelling, ambush-suction predators. Numerous studies describe Nassau grouper as piscivorous as adults. Feeding takes place throughout the diel cycle although most fresh food is found in stomachs collected in the early morning and at dusk. Young Nassau grouper (20.2-27.2 mm SL) feed on a variety of plankton, including pteropods, amphipods, and copepods.

Spawning Behavior and Habitat: Nassau grouper form spawning aggregations at predictable locations around the winter full moon, or between full and new moons. Aggregations consist of hundreds, thousands, or, historically, tens of thousands of individuals. Some aggregations have persisted at known locations for periods of 90 years or more. Pair spawning has not been observed.

About 50 individual spawning aggregation sites have been recorded, mostly from insular areas in the Bahamas, Belize, Bermuda, British Virgin Islands, Cayman Islands, Cuba, Honduras, Jamaica, Mexico, Puerto Rico, Turks and Caicos and the USVI; however, many of these may no longer form. Recent evidence suggests that spawning is occurring at what appear to be reconstituted or novel spawning sites in both Puerto Rico and the USVI. Nassau grouper migrate to aggregation sites in groups numbering between 25 and 500, moving parallel to the coast or along shelf edges or even inshore reefs. Distance traveled by Nassau grouper to aggregation sites is highly variable with some fish moving only a few kilometers, while others move up to several hundred kilometers.

It is not known how Nassau grouper select and locate aggregation sites or why they aggregate to spawn. Spawning aggregation sites are typically located near significant geomorphological features, such as projections (promontories) of the reef as little as 50 m from the shore, and close to a drop-off into deep water over a wide (6-60 m) depth range. Sites are characteristically small, highly circumscribed areas, measuring several hundred meters in diameter, with soft corals, sponges, stony coral outcrops, and sandy depressions. Recent work has identified geomorphological similarities in spawning sites that may be useful in applying remote sensing techniques to discover previously unknown spawning sites.

Spawning aggregations usually form between December and March within the narrow water temperature range of 25-26 °C over a wide range of day-lengths. Temperature is evidently a more important stimulus for spawning than day length. In more northerly latitudes (i.e., Bermuda), the reproductive season falls between May and August, peaking in July.

Spawning occurs for up to 1.5 hours around the time of sunset for several days in each of several months.

At spawning aggregation sites, Nassau grouper tend to mill around for a day or two in a "staging area" adjacent to the core area where spawning activity actually takes place. Spawning involves a rapid horizontal swim or a "rush" of bicolor fish following dark fish closely in either a column or cone rising to within 20-25 m of the water surface where group-spawning occurs in sub-groups of 3-25 fish. Then there is release of sperm and eggs and a rapid return of the fragmented sub-group to the substrate. Similar accounts of spawning behavior from the USVI described the aggregated fish as a cone in the water column rather than being dispersed across the bottom. All spawning events have been recorded within 20 minutes of sunset and most within 10 minutes of sunset. Repeated spawning occurs at the same site for up to three consecutive months during the correct moon phase. Participation by individual fish across the time periods is unknown. It has been suggested that individual females spawn repeatedly over several different days during one aggregation based on reproductive tissue.

4. Corals

Elkhorn coral (*Acropora palmata*)

Description: It is a large, branching coral with thick and sturdy antler-like branches and is found in shallow reefs, typically in water depths from 0-35 ft, as these corals prefer areas where wave action causes constant water movement. Colonies are fast growing: branches increase in length by 2-4 inches (5-10 cm) per year, with colonies reaching their maximum size in approximately 10-12 years. Over the last 10,000 years, elkhorn coral has been one of the three most important Caribbean corals contributing to reef growth and development and providing essential fish habitat. This species was listed under the ESA as threatened on May 4, 2006.

Color: Living colonies are yellow, brown or golden with light rims.

Habitat: Elkhorn coral was formerly the dominant species in shallow water (3 ft-16 ft [1-5 m] deep) throughout the Caribbean and on the Florida Reef Tract, forming extensive, densely aggregated thickets (stands) in areas of heavy surf. Coral colonies prefer exposed reef crest and fore reef environments in depths of less than 20 ft (6 m), although isolated corals may occur to 65 ft (20 m).

Distribution/Reproduction: Elkhorn coral is found on coral reefs in southern Florida, the Bahamas, and throughout the Caribbean.

The dominant mode of reproduction for elkhorn coral is asexual, with new colonies forming when branches break off of a colony and reattach to the substrate. Sexual reproduction occurs via broadcast spawning of gametes into the water column once each year in July, August or September. Individual colonies are both male and female (simultaneous hermaphrodites) and will typically release millions of "gametes". The coral larvae (planula) live in the plankton for several days until finding a suitable area to settle, but very few larvae survive to settle and metamorphose into new colonies. The preponderance of asexual reproduction in this species raises the possibility that genetic diversity may be very low in the remnant populations.

Staghorn coral (*Acropora cervicornis*)

Description: It is a branching coral with cylindrical branches ranging from a few centimeters to over 6.5 ft (2 m) in length. This coral exhibits the fastest growth of all known western Atlantic corals, with branches increasing in length by 4-8 inches (10-20 cm) per year. This species was listed under the ESA as threatened on May 4, 2006.

Color: Living colonies are light, grayish to yellowish-brown.

Habitat: Staghorn coral occur in back reef and fore reef environments from 0-100 ft (0 to 30 m) deep. The upper limit is defined by wave forces, and the lower limit is controlled by suspended sediments and light availability. Fore reef zones at intermediate depths of 15-80 ft (5-25 m) were formerly dominated by extensive single species stands of staghorn coral until the mid-1980s.

Distribution/Reproduction: Staghorn coral is found in the Atlantic Ocean, Caribbean Sea, and western Gulf of Mexico. Specifically, staghorn coral is found throughout the Florida Keys, the Bahamas, the Caribbean islands, and Venezuela. The northern limit of staghorn coral is around Boca Raton, FL.

The dominant mode of reproduction for staghorn coral is asexual fragmentation, with new colonies forming when branches break off a colony and reattach to the substrate. Sexual reproduction occurs via broadcast spawning of gametes into the water column once each year in August or September. Individual colonies are both male and female (simultaneous hermaphrodites) and will release millions of "gametes". The coral larvae (planula) live in the plankton for several days until finding a suitable area to settle, but very few larvae survive to settle and metamorphose into new colonies. The preponderance of asexual reproduction in this species raises the possibility that genetic diversity is very low in the remnant populations

Designated Critical Habitat for Elkhorn and Staghorn Corals

On November 26, 2008, a final rule designating *Acropora* critical habitat was published in the Federal Register. Within the geographical area occupied by a listed species, critical habitat consists of specific areas on which are found those physical or biological features essential to the conservation of the species. The feature essential to the conservation of *Acropora* species (also known as essential feature) is substrate of suitable quality and availability, in water depths from the mean high water line to 30 m, to support successful larval settlement, recruitment, and reattachment of fragments. Substrate of suitable quality and availability means consolidated hardbottom or dead coral skeletons free from fleshy macroalgae or turf algae and sediment cover. Areas containing these features have been identified in Puerto Rico, St. Thomas/St. John, and St. Croix.

In addition, a 4(d) rule (50 CFR Part 223) establishing "take" prohibitions for elkhorn and staghorn corals went into effect on November 28, 2008. Take includes collect, bother, harm, harassment, damage to, death, or other actions that affect health and survival of listed species.

Lobed Star Coral (*Orbicella annularis*)

Description: The colonies grow in several morphotypes that were originally described as separate species. The species occurs as long, thick columns with enlarged, dome-like tops; large
mounds; sheets with skirt-like edges; irregularly bumpy mounds and plates or as smooth plates. Colonies up to 10 ft (3 m) in diameter with distinctive, often somewhat raised, corallites covering the surface.

Color: Shades of green to brown, yellow-brown and gray.

Habitat: Inhabit most reef environments and the species is often the predominant coral between 22-82 ft (7-25 m). The flattened plates are most common at deeper reefs, down to 165 ft (50 m).

Distribution: Common to abundant Florida, Bahamas and Caribbean.

Mountainous Star Coral (*Orbicella faveolata*)

Description: This species has been called the "dominant reef-building coral of the Atlantic". *Orbicella faveolata* buds extratentacularly to form head or sheet colonies with corallites that are uniformly distributed and closely packed, but sometimes unevenly exsert. Septa are highly exsert, with septocostae arranged in a variably conspicuous fan system, and the skeleton is generally far less dense than those of its sibling species. Active growth is typically found at the edges of colonies, forming a smooth outline with many small polyps.

Color: It is usually pale brown but may be bright, fluorescent green over the dark brown.

Habitat: *O. faveolata* is found from 3-100 ft (1-30 m) in backreef and forereef habitats, and is often the most abundant coral between 30-65 ft (10-20 m) in fore-reef environments.

Distribution: This species occurs in the Caribbean, the Gulf of Mexico, Florida, and the Bahamas. May also be present in Bermuda, but this requires confirmation.

Boulder Star Coral (*Orbicella franksi*)

Description: This species builds massive, encrusting plate or subcolumnar colonies via extratentacular budding. The characteristically bumpy appearance of this species is caused by relatively large, unevenly exsert, and irregularly distributed corallites. *O. franksi* is distinguished from its sibling *Orbicella* (formerly *Montastraea*) species by this irregular or bumpy appearance; a relatively dense, heavy, and hard skeleton (corallum); thicker septo-costae with a conspicuous septocostal midline row of lacerate teeth; and a greater degree of interspecies aggression.

Color: It is basically orange-brown with many pale patches on the lumpy surface, but may be grey or greenish-brown.

Habitat: This species mostly grows in the open like other species of this genus but smaller, encrusting colonies are common in shaded overhangs. It is uncommon in very shallow water, but becomes common deeper.

Distribution: This species occurs in the Caribbean, the Gulf of Mexico, Florida, and the Bahamas.

Pillar Coral (*Dendrogyra cylindrus*)

Description: Colonies form numerous, heavy, cylindrical spires, that grow upwards from an encrusting base mass. The colonies can attain a height of 10 ft (3 m), with a pillar diameter of more than 4 inches (10 cm). Polyps are normally extended during the day, giving the colony a fuzzy appearance and obscuring the long, meandroid, corallite series.

Color: Light tan to golden brown and chocolate brown.

Habitat: Colonies are typically found on flat gently sloping back reef and fore reef environment in depths of 3-82 ft (1-25 m). The species does not occur in extremely exposed locations.

Distribution: This species occurs in the Caribbean, the southern Gulf of Mexico, Florida, and the Bahamas.

Rough Cactus Coral (*Mycetophyllia ferox*)

Description: Colonies consist of flat plates with radiating valleys. It is a widely recognized valid species with colonies comprised of thin, weakly attached plates with interconnecting, slightly sinuous, narrow valleys. Tentacles are generally absent and corallite centers tend to form single rows. The walls of the valleys commonly join to form closed valleys, a feature not seen in other members of *Mycetophyllia*. The ridges are usually small and square, with a groove on top. The ridges, or walls between valleys, are commonly quite thin, and are irregular, and valleys are narrower.

Color: Valleys and walls are contrasting shades of grays and browns.

Habitat: This species is most common in fore reef environments from 5-30 m (but is more abundant from 10-20 m), but also occurs at low abundance in certain deeper back reef habitats and deep lagoons.

Distribution: This species occurs in the Caribbean, southern Gulf of Mexico, Florida, and the
Bahamas

B. Essential Fish Habitat

In 1996, amendments to the Magnuson-Stevens Act, set forth a number of new mandates for NOAA Fisheries, most of which focused on the identification, establishment and management of EFH. EFH can include rivers, estuaries, bays and open ocean (out to 200 miles) that are considered "essential" for the sustainable health of commercial fisheries. Under the Act and rules promulgated by the NMFS in 2002, federal agencies must consult and submit EFH assessments to NOAA Fisheries regarding potential or actual adverse effects of all actions authorized, funded, or undertaken by the agency that may adversely impact EFH, this includes emergency responses to oil discharges and chemical releases (response actions, not the material spilled).

The Magnuson-Stevens Act requires that EFH be identified for all fisheries that are federally managed. This includes species managed by regional Fishery Management Councils (FMCs) under federal Fishery Management Plans (FMPs), as well as those managed by the NMFS, such as highly migratory species, under FMPs developed by the Secretary of Commerce. Due to the number of species managed by the NMFS and because EFH is defined in the Magnuson-Stevens Act as *"...those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity."* EFH designations are cumulatively quite expansive.

Dispersants and in-situ burning may be used in habitat where oil has been spilled, including ocean and coastal waters. Therefore, EFH as identified by the Caribbean Fishery Management Council (CFMC) and the NMFS present in the Caribbean (Puerto Rico and the USVI) may be affected. The information provided in this consultation addresses the EFH provisions of the Magnuson-Stevens Act that require consultation with the NMFS when a proposed federal activity may adversely affect identified EFH. Adverse effect means that the proposed activity may have an impact that reduces the quality and/or quantity of EFH.

1. Specific Areas that May be Affected by Oil Spill Response Operations

EFH generally encompasses broad estuarine and marine habitats, including:

Estuarine areas
- Water column
- Salt marshes
- Mangrove wetlands

- Intertidal flats/salt ponds
- Sand and shell substrate
- Live and hard bottoms
- Mud flats
- Sandy beaches
- Rocky shores

Marine areas
- Water column
- Seagrass
- Sand and shell substrate
- Coral reefs
- Algal plains
- Live bottoms

The CFMC defines EFH in each of the FMPs that it administers as:

- EFH for the spiny lobster fishery in the U.S. Caribbean consists of all waters from mean high water to the outer boundary of the EEZ (habitats used by phyllosome larvae) and seagrass, benthic algae, mangrove, coral, and live/hard bottom substrates from mean high water to 100 fathoms depth (habitats used by other life stages).

- EFH for the queen conch fishery in the U.S. Caribbean consists of all waters from mean high water to the outer boundary of the EEZ (habitats used by eggs and larvae) and seagrass, benthic algae, coral, live/hard bottom and sand/shell substrates from mean high water to 100 fathoms depth (habitats used by other life stages).

- EFH for the Reef Fish Fishery in the U.S. Caribbean consists of all waters from mean high water to the outer boundary of the EEZ (habitats used by eggs and larvae) and all substrates from mean high water to 100 fathoms depth (habitats used by other life stages).

- EFH for the Coral Fishery in the U.S. Caribbean consists of all waters from mean low water to the outer boundary of the EEZ (habitats used by larvae) and coral and hard bottom substrates from mean low water to 100 fathoms depth (habitat used by other life stages).

Because these species collectively occur in all habitats of the US Caribbean, the EFH includes all waters and substrates (mud, sand, shell, rock, and associated biological communities), including coral habitats (coral reefs, coral hardbottoms, and octocoral reefs), sub-tidal vegetation (seagrasses and algae) and adjacent intertidal vegetation (wetland and mangroves). Therefore, EFH includes virtually all marine waters and substrates (mud, shell, rock, coral reefs, and associated biological communities) from the shoreline to the seaward limit of the EEZ.

Below are life history descriptions developed and used by the CFMC (2004) for identifying and describing EFH for Council managed species.

2. Life history of Spiny lobster (*Panulirus argus*)

Eggs. Most spiny lobster mate in spring and early summer along outer reefs near the shelf edge (CFMC 1998). Male spiny lobsters deposit sperm packets on the underside of the female. The fertilized eggs stick to the swimmerets beneath her tail and hatch in about four weeks. Therefore, the eggs utilize the habitat used by the females.

Larvae. The larval stage of spiny lobster is long (0-6 months) and there are numerous planktonic larval stages (phyllosoma). The distribution of lobster phyllosoma larvae (*Panulirus*) in the water column (depth), its temporal and ontogenetic patterns across an inshore-offshore gradient is currently being undertaken (Sabater and Garcia, 1997). Seasonal variation in depth (e.g. absent from the surface (0-20 m) in May and most abundant at the surface in August) and horizontal distribution (maximum abundance at the innermost stations) are currently being assessed for the area of La Parguera, Puerto Rico. Phyllosoma larvae have been reported from beyond their normal tropical range. Temperature tolerance for post-larvae is high ranging from 10 to 35 degrees C. Post-larval recruitment has been scantly assessed in the US Caribbean (i.e., Monterrosa, 1986; Maidment, 1997 (USVI SEAMAP-Caribbean Program)). Pueruli have been collected in *Acanthopora* clumps and turtle grass beds but were never found in coral rubble (Monterrosa, 1986). Additional juvenile habitat includes sea urchins, algal mats (plains), and rock crevices. All these habitats reported from very shallow areas.

Juveniles. The most important habitats for juvenile lobster appear to be turtle grass beds and mangroves. At an age of about 2 years migration to reef areas begins.

Adults. Adult populations of *Panulirus argus* are associated with reefs and hard bottoms, mostly with coral outcrops, crevices, caves, and ledges. The association of adult *Panulirus argus* to seagrass beds and algal plains relates to their nocturnal feeding activities.

3. Life History of Queen Conch (*Strombus gigas*).

Eggs. As stated in CFMC (1996), egg masses are spawned in clean coral sand with low organic content but have also been reported from seagrass beds (historical information). Females cover the egg mass with sand grains. The production of egg masses has been correlated to temperature and weather conditions (highest temperatures and longer photoperiods increase number of egg masses while stormy weather conditions decrease the number of egg masses laid. Incubation period is about 5 days.

Larvae. The larvae (known as veligers) of the queen conch are pelagic. Substrate conditions to metamorphose and settle to the bottom seem critical but unfortunately at present the requirements are largely unknown (CFMC 1996). No additional information is available to the Council at this time. The laboratory data could be applied to environmental gradients in the field once these have been identified. Larvae have been found offshore and can be transported up to 26 miles per day (i.e., 540 miles during the 3- week larval period). Posada and Appeldoorn (1994) however conclude that even when larvae are found offshore most larvae are retained within the area where they spawned.

Juveniles. The information available to the Council that identifies EFH for juveniles is presented in CFMC (1996). Little is known about juveniles in the wild. Juveniles are found buried in the sediment, the burial depth changing with size. For example, conch 35-54 mm are found buried 3-4 cm in the sand. Predation is very high at this early stage (e.g., 50% survival reported by Sandt and Stoner, 1993). In the Bahamas, Stoner et al. (1994) found that areas of strong tidal circulation contain a higher number of juveniles. "The occurrence of sandbars, where larval settlement may occur, adjacent to seagrass meadows as nursery areas is potentially significant" at least in Lee Stocking Island (Stoner et al., 1994). Stoner and Waite (1990) suggested that seagrass biomass, as well as seagrass shoot density were critical features in these nursery habitats. Davis and Stoner (1994) showed that for laboratory cultured conch, larvae metamorphose in response to algae, epiphytes, and sediments found in natural nursery grounds. However, they reported that no conch metamorphosed when exposed to conspecifics. Required habitat for juvenile conch includes among other things a delicate balance between seagrass beds and the surrounding sandy areas. Juveniles require red algae for feeding. The degradation of these habitats worsens the problem of overfishing since for juvenile settlement the presence of other juveniles seems to be required (Stoner and Ray, 1993).

Adults. Queen conchs are found on sandy bottoms that support the growth of seagrasses and epiphytic algae upon which they feed. They also occur on gravel, coral rubble, smooth-hard coral, or beach rock bottoms. Queen conch commonly occur on sandy bottoms that support the growth of seagrasses, primarily turtle grass (*Thalassia testudinum*), manatee grass (*Syringodium filiforme*), shoal grass (*Halodule beaudettei* [formerly known as *wrightii*]), and epiphytic algae upon which they feed (Randall, 1964). They also occur on gravel, coral rubble, smooth hard coral or beach rock bottoms and sandy algal beds. They are generally restricted to waters where light can penetrate to a depth sufficient for plant growth. Queen conch are reported from depths greater than 200 ft. Queen conch are often found in sandy spurs that cut into offshore reefs.

4. Life History of Corals

Eggs and Larvae. Corals reproduce both sexually and asexually. Sexual reproduction results in the formation of minute larvae (planulae) that spend a variable amount of time in the water column as plankton (from days to weeks), eventually settling on an appropriate substrate. If reproduction is asexual, larvae are brooded in the gastric pouch of the parent and released when ready to settle. Most corals have well defined seasonal patterns of sexual reproduction (Szmant, 1986), and many have quite specific requirements for appropriate settlement substrate. Larval capacity for substrate selection is unknown for most species but is likely to vary among them. After settling, larvae develop a skeleton and, if colonial, start budding additional polyps that will eventually form an adult colony. Natural selection probably acts more intensely during initial larval recruitment (Crisp, 1977) and is probably the reason for production of vast numbers of gametes. Individuals of some species delay sexual reproduction and use their available energy for asexual growth until a colony size safe from predation has been attained (Szmant, 1985). Rock and dead coral surfaces are also vital substrates for the settlement of larval phases of benthic organisms that cannot settle onto living coral. Suitability of substrate is one of the major factors controlling the distribution of many species. For example, natural, rough, substrate covered with other living organisms, presence of other larvae, and absence of certain organisms are all-necessary for octocoral settlement. Many other coral species also have specific substrate requirements for larval settlement. Kinzie (1971) found that natural substrate cleared of other organisms had no appreciable octocoral colonization even after six months (Wheaton, 1989). Other factors that influence the settlement of sessile organisms include total surface area available for settlement, conditioning period of substrate, surface relief including crevices and ridges, substrate orientation, and substrate composition (Wheaton, 1989). Thus, both physical and biological complexity are essential for the development of the reef ecosystem. Coral reefs and live-rock habitats form the backbone of this complex.

Juveniles. Even though *Montastrea annularis* is one of the most abundant corals off La Parguera and in many reefs off Mayaguez, juvenile colonies of this species have not been commonly reported. The same can be said for other sites in the Caribbean (e.g., Bak and Engel, 1979; Hughes, 1985). Very small colonies of this species can be frequently observed in La Parguera. However, Goenaga and Boulon (1992) report that upon close inspection, it can be seen that these are remnants of larger colonies that have undergone partial mortality and the rest of their skeleton have been covered by other organisms such as filamentous algae. While this is generally true for the USVI as well, very small, apparently juvenile colonies have been observed in certain localities (e.g. Salt River submarine canyon (Boulon, 1979)

Adults. The EFH for corals which are sessile organisms is being defined on the basis of the bottom types were they are found since this is where they feed grow to maturity, breed and spawn. The ability to disperse is achieved through sexual reproduction, known for a number of these species, where eggs and sperm are shed, in most cases once a year, into the water column, mixed and are then at the mercy of the currents and tides. Forms with branching morphology and high growth rates (e.g., *Acropora palmata* and *A. cervicornis*) can disperse through breakage during storms (e.g., Highsmith, 1982). Resulting fragments can, although not always do, recruit onto the substrate, and form a new colony.

Coral reefs and other coral communities are one of the most important ecological (and economic) coastal resources in the Caribbean. These biogenic habitats act as barriers to storm waves, and their architecture provides habitats for a wide variety of marine organisms including most of the economically important species of fish and shellfish, are the primary source for carbonate sand, and serve as the basis for much of the tourism (Dayton et al. 2002). Coral reefs are built upon the accumulation of calcium carbonate produced by living animals, coral polyps, in symbiosis with a dinoflagellate, zooxanthellae. Coral reef communities or solitary specimens exist throughout the geographical areas of authority of the Council. This wide distribution places corals in many variable habitats, from nearshore environments to continental slopes and canyons, including the intermediate shelf zones. Shelf-edge reefs are the best developed, but least studied of the reef systems in this area, with live coral cover ranging from 10-50 %, and living coral occurring as deep as 40 m (Bruckner 1999; Morelock et al. 2001). Coral reefs are among the most productive and diverse tropical marine habitats. Although highly productive, they develop best in shallow, well-lighted tropical waters that are usually poor in nutrients such as nitrates, ammonia and phosphates.

The ecological importance of coral reefs is well-documented (Goenaga and Cintrón, 1979). Many fish species and crustaceans of commercial and recreational value depend on coral reefs during some or all of their life stages. They provide a buffer against shoreline erosion and influence the deposition and maintenance of sand on the beaches that they protect. The sand in these beaches originates principally from the reefs.

5. Life History of Reef Fish

In the 2004 EFH Final Environmental Impact Statement (EFH-FEIS) and the 2005 Comprehensive Sustainable Fisheries Act Amendment (SFA), EFH for the Reef Fish FMP in the U.S. Caribbean was defined as all waters from mean high water to the outer boundary of the EEZ – habitats used by eggs and larvae – and all substrates from mean high water to 100 fathoms depth – used by other life stages.

As described in the 2004 EFH-FEIS, the US Caribbean ichthyofauna has been characterized as being composed by three groups (in terms of energetics): large, fast-swimming pelagic apex predators with loose reef affiliations, strongly reef-associated carnivores, and reef-associated herbivores (Opitz 1996). While reef-associated carnivores represent 70-80% of reef fish species in the US Caribbean and herbivores only 10%, the herbivores comprise around 40% of total fish biomass. However, large to intermediate-sized herbivores are not a preferred prey for the larger piscivorous fishes (Opitz 1996). Much of the literature that has been reviewed includes listing of species observed in the study areas but fail to provide information of the life stage of the individuals seen. Often there are no data on the size of the fish, an important variable in determining whether the fish are juveniles or adults. Since 2004, some research projects have addressed this issue, including the SEAMAP 2007 surveys, and the NOAA Center for Coastal Monitoring and Assessment's Biogeography Branch characterizations of fish assemblages in the USVI (Zitello et al., 2009), by measuring or estimating fish lengths and calculating length-frequency distributions by species.

Regarding habitat use by species in the Fishery Management Unit, the 2004 EFH-FEIS noted there is little information on the distribution of reef fish eggs and larvae, but most of the species have planktonic eggs. Also largely unknown, are the distribution, development, settlement, and development of fish larvae. In general, newly settled stages tend to occur at depths of 0-10 m, and primarily at 5-10 m. Grouper species may be less likely than snapper species to have local larval retention due to their longer larval duration. Based on their size and age at settlement, grunts may be considered one of the reef fish groups most likely to exhibit local retention. However, larval duration, larval behavior, variations in current patterns and other factors may play a role in determining the amount of local retention (Lindeman et al. 2000). Some of these factors have now been studied in the U.S. Caribbean, although large gaps still exist in the knowledge of reef fish eggs and larvae.

Many species of reef fishes utilize seagrass and mangrove habitats as juveniles, and then migrate to reef areas as they grow larger, showing a clear ontogenetic migration pattern. A large percentage of the demersal stages of reef fish species also exhibit a cross-shelf migration to deeper waters as ontogeny progresses. Some reef fishes have been found to use shallow reef areas as juveniles, and then move to deeper reef areas as they mature (Lindeman et al. 2000). Ontogenetic migration patterns of reef fishes across mangrove, seagrass, and shallow and mesophotic reef habitats have been examined since 2004. New, detailed information on habitat utilization patterns by some reef-fish species at certain life-history stages may justify more precise definitions of EFH (eg., grunts, snappers, parrotfishes).

Adult reef fish habitat was described in the 2004 EFH-FEIS noting that most commercially important reef fish in the Greater Caribbean area (e.g. groupers and snappers) migrate to specific places at specific times to reproduce in spawning aggregations (SPAGs). Many documented SPAG sites occur largely at reef promontories, and/or the seaward extension of reefs near deep water. In regions where no SPAG fishing has been documented, locations of promontories and reef extension may predict the location of SPAG. Reef fish spawning sites tend to occur near the edge of outer reefs or reef passes over hard sand bottom at depths around 20-50 m. SPAGs are critically important in the life cycle of many reef fishes and reproduction at these sites often represents the total annual reproductive output for specific stocks of a species (Heyman et al. 2002, Claro and Lindeman 2003).

III. ANALYSIS OF THE EFFECTS OF THE PROPOSED ACTIONS

This section will provide an analysis of the potential effects of spilled oil, dispersants and dispersed oil, in-situ burning, and the response operations associated with both dispersants and in-situ burning, on listed species, designated critical habitat, and EFH.

A. Potential Effects of Oil

When released into the aquatic environment, crude oil tends to form a thin layer, < 1 mm thick on average (Lee et al., 2011a) and typically ~0.1 mm (NRC, 2005), that spreads over the surface of the water; after oil is spilled, a number of physical, chemical, and biological factors affect its dispersion and ultimate fate (NRC, 2005). Physical factors such as surface tension (a measure of attraction between the molecules of a liquid), density, and viscosity (a measure of resistance to flow) cause the oil molecules to generally stay together, if there are no other forces at work (NRC, 2005). A chemical dispersant can cause an oil slick to either spread rapidly and then disperse, or to spread slowly through "herding" (NRC, 2005), after which additional dispersant applications may be required to remove the oil slick from the ocean's surface.

Wind, waves, and other physical forces can either enhance dispersion or mix the oil and water, forming an emulsion that remains relatively cohesive and does not disperse easily (NRC, 2005; MMS, 2010; Brandvik et al., 2010). Over time, chemical processes (e.g., volatilization and oxidation) can change the makeup and density of oil, which affects, in turn, its fate in the environment (Mackay and McAuliffe, 1988). Biodegradation occurs over time, as fractions of the oil become bioavailable (i.e., dissolve in the water column) (Prince et al., 2013); however, oil thickness, cohesiveness, viscosity, and other factors affect bacterial access to oil molecules (Prince et al., 2003).

Dispersion is a natural process that distributes petroleum at the ocean's surface into the water column over time, resulting in many small droplets that may or may not resurface and coalesce with the oil slick (NRC, 2005). This process can be very slow under natural conditions, but the addition of chemical dispersants greatly increases the rate of dispersion (NRC, 2005).

If zooplankton, fish, and other water column or benthic organisms become oiled or accumulate oil in their tissues, they could ultimately expose species that prey upon them. Marine mammals, except the manatee, are carnivores that rely on invertebrates or fish for sustenance. Several sea turtle species that occur in the area under consideration for action also prey on aquatic invertebrates and fish. Prey species that occur in open waters further from shore where dispersant use will be pre-authorized are the primary concern. Those that occur in nearshore areas where dispersant use will not be pre-authorized by the LOAs are unlikely to be impacted.

1. Effects of Oil on Species

a. Sea Turtles: The impacts of oil on marine reptiles have been studied to a lesser extent than the impacts on other groups. Oil is known to cause mortality in sea turtles, as evidenced by strandings of dead individuals after the DWH oil spill (Barron, 2012). However, it is noteworthy that according to NOAA records, for the 126 major oil spills which occurred between 1967 and 2001, the case history files indicate that few incidents reported oiling, contaminating, or killing of sea turtles or oiling of nesting sites (Shigenaka, 2003). Either turtles were rarely impacted or the historical files are not sufficiently complete or detailed to document injuries, protection strategies, or rehabilitations.

As with the early life stages of other species, mortality is likely related to PAHs in oil, which have been shown to significantly impact turtle embryos and hatchlings (Albers and Loughlin, 2003; Van Meter et al., 2006). Other noted impacts include effects on respiration, skin, blood chemistry, and salt gland functioning (Albers and Loughlin, 2003). Turtles are especially susceptible to oil spills that foul nesting areas (ITOPF, 2011).

Juvenile and adult turtles can be exposed to spilled oil when feeding, surfacing to breathe, or nesting in areas contaminated by stranded oil. Turtles are also susceptible to floating tarballs formed from weathered oil. There is no firm evidence that sea turtles are able to detect and avoid oil (Odell and MacMurray, 1986). Studies indicate oil exposure can have several adverse effects on turtles, including toxic responses to vapor inhalation or ingestion, skin irritation, interference with osmoregulation and ion balance, and reduced hatching success (Van Fleet and Pauly, 1987; Fritts and McGehee, 1982; Lutz and Lutcavage, 1989). Experiments on adult loggerhead turtles conducted by Lutcavage et al. (1993) showed that major body systems in marine turtles are adversely affected by even short exposures to weathered South Louisiana crude oil. Effects observed included alteration of blood chemistry, alteration of respiration and diving patterns, interference with salt gland function, and skin lesions. Exposure to fresh oil would likely be considerably more harmful. Though oil exposure may not directly kill adult turtles, the effects may make them more vulnerable to predation or disease.

Oiling of sea turtle nesting habitat poses a potential risk to adult nesting turtles, hatchlings, and to eggs. Turtle embryos are particularly sensitive. The effects of oil on the development and survival of marine turtles appears to be variable, depending on such factors as stage of nesting, oil type, degree of oil weathering, and amount and height of oil deposition on the beach. Studies by Fritts and McGehee (1982) indicate that fresh oil washing ashore to the level where nests with incubating eggs are located may result in extensive embryo mortality. The studies found that mortality may not be significant if eggs are deposited in sand after contamination has occurred and the oil has weathered, although hatchlings may be smaller than normal. Some evidence suggests olfactory cues are imprinted on sea turtles as hatchlings and guide them back to their natal beaches for nesting when they reach maturity. Oil on the beach could interfere with these chemical guides (Lutz et al., 1989). Response activities to clean oil stranded on beaches may pose an addition risk of injury to eggs, hatchlings, and nesting adults.

b. Marine Mammals (Cetaceans): Cetaceans spend considerable time at the surface swimming, breathing, feeding, or resting and so are at risk of exposure to a surface oil slick, water-in-oil emulsion, or tar balls. Although there is evidence that some cetacean species are able to detect oil, they do not always avoid it. Oil spills can be intense anthropogenic stressors for marine mammals. Dolphin health assessment undertaken in the wake of the DWH oil spill revealed a high prevalence of moderate to severe lung disease and evidence of hypoadrenocorticism in dolphins (Schwacke et al., 2013). These physical impacts are consistent with adverse health effects observed following oil exposure on other mammal species (Schwacke et al., 2013). A study following the March 2014 oil spill in the Galveston ship channel determined that common bottlenose dolphins, (*Tursiops truncatus*) are susceptible to health complications from both oil ingestion and the inhalation of toxic fumes from oil slicks. Bottlenose dolphins try to avoid thicker oils, but have difficulty avoiding thinner, lighter sheen oils. Potential behavioral responses to the spill did not appear overtly negative or prolonged (Mattson et al, 2015). The volatile fraction of crude oil contains many toxic hydrocarbons that evaporate and can create hazardous air concentrations in the vicinity of a spill (Allen and Ferek, 1993).

The most serious potential risk to cetaceans appears to be inhalation of these toxic vapors, which can cause inflammation of mucous membranes of the eyes and airways, lung congestion, and possibly pneumonia. At very high exposure levels, volatile hydrocarbons can potentially result in neurological disorders and liver damage. Effects from direct contact or ingestion of oil are generally temporary and of less concern for cetaceans. Oil is unlikely to adhere to the surface of their skin, which is also relatively impermeable to the oil's toxic components. Baleen plates of skim-feeding baleen whales may become fouled by oil on the water surface, temporarily interfering with feeding. For a few days or weeks, hydrocarbons or their metabolites in exposed marine invertebrates could be transferred to cetaceans preying upon them. This exposure would likely be short-term and is not expected to result in serious effects (Geraci and St. Aubin, 1990). Benthic invertebrates accumulating residues from contaminated sediments could provide a potential source of longer-term exposure to bottom-feeding cetaceans. Cetaceans might also be indirectly affected if an oil spill resulted in destruction or significant shifts in the distribution of key prey species populations.

c. Fish

The exposure of fish to oil (and its component chemicals) appears to occur predominately across the gill surface or through ingestion of contaminated food (Baussant et al., 2001; Cohen et al., 2001; Milinkovitch et al., 2011b). If exposed continuously to PAHs dissolved in the water column, oil may require as many as seven days to reach a maximum concentration in fish (Logan, 2007). The more soluble components of oil (e.g., low-molecular-weight PAHs [LPAHs]) are internalized across the gills more efficiently than the larger molecules, resulting in a greater exposure to LPAHs than to high-molecular-weight PAHs (HPAHs) over short time periods (Baussant et al., 2001; Cohen et al., 2001; Wolfe et al., 2001). HPAHs may be quickly and efficiently metabolized and depurated from some fish (e.g., turbot) (Baussant et al., 2001), whereas they are concentrated in invertebrates (e.g., *Mytilus edulus*) (Baussant et al., 2001). Due to the rapid depuration of the LPAHs, Wolfe et al. (2001) did not find a significant increase in the accumulation of an LPAH (i.e., naphthalene) or its metabolites after 12 hours of depuration in larval topsmelt.

HPAHs, which fish can also internalize across the gills, are metabolized and excreted from the fish body at a slower rate than LPAHs (Logan, 2007; Payne et al., 2003); their solubility also increases after dispersant application, resulting in greater exposure for fish to HPAHs than after exposure to untreated crude oil (Couillard et al., 2005; Cohen et al., 2001). HPAH accumulation is more strongly correlated with enzymatic responses indicative of metabolism in fish (and subsequent exposure to toxic PAH metabolites) (Couillard et al., 2005). The correlation between HPAH exposure and metabolic activity further indicates that these chemicals are efficiently metabolized to forms that can be removed from the body, limiting trophic transfer.[9]

Similarly, the accumulation of oil and its components in invertebrates, which is enhanced by the addition of chemical dispersants (Wolfe et al., 1998; Jensen et al., 2011), can influence uptake in fish species through ingestion. Ingestion of contaminated food appears to be more important in the exposure of fish to HPAHs, because lipids in prey items, particularly invertebrates, accumulate organic, lipophilic compounds such as HPAHs (Logan, 2007). However, the apparent exposure of fish to HPAHs when fed dispersed oil-contaminated prey was not significantly different from the exposure of fish fed crude oil-contaminated prey (Cohen et al., 2001). Wolfe et al. (2001) reported a similar result for the accumulation of naphthalene and its metabolites in larval topsmelt exposed to both contaminated food and exposure solution.

Reported individual-level impacts on fish include abnormal growth, reduced growth (Claireaux et al., 2013; Couillard et al., 2005), reduced hatch (Greer et al., 2012; Anderson et al., 2009), and mortality (Van Scoy et al., 2012). An additional impact of note is the onset of blue sac disease, which was observed in Atlantic herring (*Clupea harengus*) by Greer et al. (2012). However, Greer et al. (2012) showed that dispersion reduced the acute toxicity of oil to Atlantic herring embryos 5, 30, and 60 minutes post-dispersion, even though blue sac disease had been induced.[10] This disease has been observed in fish exposed to either oil alone or dispersed oil (Greer et al., 2012; Colavecchia et al., 2006). Reduced acute toxicity in Chinook salmon was observed by both Lin et al. (2009) and Van Scoy et al. (2010). Therefore, the impact of chemical dispersion on oil toxicity to fish is uncertain, although likely to be enhanced in embryonic and larval life stages in planktonic fish species (e.g., Pacific herring).

In addition to causing internal impacts, dispersed oil affects transfer across the gills of fish (Singer et al., 1996), particularly by affecting Na^+/K^+-ATPase pumps (Duarte et al., 2010), which are necessary for regulating ionic and osmotic gradients in fish tissues. Duarte et al. (2010) showed that the flux of ions across fish gills significantly increased (both influx and efflux), and that the net flux significantly decreased, such that more sodium was lost from the gill surface, when fish were exposed to dispersed oil, relative to the control, dispersant-only, or oil-only treatments. Such a disruption could lead to increased stress in fish. However, the effect does not directly relate to an impact at the individual level.

Although bioaccumulation of PAHs has been shown to occur in fish over short time periods, efficient metabolic processes limit the bioconcentration of PAHs in fish tissues over time (Logan, 2007; Payne et al., 2003) and the transfer of parent PAHs from fish to higher trophic levels (i.e., birds and mammals) (Payne et al., 2003; Albers and Loughlin, 2003). The transfer or bioconcentration of PAH metabolites in higher trophic levels has not been

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[9] HPAHs are known to be broken down into much more toxic metabolites prior to egestion, and metabolites have been linked to various sublethal impacts on fish (Logan, 2007; Payne et al., 2003). Although PAHs are actively metabolized and excreted, it is not implied here that sublethal impacts will not result.

[10] Solution collected 15 minutes post-dispersion from the wave tanks where dispersion was conducted was more toxic than oil alone (Greer et al., 2012); it is unclear why this duration resulted in a conflicting result.

extensively studied; it is possible that metabolites stored in fish lipids could be transferred to higher trophic levels, resulting in PAH-related toxicity in those species.

An increased concentration of small oil droplets is environmentally relevant because the exposure of water column biota to particulate oil droplets may have a different mode of toxicity (e.g., physical coating of body surfaces, gill uptake, ingestion) than dissolved oil (primarily narcosis). Oil droplets are within the normal food size range of naturally occurring food particles ingested by suspension feeding zooplankton (Berggreen et al. 1988; Conover 1971; Paffenhöfer 1984), and can adhere to feeding appendages interfering with normal feeding (Hansen et al. 2009). In addition, these zooplankton can be prey items for larger animals (Bejarano et al, 2013).

The chemical profile of crude oil will change over time due to natural weathering, a process that is influenced by environmental factors such as salinity. Natural weathering and the addition of chemical dispersants increase the rate of dissolution of oil into the water column. Consequently, they increase the rate of biodegradation, affecting the bioavailability of the toxic components of crude oil to aquatic organisms. Although little is known about the mechanisms of toxicity for many of the chemical components of crude oil, recent studies have demonstrated that different constituents in crude oil will elicit different toxicological effects in aquatic organisms. In a study by Brown et al (2015), hatching success reductions and developmental delays were observed in embryos exposed to oil-dispersed treatment waters weathered up to 4 weeks, with toxicity decreasing over time. Embryos incubated in 4 week weathered oil in 12 g/L seawater exhibited decreased hatching success and increased time to hatch, suggesting a time- and salinity- dependent mechanism of toxicity different from oil-dispersed waters. These data demonstrate how environmental conditions can influence the biodegradation of crude oil, and, in turn, affect biological responses (Brown et al, 2015).

Variability in fish recruitment to adult populations is largely related to variability in survival during early life. In addition to direct mortality (e.g., toxicity), the release of oil and application of dispersant after the DWH oil spill may have impacted larval fish survival by disrupting planktonic food webs. Analyses of zooplankton suggest many potential larval fish prey were abundant, and assemblages were largely resilient to DWH oil spill impacts (Hernandez et al, 2015).

Lessons from past large-scale oil spills suggest that chronic impacts on ecosystem services may not be apparent for years following a given spill, but also that the existence of pre-spill baseline data is critical for evaluating acute as well as chronic impacts. Reef fishes have been examined as indicator species of ecological impacts of the 2010 DWH in the northern Gulf of Mexico due to extensive pre-spill data on their community structure, feeding ecology, and population dynamics. Liver PAH concentrations and enzyme activities clearly indicate that reef fishes were exposed to petroleum hydrocarbons in the weeks to months following the DWH spill, although exact mechanisms of exposure remain unclear. Acute impacts include declines in reef fish numbers and biomass on natural and artificial reefs across the northern Gulf of Mexico shelf study region, with the most severe declines observed in small demersal fishes, such as damselfishes, cardinalfishes, and wrasses. Chronic effects include food web impacts, dietary shifts, and lower growth rates in the years following the spill. By the fourth year post-DWH, fish communities were showing signs of resiliency. Furthermore, evidence from stable isotope analysis of reef fish muscle tissue indicates food web effects of the spill persisted at least into 2014 (Patterson III et al, 2015).

Sharks are a valuable part of marine ecosystems, where they serve as top predators that help maintain balance. More than a dozen species—including the threatened scalloped hammerhead, which has severely declined over the past several decades—are present in the Caribbean. Oil spills degrade sharks' habitat and injure them through direct contact or when they consume contaminated prey. Sharks are generally unable to adapt to rapidly changing conditions; therefore, recovery from an oil spill can take decades.

More than 2 million tons of oil enter the marine environment each year. Apart from about 15% from natural oil seeps, a major source is runoff from terrestrial uses. Other sources are discharges from tankers and shipping along major routes, discharges from production platforms, storage facilities and refineries and accidental events such as oil spills and rupture of pipelines. Although not identified as a major problem in open waters, hydrocarbons and other toxicants in oil can impact sharks and other fish either through direct contact or via the food chain. Impacts on sharks from oil spills are most likely through the effects on vulnerable and sensitive coastal seagrass, mangrove, salt marsh, coral reef, and rocky reef.

d. Corals

Shallow water coral reef ecosystems have an elevated chance of exposure to hydrocarbons due to their close proximity to the coastline. Previous research to evaluate hydrocarbon toxicity to corals and coral reefs has generally focused on community level effects, and results are often not comparable between studies. Thus, a significant data gap exists on the toxicity thresholds of hydrocarbons to corals, from the organismal to cellular level (Renegar et al. 2015).

There are three primary modes of exposure for coral reefs in oil spills. In some areas, direct contact is possible when surface oil is deposited on intertidal corals. Presuming that some portion of spilled oil will enter the water column either as a dissolved fraction or suspended in small droplets, this potential exposure pathway must be considered in most cases. Subsurface oil is a possibility in some spills, particularly if the spilled product is heavy, with a density approaching or exceeding that of seawater, and if conditions permit oil to mix with sediment material to further increase density.

Evaluation of risk based on exposure pathway is a complex calculus that is highly spill-dependent. Relevant questions that feed into the determination are linked to the considerations above and include:

- Are corals in the affected area intertidal?
- At what depths do the subtidal corals occur?
- Does this spilled oil have a component of lighter, more water-soluble compounds?
- Will sea conditions mix oil on the surface into the water column?
- Is there a heavier component to the oil that raises the possibility of a density increase through weathering and association with sediment that could sink the oil to the bottom?

- Will residual oil on adjacent shorelines (e.g., mangrove forests) provide a source of chronic oil exposure?

  Areas with intertidal corals could be considered at greatest risk in a spill because of the increased potential for direct contact with a relatively fresh oil slick. Regardless of differences in susceptibility by species or physical form, direct oil contact is most likely to result in acute impact because in this kind of exposure scenario the oil is fresher, with a greater proportion of more toxic lighter aromatic hydrocarbons.

  Coral exposure via the water column can be a serious route under some circumstances. Because much of the constituent material in oil has a relatively low solubility in water, in general coral may be protected from exposure by overlying waters. However, if rough seas and a lighter, more soluble product are involved, subtidal corals may experience harmful exposure when oil mixes into the water column. The absolute levels of exposure would be expected to be much lower than those encountered by direct contact with intertidal slicks, since only a small fraction of the total oil can mix into the water column either in solution or physically suspended. However, the components of the oil mix most likely to enter the water column are those generally considered to be most acutely toxic. Corals may therefore be exposed to "clouds" of naturally dispersed oil driven into the water column under turbulent conditions, with impacts dependent on exposure concentrations and length of exposure.

  Smaller spill events, such as the fishing vessel grounding at the Rose Atoll National Wildlife Refuge in American Samoa described in Shigenaka (2001) that resulted in the release of diesel fuel over 6 weeks, are reported to affect coralline algae and many sessile benthic invertebrates, as well as lead to a shift in the structure of algal communities. Four years after the grounding, the affected areas remained visibly impacted (Shigenaka 2001). Overall, corals were found to be most affected if they are in shallow areas where direct contact with the petroleum product occurs but all researchers studying chronic effects documented sublethal changes in exposed corals in some form (Shigenaka 2001). Polycyclic aromatic hydrocarbons (PAHs) are known to accumulate in the living tissues of corals and in zooxanthellae (Pait et al. 2007).

  Bak and Elgershuizen (1976) used the ability of corals to clear sediment as an endpoint for sublethal oil toxicity. Research Planning, Inc. (1986) suggested that the sensitivity of different coral species to sedimentation and turbidity could be used as a corollary to sensitivity to oil. *Acropora palmata* was found to be highly sensitive and the *Orbicella annularis* complex was found to have an intermediate sensitivity (Shigenaka 2001).

2. Effects of Oil on Habitat

a. Coral

Where significant amounts of oil remain on adjacent shorelines after termination of cleanup efforts, such as in mangrove forests, where inaccessible oil is left for natural removal, coral reefs may be affected by chronic oil exposure. There could also be episodic releases of higher amounts of oil during high-energy events. This process was one factor in the long-term effects of the 1986 Bahía Las Minas crude oil spill in Panama.

Heavier fuel oils contain fewer of the light fractions identified with acute toxicity than do refined and crude oils (although these bunker type oils are sometimes "cut" with lighter materials to facilitate loading and transfer). If they remain on the water surface, spills of heavier fuel oils are less of a concern from a reef perspective, but more of a concern for protection of other habitats like mangrove forests where they can strand and persist for long periods of time, indirectly affecting coral reefs by chronic exposures as the oil is removed by natural processes. However, the heavy oils can also weather or mix with sediment material and increase in density to the point where they may actually sink—which provides a direct route of exposure to subtidal corals.

In the Caribbean, shorelines adjacent to coral reefs often contain mangrove forests, which are highly sensitive to oil. At many tropical oil spills, the most severe impact is to mangroves, sea turtle nesting areas, or other sensitive shoreline habitats. Recovery of severely impacted mangrove forests takes decades and often has associated impacts, including shoreline erosion, decimated marine life nursery areas, or chronic leaching of oil from contaminated sediments (see Bahía las Minas case study). When conditions are appropriate to minimize impacts to coral, using dispersants or in-situ burning on floating oil can be an effective and environmentally advantageous cleanup strategy that reduces long-term impacts.

Coral communities may recover more rapidly from oil exposure alone than from mechanical damage. Coral reefs exposed to crude oil and chemically dispersed crude oil in field experiments conducted in Panama showed recovery, with no significant differences between the exposed and control sites, after ten years. Short-term bioassays of corals exposed to oil have revealed temporary effects followed by recovery generally within one week.

Recovery of coral reefs after oil exposure, however, may depend partly on the recovery of associated communities that may be more seriously affected, such as mangroves and seagrass beds. For example, the 1986 Bahía las Minas oil spill in Panama impacted mangrove and seagrass communities, as well as corals. Death and injury of these habitat-structuring organisms physically destroyed habitats. The secondary biological effects of erosion and redeposition of oily sediments included greatly increased levels of injuries and decreased growth and sexual reproduction for surviving subtidal reef corals in Bahía Las Minas compared to coral reefs outside the bay. The entire Bahía Las Minas ecosystem became more vulnerable to subsequent natural or anthropogenic disturbances. It has been estimated that recovery of the dominant reef-building corals may require at least a century or more to reach the size of many of the colonies killed by this spill.

b. Seagrass

The TROPICS (Tropical Investigations in Coastal Systems) field study began in 1983-84 near Bocas del Toro, Panama. The study was designed to examine the relative short and long-term effects of dispersed crude oil versus non-dispersed crude oil on tropical marine ecosystems. After baseline studies in 1983, two 900 m^2 sites composed of intertidal mangrove and subtidal seagrass-coral zones were dosed in 1984 with untreated Prudhoe Bay crude oil and Prudhoe Bay crude oil dispersed with Corexit® 9527. At periodic intervals over 25 years, the sites were monitored and effects were compared to a nearby reference site. A number of papers were published during the study period.

Seagrass beds of *Thalassia testudinum* at the non-dispersed site had been overrun by finger coral (*Porites porites* a.k.a. *P. furcata*), a condition not found at the dispersed site or reference site. Core samples indicated elevated levels of aromatic hydrocarbons (naphthalene) at the non-dispersed site (DeMicco et al, 2011).

Seagrass declined, ending at 58% original coverage at the non-dispersed site. With the exception of Ward (2003), other seagrass metrics besides density were not pursued after 1994 due to time and funding constraints. Reviewing the TROPICS field notebooks, there are various measures of area and depths which indicate that the subtidal shelf off the non-dispersed site became narrower and deeper over time. For example, the average width of the non-dispersed site was 7.8m in 1984 (Baca unpublished). In contrast, measures recorded by Ward in 2001 (unpublished) showed widths of 1.5m for the non-dispersed site. This would indicate a loss of 6.3m at the non-dispersed site, a probable measure of impacts (DeMicco et al, 2011).

c. Mangroves

Observations from many spill events around the world have shown that mangroves suffer both lethal and sublethal effects from oil exposure. Past experience has also taught us that such forests are particularly difficult to protect and clean up once a spill has occurred because they are physically intricate, relatively hard to access, and inhospitable to humans. In the rankings of coastal areas in NOAA's Environmental Sensitivity Indices, commonly used as a tool for spill contingency planning around the world, mangrove forests are ranked as the most sensitive of tropical habitats. NOAA recently updated the job aid on Oil Spills in Mangroves Planning & Response Considerations (Hoff and Michel, 2014), which provides a comprehensive summary of the impacts of oil and the effects of various response options, including dispersing or burning offshore to reduce impacts to mangrove-dominated shorelines.

It is clear from spills, and field and laboratory studies, that oil can harm or kill mangroves. What is less obvious is how that harm occurs and the mechanism of toxicity. Although there is some consensus that oil causes physical suffocation and toxicological/physiological impacts, researchers disagree as to the relative contributions of each mechanism, which may vary with type of oil and time since the spill (Proffitt et al. 1997).

Similar to the oil toxicity situation for many other intertidal environments, the mangrove-related biological resources at risk in a spill situation can be affected in at least two principal ways: first, from physical effects; second, the true toxicological effects of the petroleum.

Many oil products are highly viscous. In particular, crude oils and heavy fuel oils can be deposited on shorelines and shoreline resources in thick, sticky layers that may either disrupt or completely prevent normal biological processes of exchange with the environment. Even if a petroleum product is not especially toxic in its own right, when oil physically covers plants and animals, they may die from suffocation, starvation, or other physical interference with normal physiological function.

Mangroves have developed a complex series of physiological mechanisms to enable them to survive in a low-oxygen, high-salinity world. A major point to remember in terms of physical effects of oil spills on mangroves is that many, if not most, of these adaptations depend on unimpeded exchange with either water or air. Pneumatophores and their lenticels tend to be located in the same portions of the intertidal most heavily impacted by stranded oil. While coatings of oil can also interfere with salt exchange, the leaves and submerged roots of the mangrove responsible for mediation of salts are often located away from the tidally influenced (and most likely to be oiled) portions of the plant. Thus, salt mediation is less susceptible to impacts from oil than are respiratory functions occurring at the air-water interface.

These physical impacts of oil are linked to adaptive physiology of the mangrove plants, but are independent of any inherent chemical toxicity in the oil itself. The additional impact from acute or chronic toxicity of the oil would exacerbate the influence of physical smothering. Although many studies and reviews of mangroves and oil indicate that physical mechanisms are the primary means by which oil adversely affects mangroves, other reviewers and mangrove experts discount this weighting. See, for example, Snedaker et al. (1997). They suggest that at least some species can tolerate or accommodate exposure to moderate amounts of oil on breathing roots.

The acute toxicity of oil to mangroves has been clearly shown in laboratory and field experiments, as well as observed after actual spills. Seedlings and saplings, in particular, are susceptible to oil exposure: in field studies with *Avicennia marina*, greater than 96% of seedlings exposed to a weathered crude oil died, compared to no deaths among the unoiled controls (Grant et al. 1993). Other studies found that mangrove seedlings could survive in oiled sediments up to the point where food reserves stored in propagules were exhausted, whereupon the plants died.

The *Avicennia* study cited above also found that fresh crude oil was more toxic than weathered crude. Based on laboratory and field oiling experiments in Australia, the authors cautioned against readily extrapolating results from the laboratory to what could be expected during an actual spill. Container size and adherence of oil to container walls were thought to be important factors that may have skewed laboratory toxicity results by lowering actual exposure concentrations (Grant et al. 1993).

Obvious signs of mangrove stress often begin occurring within the first two weeks of a spill event, and these can range from chlorosis to defoliation to tree death. In the 1999 Roosevelt Roads Naval Air Station (Puerto Rico) spill of JP-5 jet fuel, an initial damage assessment survey conducted in the first month post-spill determined that 46% of mangrove trees, saplings, and seedlings along a transect in the most impacted basin area were stressed (defined as showing yellowed, or chlorotic, leaf color). This compared to 0% along the unoiled reference transect (Geo-Marine, Inc. 2000). Color infrared, aerial photography taken at regular intervals through 19 months post-spill confirmed the visual observations. Analysis of the infrared photographs of the affected mangrove area indicated that two weeks after the release, 82% of the total mangrove area was classified as "impacted" relative to pre-spill conditions. Under more controlled conditions, studies using fresh crude oils have suggested that defoliation, when it occurs, should reach a maximum between 4-12 weeks post-spill.

Mangroves can be chronically impacted by oil in several ways. Stressed mangroves could show differences in growth rates or altered reproductive timing or strategy. They may also develop morphological adaptations to help them survive either the physical or chemical consequences of residual oil contamination. Such modifications may require expending additional energy, which in turn, could reduce the mangroves' ability to withstand other non-spill-related stresses they may encounter. One consequence of the complex physical structure and habitat created by mangrove trees is that oil spilled into the environment is very difficult to clean up. The challenge and cost of doing so, and the remote locations of many mangrove forests, often results in unrecovered oil in mangrove areas affected by spills. This, in turn, may expose the trees and other components of the mangrove community to chronic releases of petroleum as the oil slowly leaches from the substrate, particularly where organic-rich soils are heavily oiled. Interestingly, the presence and density of burrowing animals like crabs also affects the persistence of oil in mangrove areas and can determine whether an exposure is short- or long-term, because of oil penetration via burrows into an otherwise impermeable sediment.

The experimental (i.e., intentional and controlled) 1984 TROPICS spill in Panama confirmed long-term impacts to oiled mangroves, termed "devastating" by the original researchers who returned to the study sites ten years later. They found a total mortality of nearly half of the affected trees and a significant subsidence of the underlying sediment. This was compared to a 17-percent mortality at seven months post-oiling, a level that appeared to be stable after 20 months (Dodge et al. 1995). Over the decades, the dead trees at the oiled site were slowly replaced by "waves" of seedlings because early ones did not survive to produce trees (Baca et al. 2014). In 2009, 25 years after oiling, the oiled site exhibited a decline of the mangroves (there were 1,085 small trees), whereas the dispersed and reference sites remained at baseline levels (with 124 and 392 small trees, respectively) (DiMicco et al. 2011). In 2013, 29 years later, the counts of adult trees had recovered though there were still an abundance of small trees at the oiled site, and curling and distortions of prop roots in small trees and seedlings were noted at the oiled site, but not at the other sites (Baca et al. 2014).

With the realization that mangrove stands provide key habitat and nursery areas for many plants and animals in the tropical coastal environment, many researchers have included the associated biological communities in their assessments of oil impacts. Of course, this considerably broadens the scope of spill-related studies, but realistically, it would be arbitrary and artificial to consider only the impacts of oil on the mangroves themselves.

Studies of the Bahía las Minas spill in Panama concluded that significant long-term impacts occurred to mangrove communities. Both the habitat itself and the epibiotic community changed in oiled areas. After five years, the length of shoreline fringed by mangroves had decreased in oiled areas relative to unoiled areas, and this translated to a decrease in available surface area ranging from 33 to 74 %, depending on habitat type. In addition, defoliation increased the amount of light reaching the lower portions of the mangrove forest (Burns et al. 1993).

Some studies have looked at the toxicity of undispersed and dispersed oil to both mangroves and the associated invertebrate community. The limited findings are somewhat equivocal: one study found that dispersing oil appears to reduce the inherent toxicity of the oil to mangroves, but increases the impacts to exposed invertebrates (Lai 1986). Another assessment concluded no difference in toxicity to crustaceans from dispersed and undispersed crude oil (Duke et al. 2000). However, the same study also evaluated toxicity of Bunker C fuel oil and found that crude oil was more acutely toxic than the Bunker C fuel oil. The authors attributed this to the physical and chemical differences between the oil types.

Australian researchers studying the effects of the 1992 Era spill on fish populations around oiled mangroves found no measurable assemblage differences between groups inside and outside oiled zones, although juveniles of several species were significantly smaller in oiled creeks than in unoiled creeks (Connolly and Jones 1996).

d. Stranded Oil Behavior in Mangroves

Mangroves grow in low-energy depositional areas, which also tend to be the sites where oil accumulates. Spilled oil is carried into mangrove forests by winds and tidal currents. Oil slicks generally move into mangrove forests when the tide is high, depositing on the soil surface and on aerial roots and propagules when the tide recedes. The resulting distribution of deposited oil is typically patchy due to the variability in tidal heights within the forest. If there is a berm or shoreline present, oil tends to concentrate and penetrate into the berm or accumulated detrital wrack–organic material, usually from dead seagrass or algae that washes up on shorelines. The oil can penetrate into the soil, particularly through burrows and other voids like those formed by dead mangrove roots. Lighter oils tend to penetrate more deeply into mangrove forests than heavier and more weathered oils, but will not persist unless they mix into the soil. However, crude oils and heavier refined products can pool onto sediment surfaces and can be highly persistent. These heavy oils and emulsified oil can be trapped in thickets of red mangrove prop roots and black mangrove pneumatophores and are likely to adhere to and coat these surfaces, as well as other organic materials, such as seagrass wrack. Re-oiling from resuspended oil, particularly as tides rise and fall, may further injure plants over time. Where oil persists, sheens may be generated for months or years.

B. Potential Impacts of Dispersants and Dispersed Oil

A primary objective of an oil spill response is to quickly remove as much oil as possible from the surface of the water, thereby minimizing direct contact with wildlife and preventing movement of the oil into nearshore and shoreline areas where removal is more difficult and environmental impacts severe. Dispersants, applied under appropriate conditions, may offer the best response option to help achieve this objective. Dispersion of oil at sea, before a slick washes ashore, reduces the overall and particularly the chronic impacts of oil on sensitive inshore habitats including salt marshes, coral reefs, sea grasses, and mangroves. Dispersed oil is less likely than a surface slick to reach shoreline areas. Any dispersed oil that does move inshore is less likely to stick to shorelines and vegetation because dispersants alter the adhering property of oil droplets. Consequently, habitats recover faster if the oil is dispersed before it reaches them (NRC, 1989, 2005). By protecting nearshore and shoreline habitats from contamination, dispersant use benefits listed species and other wildlife that rely on them including sea turtles and cetaceans.

Dispersants are not intended to be applied to wildlife at all, neither directly nor indirectly; therefore, concentrated exposure to dispersants is not expected as a result of their application. Under the CRRT's dispersant guidance documents, dispersants will not be applied near ESA-listed species or other wildlife. Data indicate that dispersant use alone is unlikely to contribute significantly to adverse biological effects in the water column. Within the normal range of operating dosages, biological effects are due to the dispersed oil, not the dispersant (NRC, 1989; 2005).

Dispersants are known to have a variety of effects on aquatic species. However, the toxicities of various dispersants (e.g. Corexit® 9500 and Corexit® 9527) are known to be less than that of crude oil alone (Fingas, 2008; NRC, 2005); conversely, some have shown dispersed oil to be more toxic than either oil or dispersants alone (Fingas, 2008; NRC, 2005). Therefore, the impacts of dispersed oil are caused primarily by the toxicity of oil, and may be enhanced by its interaction with dispersants. The enhanced toxicity of dispersed oil (over oil alone) is frequently attributed to the increased bioavailability of the toxic components of oil, principally PAHs (Wolfe et al., 1998; Wolfe et al., 2001; Yamada et al., 2003; Ramachandran et al., 2004; Milinkovitch et al., 2011a). Dispersants have been shown to increase the acute toxicity (e.g., lethality) of oil in only about half of the comparable studies; the other half of these studies showed that chemical dispersants actually decrease the lethality of oil in a mixture. It appears that the acute lethality of oil is generally decreased by chemical dispersants.

The sublethal impacts of dispersed oil are generally enhanced relative to those of oil alone, suggesting that an immediate response to dispersed oil exposure is generally less likely than a delayed response (e.g., decreased fitness leading to death). Due to diminishing concentrations of dissolved and dispersed components of oil in the water column over time, long-term impacts are unlikely within an area. Observed impacts (i.e., toxicity endpoints) of chronic exposure to PAHs include genotoxicity, immunotoxicity, histopathological impacts (e.g., hepatic lesions), behavioral impacts, and reproductive impacts (Payne et al., 2003; Albers and Loughlin, 2003; Malcolm and Shore, 2003; Besten et al., 2003; Meador, 2003; Barron, 2012; Godschalk et al., 2000; Lemiere et al., 2005; Carls et al., 1999; Jonsson et al., 2010). The likelihood of such impacts affecting listed species as a result of short-term exposure is a point of uncertainty, although the rapid reduction in exposure concentrations and biodegradation of dispersed oil within a relatively short time period may limit the likelihood. Changes in enzyme activity, blood plasma chemistry, and increased PAH metabolites in bile have been observed in various species after exposure to dispersed oil, suggesting that exposure increases, but not necessarily that impacts at the individual level (i.e., reduced growth, reproduction, or survival) occurs (Lee and Anderson, 2005; Cohen et al., 2001; Ramachandran et al., 2004; Baklien et al., 1986).

Exposures to very diluted concentrations of the components of the dispersants may occur as a result of leaching to the water column from micelles over time (Fingas, 2008) or, to a limited extent, as a result of overspray during application (Butler et al., 1988; Scelfo and Tjeerdema, 1991). Although dispersants have inherently toxic characteristics, there is also evidence that dispersants may mitigate the acute (i.e., lethal) toxicity of oil alone to certain species (e.g., larval fish and invertebrates), or have little to no effect on species that pass through the upper 10 m of the ocean, but generally reside much deeper (e.g., cetaceans, fish, and sea turtles). By removing the surface oil slick, dispersants reduce the risk of direct contact with wildlife that dwell at or pass through the water surface to feed or breathe such as sea birds, sea turtles, and cetaceans.

Most aquatic organisms have the ability to metabolize and depurate petroleum hydrocarbons. Existing data demonstrate that complete depuration occurs once the source of the contamination is removed. It is unlikely that significant amounts of petroleum hydrocarbons will be accumulated by pelagic organisms during a dispersant application because of the short duration and low concentration expected in the water column. Under such conditions, any accumulated petroleum hydrocarbons should be rapidly depurated. Marine food chain biomagnification does not occur because vertebrate predators readily metabolize and depurate hydrocarbons from their tissues. Most marine organisms also metabolize and excrete the surfactants in dispersants. Metabolism of surfactants is rapid enough that there is little likelihood of food chain transfer from marine invertebrates and fish to predators, including the listed sea turtles and cetaceans (Neff, 1990).

Acute toxicity data for 48- and 96-hour exposures to Corexit® 9527 were compiled from 48 tests on 34 species within 31 different genera. Specifically, for invertebrates and aquatic plants, toxicity tests that lasted only 48 hours were included, because these species tend to have shorter periods of development than fish. Only 96-hour toxicity test data were included for fish species, with the exception of embryo-larval tests using Atlantic menhaden, red drum, and spot croaker (Fucik et al., 1995; Slade, 1982). Spiked tests had non-specific exposure durations, but they are expected to be ecologically relevant (Clark et al., 2001). Of the tests conducted, 2 used plants, 28 used invertebrates, and 18 used fish species. The observed LC50s for all species were between 2.4 and 840 ppm or mg dispersant/L water. Only bounded data were included in the calculation of HC5s; unbounded values (e.g., LC50 > 1,000 ppm) were omitted. Tests were carried out under various temperatures, each assumedly appropriate to the test species; therefore, not all tests are entirely applicable to waters in the U.S. Caribbean. Acute toxicity data for spiked and 48- to 96-hour exposures to Corexit® 9500 were compiled from 48 tests with 26 species and 24 genera. Of the tests conducted, 26 used invertebrates and 22 used fish. The observed range of 48- to 96-hour LC50s was between 3.5 and 1,038 ppm, the highest values being for spiked exposures.

Potential effects of dispersant use on listed species within the Caribbean region are described below.

1. Sea Turtles

At present, there are no known studies investigating the impacts of dispersants alone on marine reptiles, such as sea turtles. There is extensive research on the effects of oil alone, and at least one study investigating dispersed oil. Dispersants are not intended for direct application to sea turtles, so direct toxicity due to dispersants alone is unlikely. Nesting does occur in the U.S. Caribbean and adult and juvenile green and hawksbill sea turtles are known to frequent nearshore areas year-round, so ESA-listed marine reptiles in sensitive life stages could be exposed to dispersants (or dispersed oil) as a result of an oil spill response in Puerto Rico or USVI.

Another aspect of dispersion that is important to the assessment of sea turtles, is that dispersants are known to reduce the formation of buoyant tarballs (Shigenaka, 2003). It is speculated that the major route of oil exposure for adult sea turtles ingestion, particularly the ingestion of tarballs; this is based on the facts that oil has been found in turtle stomachs following field exposure, turtles apparently do not avoid oiled waters, and tarballs are known hazards for turtles (Shigenaka, 2003). It is therefore suggested that dispersant use would reduce the concentration of oil at the surface, and sea turtles' contact with it, or reduce the prevalence of tarballs that might be ingested incidentally by sea turtles. This conclusion was also reached by Shigenaka (2003), who noted that, prior to dispersant application, on-scene coordinators must take into account area contingencies (e.g., presence of eelgrass beds, depth of water column, presence of nesting habitat, etc.) in order to ensure the protectiveness of dispersion. It is not suggested that oil dispersion will entirely mitigate the mortality of sea turtles, since observations during the DWH event suggest the opposite (Barron, 2012).

It is also important to note that the only available study observing the impacts of dispersed oil on sea turtle embryos resulted in no adverse impacts (Van Meter et al., 2006); it was found that the percolation of oil through sediment in simulated nests resulted in a very low transfer of PAHs to the interior of the nest and eggs. It is still possible that the emergence of juveniles would result in exposure to those PAHs, but the bioavailability of PAHs in sediment would be significantly less than the bioavailability of dissolved PAHs initially in the water column (Albers and Loughlin, 2003). Exposure of adults to increased PAHs is not likely to result in acute toxicity, due to the rapid dilution and degradation of oil and its components after a dispersant application. Also, reptiles are able to efficiently metabolize and excrete ingested hydrocarbons (Albers and Loughlin, 2003), which should limit the bioaccumulation of PAHs after a dispersant application.

Juvenile sea turtles, which often are found with drifting sargassum mats in convergence areas further from shore, would particularly benefit from reduced surface oil exposure in the area under consideration. Exposure of sea turtles to tar balls, which they are known to ingest and which also adhere to juveniles, would be reduced because dispersants help prevent tarball formation. There are currently no specific data on the potential effects of dispersants and chemically dispersed oil on sea turtles. However, any effects from exposure to dispersants or chemically dispersed oil would be most likely be confined to the approximate footprint of the treated area and limited to a few hours post dispersant application due to dilution in the offshore water column.

Exposure of sea turtles to toxic volatile chemicals through inhalation of dispersed oil is expected to be less than through inhalation of oil alone (NRC, 2013), even though at least one component of dispersants is volatile (i.e., petroleum distillates, 2-butoxyethanol). This is achieved through the dispersion of volatile chemicals into the water column, another trade-off in toxicity between protecting species that breathe air (e.g., reptiles) and protecting those that do not surface to breathe (e.g., fish).

There could be indirect effects of dispersed oil on sea turtles through impacts to their prey. Loggerhead, hawksbill, and green sea turtles feed primarily on benthic prey that are unlikely to be adversely affected by dispersed oil. However, leatherback sea turtles feed on jellyfish. To date, there is little information on the toxicity of dispersants and chemically dispersed to jellyfish. A recent laboratory study found LC50 values as low as 0.15 ppm for a 3-day continuous exposure of larvae gelatinous zooplankton to physically dispersed oil (Almeda et al. 2013). The same study also found bioaccumulation of PAHs in 6-day continuous exposures. Unfortunately, effects concentrations and bioaccumulation factors were reported on a nominal, and not on a measured basis, limiting the applicability of this study to the assessment of the effects of chemically dispersed oil on jellyfish.

Within 10 m of the surface, potential exposure of water column organisms to concentrations of 10 ppm or higher dispersed oil would be brief and most of these organisms have the ability to rapidly metabolize petroleum hydrocarbons once exposure ceases. Therefore, population level effects from a short-term, pulsed exposure to low concentrations of dispersed oil are not expected so sea turtle species that prey on organisms such as jellyfish are not expected to be affected on a long-term basis from a temporary decline in prey items.

2. Marine Mammals

Detrimental effects of exposure of dispersants or chemically dispersed oil on the skin of cetaceans are not likely because the dermal shield is considered to be a highly effective barrier to the toxic compounds found in oil (short and any dispersants, which are water soluble, on the skin would be quickly washed off during the subsequent dive (Geraci and St. Aubin, 1990). It is important to note that particularly those marine mammals that develop subcutaneous blubber are not expected to be impacted by physical effects of oiling. Primary examples include cetaceans and pinnipeds, which regulate their body heat with blubber. According to modeling conducted by French-McCay (2004), the probability of surface oiling in the open ocean leading to death is 0.1% for cetaceans, 1% for pinnipeds, and 75% for furbearing marine mammals (e.g., sea otter).

Fouling of the baleen plates with oil while feeding at or near the surface of the ocean has been suggested to present a potential risk to the feeding capabilities of baleen whales. Laboratory studies have shown that oil fouling of feeding apparatus of baleen whales can temporarily restrict the flow of water (Engelhardt 1983; Geraci and St. Aubin 1985). However, studies have shown that 70% of the oil was removed within 30 minutes and >95% was removed by flowing water within 24 hours, leading the researchers to conclude that oiling of the baleen has a relatively short-term impact on baleen function, and would be reversible within a few days following exposure (Geraci and St. Aubin 1985). In a free-ranging whale, reduced feeding filtering and efficiency could impact the available energy storage to meet the requirements associated with migration and reproduction. However, such effects are likely to be short term (hours to days).

Data on the toxicity of dispersants to mammals are very limited. Inhalation of volatile compounds from a fresh oil slick at the surface may pose the greatest risk to cetaceans (Geraci and St. Aubin 1990; Schwacke 2013). The inhalation of fumes from dispersants could lead to various localized or systemic impacts including chemical pneumonia; inflammation of organ tissues (e.g., eyes and respiratory tract); increased difficulty breathing (not directly related to inflammation) (Roberts et al., 2011); injury to kidneys, liver, and blood cells (i.e., hemolysis); nausea; vomiting; narcosis; defatting and drying of skin; dermatitis (Nalco, 2005, 2010; CDC and ATSDR, 2010); and acute neurological impacts (e.g., altered neurotransmitter signaling) potentially leading to chronic depression, lack of motor coordination, and short-term memory loss (Sriram et al., 2011). Cetaceans are obligatory surface breathers and they may be exposed to volatile compounds concentrated above the water's surface. Exposure to these volatile compounds may result in mild irritation, neurological damage, and/or permanent damage to the respiratory surfaces, mucosal membranes, and liver. Furthermore, toxic compounds may be absorbed into the circulatory system. Use of dispersants on fresh, unweathered oil near the source early in the response can reduce the amount of volatile compounds in the vicinity of the slick (Curd 2011), thus reducing the risk of inhalation.

It is unclear how neurological impacts from inhalation of volatile compounds could affect ESA-listed mammals at the individual level, but behavioral impacts could assumedly result in a diminished ability to forage or avoid predation. It is not clear whether ecologically relevant concentrations of chemical dispersants will result in such impacts on marine mammals, particularly after dispersants mix into the water column. Direct application to mammals is not the intended or suggested use of chemical dispersants, and BMPs or response actions (e.g., avoidance of wildlife, monitoring for mammal presence, and hazing in an area to intentionally disperse wildlife) should mitigate animal exposures to concentrated dispersant chemicals.

Concerns have been expressed that listed marine species, namely baleen whales, could be adversely affected if major populations of key pelagic or benthic prey species were severely impacted. Though some studies do indicate toxic effects to zooplankton from dispersed oil, serious population impacts are unlikely at the short-term exposures that would result following dispersion in the zones pre-authorized under the LOAs.

Zooplankton, which are a particularly important food source for baleen whales, can become contaminated by assimilating hydrocarbons directly from seawater and by ingesting oil droplets and tainted food. Planktonic crustaceans can transform aromatic hydrocarbons to polar metabolites that may be excreted or bound to tissues. For a few days or weeks, unmetabolized or metabolized hydrocarbons in zooplankton could be transferred to predators. Geraci (1990) has estimated a forty-ton whale would have to consume approximately 150 gallons of oil to result in harmful effects. Considering the low concentrations and short duration of exposure to dispersed oil, as described earlier, it is unlikely the listed whales would ingest this volume of oil through consuming contaminated zooplankton.

Available toxicological data indicate the range of sublethal and lethal threshold concentrations for most aquatic organisms is above 10 ppm over an exposure period of 48 to 96 hours. It is unlikely that dispersed oil would exceed 10 ppm concentration and 2-4 hour duration at depths below the upper 10 m of the water column (SEA, 1995). Consequently, adverse effects are not expected below the upper 10 m of the water column following oil dispersion. Within 10 m of the surface, potential exposure of water column organisms to concentrations of 10 ppm or higher dispersed oil would be brief, lasting no longer than a few hours. Most of these organisms have the ability to rapidly metabolize and completely depurate petroleum hydrocarbons once exposure ceases. Although such exposures could result in temporary sublethal effects on physiological functions in some planktonic organisms, the existing data indicate that chronic effects are unlikely (NRC, 1989; SEA, Inc., 1995). The range of sublethal and lethal thresholds measured for modern dispersants in the absence of oil as determined by laboratory tests with sensitive species is much greater than concentrations that occur in the water column following dispersant application (NRC, 1989; Rycroft, et. al., 1994). Considering the broad distribution and relatively short life cycle of zooplankton, population level effects from such a short-term, pulsed exposure to low concentrations of dispersed oil are not expected and, therefore, unlikely to adversely impact predators such as baleen whales. Similarly, for beaked whales it is expected that any declines in the populations of prey items related to the use of dispersants will be short-term and will not have a measurable impact on the population of these species.

3. Fish

Abnormal development and narcosis are the most often cited modes of toxicity (NRC, 2005). At very low doses, dispersants have been shown to be embryotoxic to fish exposed at early life stages (Lonning and Falk-Petersen, 1978; Falk-Petersen et al., 1983). This is relevant to fish species that spawn in nearshore waters where larvae drift into shallow nursery habitats. While the direct application of dispersants is not intended for nearshore waters, dispersion in open water that, over time, results in diluted dispersant concentrations in nearshore waters could have a marked impact on shallow water reef species such as Nassau grouper. However, given the toxicity of oil alone and the potential impacts caused by oiling of nearshore areas and intertidal shorelines, it may still be beneficial (relative to baseline oiling) to apply dispersants, if done at a distance from known spawning habitat.

Marine finfish, for example, take up petroleum hydrocarbons from water and food. The compounds induce the hepatic Mixed-Function-Oxidase (MFO) system and within a few days following exposure, aromatic hydrocarbons are oxygenated to polar metabolites and excreted. For this reason, most fish do not accumulate and retain high concentrations of petroleum hydrocarbons and so are unlikely to transfer them to predators, such as the listed sea turtles and cetaceans. The fish may be tainted with metabolites bound to tissue macromolecules, but these metabolites are so reactive that it is unlikely that they would be released in a toxic form during digestion by the consumer and so would not pose a serious risk (Neff, 1990).

As noted previously, when dispersants are applied in deep water to turbulent seas, the resulting oil concentrations in the water column will remain below levels observed to cause adverse biological effects to zooplankton in laboratory tests. The range of sublethal and lethal thresholds measured for modern dispersants in the absence of oil as determined by laboratory tests with sensitive species is much greater than concentrations that occur in the water column following dispersant application (NRC, 1989; Rycroft, et. al., 1994). Considering the broad distribution and relatively short life cycle of zooplankton, population level effects from such a short-term, pulsed exposure to low concentrations of dispersed oil are not expected and, therefore, unlikely to adversely impact predators such as scalloped hammerhead sharks.

Acute toxicity data for 48- and 96-hour exposures to Corexit® 9527 were compiled from 48 tests on 34 species within 31 different genera (Fucik et al., 1995; Slade, 1982; Clark et al., 2001). The majority of fish were less sensitive than invertebrates, and as sensitive as plant species. The range of LC50s for rainbow trout was between 96 and 260 ppm Corexit® 9527 (Doe and Wells, 1978; Wells and Doe, 1976). Acute toxicity data for spiked and 48- to 96-hour exposures to Corexit® 9500 were compiled from 48 tests with 26 species and 24 genera. Fish were generally less sensitive to Corexit® 9500 than to Corexit® 9527. Of the fish tested, rainbow trout and red drum were the least sensitive; rainbow trout had a 96-hour LC50 of 354 ppm, and red drum had a spiked LC50 of 744 ppm. Other relatively insensitive species included the sheepshead minnow (*Cyprinodon variegatus*) and gulf killifish (*Fundulus grandis*). In addition some tests, but not all, indicated inland silverside to be relatively insensitive.

4. Corals

Abnormal development and narcosis are the most often cited modes of toxicity (NRC, 2005), although numerous sublethal impacts on invertebrates may also occur. Dispersants have been shown to be toxic to invertebrates at early life stages at very low doses (Lonning and Falk-Petersen, 1978; Falk-Petersen et al., 1983), but dispersants have also been shown to be less toxic than oil alone (Fingas, 2008; NRC, 2005). Therefore, dispersants alone do not pose a greater threat than that of the baseline condition for a spill cleanup.

Invertebrates are known to bioaccumulate hydrocarbons and PAHs (Boehm et al., 2004; Meador, 2003), which can lead to narcosis (Logan, 2007). Early-life-stage exposures to oil (including PAHs) can lead to developmental impacts, reduced growth, and death (Lee, 2013; Lonning and Falk-Petersen, 1978; Falk-Petersen et al., 1983; Albers and Loughlin, 2003). Exposure to oil can also lead to localized lesions on organ tissues (Brown, 1992), although it is unclear whether lesions have an impact at the population level that can, in turn, indirectly impact ESA-listed species by significantly reducing their prey base. Various other effects have been noted, including reduced respiration and movement (related to physical smothering), cytotoxicity and cytogenotoxicity, and altered feeding and excretion (Suchanek, 1993). These sublethal impacts may lead to mortality, but it is unclear whether, in an oil dispersion situation, PAH concentrations would be high enough, or exposures to PAHs sufficiently long, to cause such impacts on a broad scale (i.e., in a large enough area to reduce the prey base of ESA-listed species or the population of ESA-listed corals themselves).

It has been commonly noted that dispersants are less toxic than oil alone, but that dispersed oil is more toxic than oil alone (Fingas, 2008; NRC, 2005); therefore, the addition of dispersants is typically considered a direct threat to pelagic invertebrates and fish, and an indirect threat to mammals, birds, and reptiles. An example of such impacts on a planktonic community is presented by Jung et al. (2012), who observed greater impacts in a mesocosm study after dispersants had been applied to oil (relative to oil alone). Similarly, Scholten and Kuiper (1987) observed impacts on planktonic communities relating to the bioavailable fraction of oil; they warned against the use of dispersants, which enhance the dissolved (and therefore bioavailable) fraction of hydrocarbons in the water column. Many invertebrates, particularly during larval life stages, are found in shallow water, where they are exposed to high concentrations of oil and dispersed oil during a spill event. Acute mortality in the vicinity of the dispersed spill may occur in many sensitive species (French-McCay, 2010; Scholten and Kuiper, 1987; Stige et al., 2011), but widespread mortality will result from the uncontrolled spread of an oil spill (i.e., associated with baseline condition) (Abbriano et al., 2011). Reducing the overall area of a spill by rapidly dispersing oil into the water column may result in diminished impacts over a broader area; quickly removing oil from the ocean surface may result in negligible impacts to the planktonic community, though benthic species such as corals could be impacted instead (Varela et al., 2006).

It is possible that the addition of oil and dispersant to a natural system may cause a planktonic or benthic community to become dominated by species that are already present (i.e., to tolerant species) (Ortmann et al., 2012; Atlas and Hazen, 2011; Parsons et al., 1984) and while such a shift may not result in an overall reduction in biomass (Varela et al., 2006) or a sustained impact (Abbriano et al., 2011), even in low-productivity environments (Cross and Martin, 1987), it could result in a significant shift in the coral community. Sublethal responses (e.g., reduced immune responses) measured in invertebrate communities resulting from chronic exposures to oil (and PAHs in particular) are often temporary within a population, such that a community may return to pre-spill conditions within a matter of months or years (Edwards and White, 1999; Dyrynda et al., 2000).

Given the right oceanographic conditions, dispersant use may be appropriate near submerged coral reefs (i.e., where water depth increases quickly, and subsurface currents are likely to carry dispersed oil away from the reef). NOAA has modeled such situations for potential spills, when dispersants were considered as a response tool near a coral reef. Usually, these determinations are made after considering the potential shoreline impacts and evaluating local weather and oceanographic conditions at the time of the spill.

Acute toxicity data for 48- and 96-hour exposures to Corexit® 9527 were compiled from 48 tests on 34 species within 31 different genera (Fucik et al., 1995; Slade, 1982; Clark et al., 2001). Invertebrate species had more varied LC50s than did fish or plants, likely due to the greater number of tests and test conditions conducted for invertebrates. Green hydra (*Hydra viridissima*) and grass shrimp (*Palaemonetes pugio*) were the least sensitive invertebrate species and least sensitive species, overall. Various crustaceans (*Allorchestes compressa, Pseudocalanus minutes, Penaeus setiferus*) and Pacific oyster (*Crassostrea gigas*) were the most sensitive invertebrates and most sensitive species, overall. Acute toxicity data for spiked and 48- to 96-hour exposures to Corexit ® 9500 were compiled from 48 tests with 26 species and 24 genera. Of the tests conducted, 26 used invertebrates and 22 used fish. Invertebrates that were less sensitive to Corexit® 9527 included the green hydra and Eastern oyster (*Crassostrea virginica*). Sensitive species included the amphipod (*A. compressa*), copepods (*Eurytemora affinis* and *Tigriopus japonicus*), and red abalone (*Haliotis rufescens*).

Field studies from Bermuda, the Arabian Gulf, and Panama on impacts to coral from chemically dispersed oil indicate that coral follows the pattern of most other organisms: its sensitivity to dispersed oil depends on the dose (the concentration and the length of exposure). Very high dispersed oil concentrations and long exposures can kill coral, whereas lower doses and short-term exposure show few, if any, impacts, many of which are reversible. Thus, dispersants and dispersed oil may be less damaging in real field conditions than had been suggested by older laboratory toxicity tests. These results highlight the importance of evaluating toxicity in the context of realistic exposures, which can be estimated from oceanographic models.

A qualitatively different kind of synergistic spill effect consideration that should be mentioned is the potential synergistic increase in impact with the combination of oil and dispersants. Cook and Knap (1983) found that, by themselves, exposure to crude oil and a dispersant had no effect on coral photosynthesis. However, combining the two significantly depressed total carbon fixation, and significantly altered into which chemical fraction the carbon was fixed. Although this was a transient impact, it was one that occurred only when the two materials (oil and dispersant) were combined. Other studies that comparatively examined adverse effects to coral from oil alone and oil + chemical dispersant found that combining the compounds increased toxicity. This, of course, carries significant implications for spill response.

5. Habitat Effects

a. TROPICS Study

The TROPICS (Tropical Investigations in Coastal Systems) field study began in 1983-84 near Bocas del Toro, Panama. The study was designed to examine the relative short and long-term effects of dispersed crude oil versus non-dispersed crude oil on nearshore tropical marine ecosystems. After baseline studies (1983), two 900 m^2 sites composed of intertidal mangrove and subtidal seagrass-coral zones were dosed (1984) with untreated Prudhoe Bay crude oil and Prudhoe Bay crude oil dispersed with Corexit® 9527. At periodic intervals over 25 years, the sites were monitored and effects were compared to a nearby reference site. A number of papers were published during the study period. It should be emphasized that while the TROPICS study provided good data, the exposure conditions are not realistic, as dispersants would not be applied in such shallow waters or intertidal environments as those in the study.

In the short term, mortality to invertebrate fauna, seagrass, and corals was observed at both the dispersed oil and non-dispersed oil sites. In the long-term (10-25 years), as compared to the reference site, there was little to no oil detected and the ecosystem appeared to have returned to pre-dosing condition at the dispersed oil site. At the non-dispersed site, substrate core samples revealed the continued presence of oil. The effects of non-dispersed and dispersed crude oil at the TROPICS site have been documented and determined to be short- and long-term for various parameters. While dispersed crude oil showed serious and immediate effects on fauna, these effects were short-lived. In contrast, the non-dispersed crude oil site has undergone considerable long-term changes.

b. Coral

Coral cover increased at all sites, culminating at approximately 200% over baseline at non-dispersed and dispersed sites, and almost 400% at the reference site. However, through 2001-2002, replacement by flora (seagrass and algae) was apparently a cause at the dispersed and reference sites, but not at the non-dispersed site. By contrast, percent flora in coral habitat increased at the non-dispersed and dispersed sites, while dropping to baseline at the reference site (DeMicco et al, 2011).

The dispersant treatment area for corals immediately following treatment found that coral coverage was greatly reduced, and that growth in one out of four species tested following dispersant use was reduced. However, the recovery and coverage to pretreatment levels happened after 10 years. With oil, there were no effects to coral because the oil traveled on the surface of the water (Dodge et al, 2014).

c. Seagrass

Seagrass declined overall at both sites, ending at 58% (non-dispersed) and 88% (dispersed) of original coverage. With the exception of Ward (2003), other seagrass metrics besides density were not pursued after 1994 due to time and funding constraints. Reviewing the TROPICS field notebooks, there are various measures of area and depths which indicate that the subtidal shelf off the non-dispersed site became narrower and deeper over time. For example, the average width of the non-dispersed site was 7.8m in 1984 while the dispersed site was 6.9m (Baca unpublished). In contrast, measures recorded by Ward in 2001 (unpublished) showed widths of 1.5m and 6.5m for the non-dispersed and dispersed sites, respectively. This would indicate a loss of 6.3m at the non-dispersed site, a probable measure of impacts (DeMicco et al, 2011).

Thorhaug et al (1986) studied a range of concentrations of dispersed oil on several species of seagrasses dominant throughout the Atlantic subtropical Greater Caribbean basin. Results showed *Halodule wrightii* and *Syringodium filiforme* had an LD50 at 75 ml dispersed oil in 100 liters seawater for 100 hours exposure, whereas *Thalassia* was more tolerant with an LD50 at 125 ml in 100 liters seawater for 100 hours. Dispersant alone had a significant effect on *Halodule* and *Syringodium*, but not on *Thalassia*. Louisiana crude oil had a slightly lesser effect than Murban. However, the difference between seagrass species was greater than between oils.

Acute toxicity data for 48- and 96-hour exposures to Corexit® 9527 were compiled from 48 tests on 34 species within 31 different genera (Fucik et al., 1995; Slade, 1982; Clark et al., 2001). Two aquatic plant species were tested: a brown alga (*Phyllospora comosa)* and turtle grass (*Thalassia testudinum*). The 48-hour LC50 for the brown alga was 30 ppm (Burridge and Shir, 1995), and the 96-hour LC50 for turtle grass was 200 ppm (Baca and Getter, 1984).

d. Mangroves

Oil was still present in sediments more than 10 years after both the dispersed and non-dispersed oil areas. In the short term, the oil showed very high mortality to mangroves, but sublethal effects only from dispersed oil. In the long term, even at 10 years, there was still very high mortality, but at the dispersed oil site, after 10 years, growth was similar to the reference site, with some reduction in leafing (Dodge et al, 2014).

Mangrove, primarily *Rhizophora mangle*, repopulation was impeded and substrate erosion was observed at the non-dispersed site. The mangrove forest lost a significant portion of trees which were slowly replaced by "waves" of seedlings that never reached maturity, presumably due to the long-term exposure to toxic petroleum compounds that remained trapped in the substrate.

Dispersants appeared to prevent long-term contamination to mangrove forests and the resulting absence of oil in the substrate, made possible by advantageous properties of dispersed oil, avoided long-term exposure to toxic compounds and provided the conditions for ecosystem and habitat recovery. Most significantly, after 25 years, components of non-dispersed crude oil, specifically aromatic hydrocarbons, remain in the mangrove substrate in the non-dispersed site, where chronic exposure continues to inhibit recovery and repopulation. The long-term (post-ten-year) effects on mangrove fauna at the sites have not been extensively studied (DeMicco et al, 2011).

C. Potential Impacts of In-Situ Burning

The primary objectives of a spill response are to remove as much oil as possible from the surface of the water as quickly as possible and to prevent oil from moving into nearshore and shoreline areas where removal is more difficult and environmental impacts most severe. In-situ burning, under appropriate conditions, may offer the best response option to help achieve these objectives by rapidly and efficiently removing large volumes of oil from the water surface. The benefits to listed and other species include reduced risk of oil exposure in the aquatic environment and of contamination of critical intertidal areas.

In-situ burning, however, may pose some risks to the listed species. Because both cetaceans and sea turtles must surface to breathe, there is conceivably potential risk of injury from surfacing in the area of the burn. In order to maintain control of the burn, though, the area in which it is actually conducted is kept relatively small. Furthermore, an in-situ burn is of relatively short duration, typically only a few hours, due to the efficiency of the technique. The overall impacts of combustion products, thermal effects, and floating burn residue are minimal in light of their short-term, localized influences and the ease with which such influences can be controlled. The location and timing of the in-situ burning, for example, can be controlled in order to minimize any exposure to wildlife, particularly listed species. Given the alternative of much greater volumes of unburned oil impacting nearby resources, any impacts resulting from the burn would be expected to be much less severe than those manifested through exposure to a large, uncontained spill.

There is no reason to suspect that in situ burning will add to the cumulative environmental stresses currently acting on the listed species. The effect of in-situ burning is to speed up and increase the efficiency of removal of spilled oil from the environment, and thus, to reduce the net environmental impact, including impacts to listed species.

1. Inhalation

The approximate levels of concern for mammals and sea turtles to smoke and gases from in-situ burning have not been determined. One perspective is that "lungs are lungs" and the effects should be similar for air-breathing vertebrates (NOAA, 2010). Also, little is known about what would be the smoke exposures levels and durations for animals that: 1) come to the water surface to breathe (such as whales); 2) fly through the smoke plume or spend a lot of time on the water surface (such as ducks); or 3) live on land in areas downwind of a burn, who might (or might not) be able to move out of the plumes (such as marsh birds, mammals, and reptiles).

Effects from exposures to in situ burn gas by-products are likely to be similar to those experienced by humans, potentially causing irritation of the eyes, nose, throat, as well as potentially life-threatening impacts to an individual due to impacts to air exchange from effects deep in the lung (pulmonary edema) or death due to oxygen displacement. Guidelines for use of in-situ burning on water exclude its use in areas proximal to bird flocks, sea turtles, marine mammals, etc., and often require observers to assure that these guidelines are being followed, which should reduce the risk of effects from inhalation.

2. Floating/Stranded Burn Residue Contact Hazards

The burning process of fresh oil generally consumes the majority of the oil, upwards of 90%, but does leave unburned, fairly viscous and dense residues behind that have a tendency to form tarballs and sink. Tarballs and residues from in-situ burning are not subjected to the normal weathering process. Asphaltenes appear to be much higher in tarballs that are in-situ burn residues (Meyer et al. 2013). Pyrogenic PAH concentrations increased in post-burn samples, although in-situ burning, even of weathered oil from the DWH oil spill, was found to result in a decrease of total PAHs (Meyer et al. 2013).

Sticky burn residues that float can foul the feathers of birds and the fur of marine and terrestrial mammals (e.g., seals, otters). Seabirds are particularly at risk because even small amounts of oil interfere with a bird's ability to thermo-regulate and maintain buoyancy. In small-scale tests with seven crude oils and one diesel, Buist and Trudel (1995) reported that burn residues from the heavier oils formed brittle, non-sticky residues that would not stick to feathers and fur; conversely, the lighter crude oils and diesel did produce sticky burn residues.

Water temperatures under a burning slick can reach near boiling in the top few millimeters in a static situation. However, water only a few centimeters below the slick remains unaffected. If a fire boom containing a burning oil slick is being towed, or if underlying water is moving, then there is no appreciable rise in water temperature. As described above, thermal effects on the water underlying the burn are negligible, and so pose little risk to listed species.

The Response and Chemical Assessment Team at Louisiana State University analyzed five tarball samples on July 06, 2011 collected from a commercial fisherman's trolling net in the Gulf of Mexico. Gas chromatography/mass spectrometry fingerprinting was performed for each sample to provide oil characteristic and quantitative information. Additionally, the oil fingerprints from these five tarball samples were compared to oil fingerprints from tarballs collected by the *F/V Our Mother* (8 samples) in January 2011 and the *F/V Aubreigh Marie* (12 samples) in March 2011. All the tarball samples were compared to oil fingerprints of the MC 252 source oil from the DWH oil spill. It was determined that all of the tarball samples came from the same crude oil, and that oil was a match to the MC 252 crude from the DWH oil spill in 2010. Based on the hydrocarbon profile, there is a very high probability that all tarball samples were residue from the in-situ burning of the MC 252 oil during the active portion of the spill response. The data from Meyer et al. (2013) demonstrate that residues from in-situ burning are not subjected to the normal weathering processes due to the fact that dense globs of oily residue are often encapsulated within a hardened exterior.

3. Burn Residue Properties, Toxicity, and Sinking Hazards

Though most burn residues float and are collected, negatively buoyant residues and those that escape collection could pose some risk of exposure to sea turtles and cetaceans through ingestion or fouling of baleen. The tendency of burn residues to sink renders them difficult to remove from the marine environment and difficult to sample in order to determine physical and chemical changes (Shigenaka et al. 2015). The effects of ingestion of these residues are not completely known. Even if they do cause some toxic effects, exposure is likely to be low considering the small volume of residues produced. Typically, only a small percentage of the original oil volume remains as residue following an in-situ burn. Any unrecovered residue would certainly pose lower exposure risk than the volume of originally released product.

Blenkinsopp et al. (1997) evaluated the aquatic toxicity of samples from the NOBE burn. The burn residue was stirred in water for 48 hours to generate a water-accommodated fraction (WAF) for different oil loadings. The WAF was then used in toxicity tests for three-spine stickleback, rainbow trout, and gametes of the white sea urchin. The samples were found to be nontoxic to the species tested. The authors concluded that in situ burn does not generate a burn residue that is more toxic that the weathered oil.

In Australia, bioassays using crude oil burn residues created in the laboratory showed no acute toxicity to amphipods and very low sub-lethal toxicity (burying behavior) to marine snails (Gulec and Holdway, 1999). Based on these limited tests and the chemical composition of the burn residues, they are expected to yield little or no chemical toxicity.

Measurements at field and laboratory test burns under open water conditions show that burning does not accelerate the release of oil components or combustion by-products to the water column and does not increase in acute toxicity using standard bioassays (Daykin et al., 1994; Ferek et al., 1997). In these cases, water quality and toxicity after a burn are expected to be the same or less as before the burn.

4. Habitat Effects of In-Situ Burning

A potential disadvantage of open-water in situ burning is that a small percentage of the original oil volume may remain as a taffy-like residue after the burn. Floating residue can be collected but residues that sink or escape collection and move inshore could potentially contaminate mangroves or other habitats. It is important to note that, in contrast to open-water burning, in situ burning should not be conducted within mangrove forests. Under no circumstances should live mangrove vegetation be cut or burned. Mangrove trees are slow-growing and take decades to reach a mature stage. The loss of a large number of trees may compromise the forest structure, making it unlikely to recover naturally.

D. Physical Impacts of Response Operations

1. Vessel Operations

Vessel traffic, anchor deployment, dragging lines, and physical contact by response workers can detrimentally impact coral reefs and associated habitats. Responders working near coral reefs should take care to avoid physical damage to coral, especially in shallow waters. Using floating lines, especially for salvage, but also for boom deployment and other operations, prevents damage to coral structures from dragging heavy lines over the reef itself. Response methods such as skimming and placement of certain types of booms will need to be limited to deeper waters (greater than 3 m) to avoid direct physical impacts to coral. Anchor locations for booms also need to be carefully selected to avoid impacts to corals from the anchors themselves and from lines. In areas where coral is exposed at low tides, workers should avoid walking on reefs.

Collision with vessels poses a serious threat to some endangered species. Right whales are particularly susceptible to injury or death from ship collisions because they surface skim-feed and often rest at the surface. Response vessel speeds should be restricted any time endangered species are in the area of an oil spill, especially when visibility is limited.

Adverse impacts on EFH from any accidental oil spills from response vessels are a potential source of contamination to the marine environment. Any spills would be expected to be of a small size, although there is a potential for large spills to occur. Many factors determine the degree of damage from a spill, including the type of oil, size and duration of the spill, geographic location of the spill, and the season. Although oil is toxic to all marine organisms at high concentrations, certain species are more sensitive than others. In general, the early life stages (eggs and larvae) are most sensitive, juveniles are less sensitive, and adults are least so (Rice et al. 2000), thereby causing potential mortality to EFH species and impacting water quality of the EFH.

2. Physical Removal

As previously indicated, physical removal of oil is normally the preferred spill response option. Mechanical/manual removal of oil will remain the predominant response tool due to the nature and size of most spills, which usually are close to shore and in areas where dispersant usage would not be applicable, and in-situ burning would not be appropriate due to human health concerns, environmental, economic and logistical considerations.

Experience has shown, however, that physical response often cannot adequately deal with very large spills offshore. Performance of physical methods can be severely limited by weather and oceanic conditions and by the nature of the oil slick. Booms and skimmers are of limited use even in moderate seas and are usually effective only at slow currents (less than 1 knot) and low wave heights (less than 2 m). Consequently, physical recovery rates are often poor. Even under calm conditions, use of physical equipment alone to deal with large spills in which oil rapidly spreads over large areas may not be feasible.

Physical containment and collection of spilled oil on water using booms and skimmers is the primary cleanup method used at most spills. High current speeds, heavy wave action, or shallow water may limit the effectiveness of either booms or skimmers and necessitate consideration of alternative cleanup strategies. Skimming operations will most likely be conducted outside the reef in deeper waters to prevent oil from coming over the reef crest. It may be difficult to anchor collection booms on the reef slope because of great depth.

Booms should be anchored in a way that does not damage coral, and tended regularly to maintain effective positioning and avoid damaging shallow coral.

The use of dispersants and in-situ burning will be considered when and where physical removal is impossible or insufficient for protecting natural resources, including listed species and designated critical habitat and EFH. As discussed above, research, field studies and actual response experience indicate the use of dispersants and in-situ burning under appropriate conditions are more beneficial than not for the listed species, and the environment overall.

IV. AVOIDANCE AND MINIMIZATION PROCEDURES

The CRRT is committed to implementing measures to reduce and avoid potential impacts on federally listed species, designated critical habitat and EFH. The CRRT has compiled the attached collection of Best Management Practices (BMPs) to avoid and minimize impacts to fish and wildlife resources during a response to an oil spill.

This compilation of BMPs includes measures applicable to the protection of corals, sea turtles and marine mammals during dispersant and in-situ burning operations. Guidance to response personnel is offered for skimming and in-situ burn operations, and protocols for field observers are also provided.

Measures to reduce or eliminate potential adverse impact on managed fisheries and EFH include vessel speed restrictions, use of experienced response personnel, use of dynamic positioning rather than anchoring, and advising staff on the importance of avoiding contact with coral during implementation of dispersant and in-situ burning response operations.

Due to area, environmental, and situational differences amongst potential operation areas, the CRRT will maintain this list of BMPs as a separate appendix in the Regional Contingency Plan, and will update and revise the list as additional BMPs are developed, and/or updated information becomes available.

V. EFFECTS DETERMINATIONS

A. Effects of Dispersants/Dispersed Oil

1. Sea Turtles and Critical Habitat

Green, loggerhead, leatherback, and hawksbill sea turtles:

As discussed previously, there are no known studies on the effects of dispersants alone on sea turtles, although there is one study on the effects of oil and dispersed oil. Dispersants could affect early life stages of sea turtles if transport of dispersants and dispersed oil to nesting beaches occurs. Dispersants are not intended for application in shallow nearshore waters to the anticipated impacts of dispersants on eggs and hatchlings are expected to be minimal. In addition, the one study on impacts of dispersed oil on sea turtle embryos found no adverse impacts (Van Meter et al. 2006). It is possible that the emergence of juveniles would result in exposure to those PAHs, but the bioavailability of PAHs in sediment would be significantly less than the bioavailability of dissolved PAHs initially in the water column (Albers and Loughlin, 2003). Exposure of adults to increased PAHs is not likely to result in acute toxicity, due to the rapid dilution and degradation of oil and its components after a dispersant application (Section 2). Also, reptiles are able to efficiently metabolize and excrete ingested hydrocarbons (Albers and Loughlin, 2003), which should limit the bioaccumulation of PAHs after a dispersant application.

Juvenile and adult sea turtles may consume prey items that occur in shallow water during a chemical dispersant application. Studies have shown that seagrass does not appear to be impacted by the application of dispersants but organisms in seagrass beds, that may include sponges that are consumed by hawksbill sea turtles, for example, do suffer mortality as a result of dispersant application. Because existing data demonstrate that complete depuration occurs once the source of the contamination is removed, we do not expect prey or foraging habitat contamination to result in significant impacts to sea turtles.

Sea turtles surface to breathe, which requires that they potentially come into contact with oil at the ocean's surface. Because dispersants remove oil from the ocean's surface and, through dilution, reduce the concentration of oil, it can be expected that the exposure of sea turtles to oil will be mitigated by dispersants.

For leatherback sea turtles, which are only found nearshore during nesting season, which is from February to August in the U.S. Caribbean, the contamination of prey items is expected to have little effect. Similarly, loggerhead sea turtles are uncommon in the U.S. Caribbean with extremely infrequent reports of nesting and few reported in stranding data. Therefore, we do not expect dispersant application to have significant effects to the Northwest Atlantic Distinct Population Segment.

For these reasons, we believe the application of dispersants may affect, but are not likely to adversely affect sea turtles especially if BMPs detailed in the previous section are employed during the response action.

Green sea turtle critical habitat:

Designated critical habitat for green sea turtles is composed of seagrass beds, embayments, and other habitats used by juvenile and adult green sea turtles around Culebra Island, Puerto Rico, and its surrounding islands and cays for refuge and foraging. Based on the TROPICS study, impacts to seagrass from dispersed oil in both the short- and long-term were not observed (Baca and Getter 1984; Baca et al. 2011). In addition, because dispersant application is not intended for nearshore waters where seagrass beds are concentrated, we do not anticipate direct impacts to shallow water habitats. Similarly, although coral cover and growth of some coral species was reduced due to the presence of dispersed oil in the TROPICS experiment, full recovery of the impacted area occurred after 10 years (Dodge et al. 2014). Therefore, we believe the impacts from the use of dispersants on green sea turtle critical habitat around Culebra will be minimal.

Leatherback sea turtle critical habitat:

This critical habitat designation is largely to protect nesting and hatchling sea turtles around Sandy Point, St. Croix. The final rule does not specify particular habitat characteristics. Because dispersant applications are not intended for nearshore waters or for areas with special protections, such as Wildlife Refuges, which is the case for Sandy Point, we do not anticipate dispersant application within leatherback critical habitat. The use of dispersants offshore could occur in the event of a spill and wind and currents could lead to the transport of dispersed oil into leatherback sea turtle critical habitat. Therefore, we believe the impacts from the use of dispersants on leatherback sea turtle critical habitat around Sandy Point will be minimal.

Hawksbill sea turtle critical habitat:

Designated critical habitat for hawksbill sea turtles is composed of reefs and colonized hard bottoms around Mona and Monito Islands, Puerto Rico, utilized as refuge and foraging habitat by adult and juvenile hawksbills. The TROPICS study found that coral habitat where dispersed oil was directly applied showed immediate declines in coral cover and species composition but demonstrated full recovery 10 years after the application (Dodge et al. 2014). Overall, there were greater increases in coral cover at all treatment sites (reference, oil only, and dispersed oil), although these increases were greatest at the reference site (DeMicco et al. 2011). The application of dispersants is not intended for nearshore waters, which would exclude application in much of the designated critical habitat around Mona and Monito Islands. Dispersed oil could be carried into the nearshore environment by wind and current transport, but the impacts to habitat are expected to be minimal.

2. Marine Mammals

Sperm, blue, fin, sei, and humpback whales:

ESA-listed whales are unlikely to be impacted by the physical effects of dispersants. These species are very large and will not likely be exposed to concentrations of dispersants or dispersed oil in quantities large enough to cause acutely toxic effects (e.g., mortality). Dispersed oil rapidly dilutes and degrades over time, so chronic exposure to dispersants or dispersed oil in the water column is unlikely, especially given that ESA-listed whale species are present in U.S. Caribbean waters mainly during their winter migration through the area. Accumulation of PAHs in tissue over time as a result of chemical dispersion is unlikely due to the ability of mammals to metabolize and excrete PAHs, as well as the expected short-term nature of a PAH exposure after a chemical dispersion event.

The prey base of humpback, sei, fin, and blue whales, which are baleen whales, are zooplankton and small fish and cephalopods. It is possible (although unlikely) that dispersed oil will be ingested by humpback, sei, fin, and blue whales, which feed through their baleen on planktonic species. However, the ingestion of even large quantities of crude oil by much smaller species has been found to cause minimal effects, and cetaceans are likely able to efficiently metabolize hydrocarbons (Albers and Loughlin, 2003). Ingestion of dispersed oil during feeding may increase, leading to fouled baleen and sublethal impacts (e.g., vomiting and tissue irritation). Such effects may reduce the feeding efficiency of baleen whales (BOEMRE, 2011).

It is highly unlikely that fin whales will ingest large quantities of dispersed oil due to the depths at which they are often found (i.e., between 50 and 600 m) (US Navy, 2011; Croll et al., 2001; Goldbogen et al., 2006; Panigada et al., 2003). Assuming that fin whale feed at depths > 10 m, it is likely that their prey are also found primarily at depths > 10 m; therefore, the prey population of fin whale is unlikely to be exposed to high concentrations of dispersed oil with the exception of the larval life stages of some prey species that may be found in shallower waters. Similarly, it is unlikely that humpback whale will ingest large quantities of dispersed oil due to the depths at which they feed, typically between 92 and 120 m deep (NMFS, 2011a), and as deep as 500 m (US Navy, 2011).

Humpback whales can also be found in the nearshore environment, in particular when calving during their winter migration through the Caribbean. Dispersant applications are not intended for nearshore habitats, although tides and currents may move a dispersed spill into the nearshore environment. If an oil spill has been appropriately dispersed (i.e., all BMPs have been observed by the Federal On-Scene Coordinator and dispersion has been effective), dilution and biodegradation are likely to occur to some extent prior to a plume reaching the nearshore environment.

Sperm whales prey on large squid, skates, sharks, and fish. Sperm whales may also ingest dispersed oil while feeding. Ingestion of dispersed oil during feeding may increase, leading to sublethal impacts (e.g., vomiting and tissue irritation). However, as for baleen whales, we anticipate that this species is also able to efficiently metabolize hydrocarbons.

Given that embryonic and larval life stages of whale’s prey may be found in shallow water during a chemical dispersant application, it is possible that these prey species may be impacted. As discussed previously, most aquatic organisms have the ability to metabolize and depurate petroleum hydrocarbons. Existing data demonstrate that complete depuration occurs once the source of the contamination is removed. Therefore, we do not expect prey contamination to result in significant impacts to whales. The trophic transfer of PAHs to invertebrates in dispersed-oil exposures does occur, but fish metabolize PAHs fairly efficiently (Wolfe et al., 2001; Logan, 2007) and we can assume that this is also true of cetaceans. Therefore, magnification of PAHs in whales through their diet is unlikely (Albers and Loughlin, 2003).

Whales periodically surface to breathe, which requires that they potentially come into contact with oil at the ocean's surface. Because dispersants remove oil from the ocean's surface and, through dilution, reduce the concentration of oil, it can be expected that the exposure of blue whale to oil will be mitigated by dispersants. Exposure will increase as the species moves from deep waters through the upper 10 m (before reaching the surface), but this is expected to result in minimal impacts. Calves tend to reside in the upper 20 m of the water column (Koski and Miller, 2009), which puts them at particular risk of exposure to both dispersed oil and oil alone. The acute impacts of dispersed oil on cetaceans are less than those of oil alone, due to the altered route of exposure (i.e., ingestion of dispersed oil as opposed to inhalation or aspiration of surface oil).

For these reasons, we believe the application of dispersants may affect, but is not likely to adversely affect whales especially if BMPs detailed in the previous section are employed during the response action.

3. Fish

Scalloped hammerhead shark and Nassau grouper:

Abnormal development and narcosis are the most often cited modes of toxicity for fish (NRC, 2005). At very low doses, dispersants have been shown to be embryotoxic to fish exposed at early life stages (Lonning and Falk-Petersen, 1978; Falk-Petersen et al., 1983), in particular species such as Nassau grouper where larvae drift into shallow nursery habitats. While the direct application of dispersants is not intended for nearshore waters, dispersion in open water that, over time, results in diluted dispersant concentrations in nearshore waters could have a marked impact on shallow water reef species such as Nassau grouper.

The embryonic and larval life stages of prey species for juvenile and adult Nassau grouper and scalloped hammerhead sharks may be found in shallow water during a chemical dispersant application. It is possible that these prey species may be impacted by the dispersant application. Because most aquatic organisms have the ability to metabolize and depurate petroleum hydrocarbon rapidly, we do not expect prey contamination to result in significant impacts to Nassau grouper or scalloped hammerhead sharks. The trophic transfer of PAHs to invertebrates in dispersed-oil exposures does occur, but fish metabolize PAHs fairly efficiently (Wolfe et al., 2001; Logan, 2007). Therefore, magnification of PAHs in these species through their diet is also unlikely.

Most fish do not accumulate and retain high concentrations of petroleum hydrocarbons and so are unlikely to transfer them to predators such as scalloped hammerhead sharks and Nassau grouper. The fish may be tainted with metabolites bound to tissue macromolecules, but these metabolites are so reactive that it is unlikely that they would be released in a toxic form during digestion by the consumer and so would not pose a serious risk (Neff, 1990). Given that scalloped hammerhead sharks are a coastal pelagic species that utilize a wide range of habitats, we do not expect them to be exposed to dispersants or contaminated prey items for a long enough period of time to result in significant adverse effects. Different life stages of Nassau grouper are more likely to be exposed to dispersants due to their use of nearshore nursery habitats through their juvenile life stage followed by their migration to deeper reef areas where dispersant application might occur.

For these reasons, we believe the application of dispersants may affect, but is not likely to adversely affect scalloped hammerhead sharks and Nassau grouper, especially if BMPs detailed in the previous section are employed during the response action.

4. Corals and Habitat

Elkhorn, staghorn, pillar, rough cactus, lobed star, boulder star, and mountainous star coral:

Invertebrates are known to bioaccumulate hydrocarbons and PAHs (Boehm et al., 2004; Meador, 2003), which can lead to narcosis (Logan, 2007). Various other effects have been noted, including physical smothering because the application of dispersants leads to dispersed oil movement through the water column. These sublethal impacts may lead to mortality, but it is unclear whether, in an oil dispersion situation, PAH concentrations would be high enough, or exposures to PAHs sufficiently long, to cause such impacts on a broad scale (i.e., in a large enough area to reduce the population of ESA-listed corals themselves or affect their prey base).

It has been commonly noted that dispersants are less toxic than oil alone, but that dispersed oil is more toxic than oil alone (Fingas, 2008; NRC, 2005). Therefore, the use of dispersants during the period of coral spawning may have the greatest impact as that is the period when coral larvae will be present in the water column. Reducing the overall area of a spill by rapidly dispersing oil into the water column may result in diminished impacts over a broader area; quickly removing oil from the ocean surface may result in negligible impacts to the planktonic community, though benthic species such as corals could be impacted instead (Varela et al., 2006). It is possible that the addition of oil and dispersant to a natural system may cause a planktonic or benthic community to become dominated by tolerant species that are already present (Ortmann et al., 2012; Atlas and Hazen, 2011; Parsons et al., 1984). Some researchers (Bak and Elgershuizen 1976; Research Planning, Inc. 1986) have suggested the sensitivity of corals to sedimentation and turbidity, based in part on their ability to clear sediment from their tissues, as a corollary to sensitivity to oil. Oil and dispersed oil affect light penetration and, if they come into direct contact with ESA-listed corals, have a smothering affect similar to that of sediment. Elkhorn corals were determined to have a high sensitivity while the star coral complex was found to have an intermediate sensitivity (Shigenaka 2001), which indicates that the use of dispersants may have varied effects on different ESA-listed coral species.

The application of dispersants is not intended for nearshore waters, which would exclude application in areas where ESA-listed coral species are common. Dispersed oil could be carried into the nearshore environment by wind and current transport, but the impacts to habitat are expected to be minimal. In addition, some of the ESA-listed coral species, such as the star corals, have a wide depth range and may be present in areas where deep water applications of dispersants could occur. Depending on the water depths and the level of mixing of the dispersed oil mixture in the water column, dispersed oil could impact colonies of ESA-listed corals. Because oil alone has been found to be extremely toxic to corals, the use of dispersants in areas where direct contact with corals is unlikely due to water depths will shorten the exposure period and reduce the risk of severe impacts to ESA-listed corals from an oil spill.

We believe the use of dispersants could lead to adverse effects to elkhorn, staghorn, pillar, rough cactus, lobed star, mountainous star, and boulder star corals if applied in shallow, nearshore waters, particularly in areas without rapid mixing. However, given the results of the ERA workshops and the dispersant use policy for the CRRT, as well as the required implementation of the BMPs detailed in the previous section during response actions, we believe that the use of dispersants may affect, but is not likely to adversely affect ESA-listed coral species.

Elkhorn and staghorn coral critical habitat and Magnuson-Stevens Act managed coral habitat:

Designated critical habitat for elkhorn and staghorn coral is defined as those areas containing the essential feature which is consolidated hard bottom or dead coral skeletons free from fleshy macroalgae or turf algae and sediment cover. There are separate critical habitat units for Puerto Rico, St. Croix, and St. Thomas/St. John. The use of dispersants is not expected to result in an increase in sediment cover or macroalgal growth on hard bottom habitats or dead coral skeletons. Therefore, we believe the use of dispersants will not affect the essential feature of elkhorn and staghorn coral critical habitat.

Similarly, we believe that the use of dispersants will not adversely affect FMP-managed coral habitat, including coral reefs, coral hardbottoms and octocoral reefs.

5. Reef Fish and Habitat

As previously cited, abnormal development and narcosis are the most often cited modes of toxicity for fish (NRC, 2005). At very low doses, dispersants have been shown to be embryotoxic to fish exposed at early life stages (Lonning and Falk-Petersen, 1978; Falk-Petersen et al., 1983), in particular species such as Nassau grouper where larvae drift into shallow nursery habitats. While the direct application of dispersants is not intended for nearshore waters, dispersion in open water that, over time, results in diluted dispersant concentrations in nearshore waters could have a marked impact on shallow water reef species.

The embryonic and larval life stages of prey species may be found in shallow water during a chemical dispersant application. It is possible that these prey species may be impacted by the dispersant application. Because most aquatic organisms have the ability to metabolize and depurate petroleum hydrocarbon rapidly, we do not expect prey contamination to result in significant impacts to Magnuson-Stevens Act managed reef fish. The trophic transfer of PAHs to invertebrates in dispersed-oil exposures does occur, but fish metabolize PAHs fairly efficiently (Wolfe et al., 2001; Logan, 2007). Therefore, magnification of PAHs in these species through their diet is also unlikely.

Most fish do not accumulate and retain high concentrations of petroleum hydrocarbons and so are unlikely to transfer them to predators such as goliath and Nassau grouper. The fish may be tainted with metabolites bound to tissue macromolecules, but these metabolites are so reactive that it is unlikely that they would be released in a toxic form during digestion by the consumer and so would not pose a serious risk (Neff, 1990). Different life stages of EFH-designated reef fish may be exposed to dispersants due to their use of nearshore nursery habitats through their juvenile life stage followed by their migration to deeper reef areas where dispersant application might occur. However, those exposures are expected to be in low concentrations and of short duration.

For these reasons, we believe that the application of dispersants will not adversely affect FMP-managed reef fish and their habitat.

6. Spiny Lobster, Queen Conch and Habitats

Due to the below-noted similarities, we believe that the use of dispersants is more protective of spiny lobster, queen conch and their habitats, in comparison with the potential impacts of non-dispersed oil.

For the spiny lobster, as previously identified, the most important habitat for juvenile lobster appear to be in *Thalassia* seagrass beds and mangroves. Adult lobster populations are associated with reefs and hardbottoms, mostly with coral outcrops, crevices, caves and ledges.

For queen conch, as stated in CFMC (1996), egg masses are spawned in clean coral sand with low organic content, but have also been reported from seagrass beds. While larvae have been found offshore and can be transported up to 26 miles per day, Posada and Appeldoorn (1994) concluded that most larvae are retained within the area where they spawned. Juveniles are found buried in the sediment, the burial depth changing with size. In the Bahamas, Stoner et al. (1994) found that areas of strong tidal circulation contain a higher number of juveniles. Stoner and Waite (1990) suggested that seagrass biomass, as well as seagrass shoot density were critical features in these nursery habitats. Required habitat for juvenile conch includes among other things a delicate balance between seagrass beds and the surrounding sandy areas. Adult queen conch are found on sandy bottoms that support the growth of seagrasses and epiphytic algae upon which they feed. They also occur on gravel, coral rubble, smooth-hard coral, or beach rock bottoms. Queen conch commonly occur on sandy bottoms that support the growth of seagrasses, primarily turtle grass (*Thalassia testudinum*), manatee grass (*Syringodium filiforme*), shoal grass (*Halodule beaudettei*), and epiphytic algae upon which they feed (Randall, 1964). They also occur on gravel, coral rubble, smooth hard coral or beach rock bottoms and sandy algal beds.

As previously cited, the TROPICS field study, which began in 1983-84 in Panama, was designed to examine the relative short and long-term effects of dispersed crude oil versus non-dispersed crude oil on nearshore tropical marine ecosystems. After baseline studies (1983), two 900 m^2 sites composed of intertidal mangrove and subtidal seagrass-coral zones were dosed (1984) with untreated Prudhoe Bay crude oil and Prudhoe Bay crude oil dispersed with Corexit® 9527 . At periodic intervals over 25 years, the sites were monitored and effects were compared to a nearby reference site.

In the short term, mortality to invertebrate fauna, seagrass, and corals was observed at both the dispersed oil and non-dispersed oil sites. In the long-term (10-25 years), as compared to the reference site, there was little to no oil detected and the ecosystem appeared to have returned to pre-dosing condition at the dispersed oil site. At the non-dispersed site, substrate core samples revealed the continued presence of oil. The effects of non-dispersed and dispersed crude oil at the TROPICS site have been documented and determined to be short- and long-term for various parameters. While dispersed crude oil showed serious and immediate effects on fauna, these effects were short-lived. In contrast, the non-dispersed crude oil site has undergone considerable long-term changes.

In the mangroves, oil was still present in sediments more than 10 years later in both the dispersed and non-dispersed oil areas. In the short term, the oil showed very high mortality to mangroves, but sublethal effects only from dispersed oil. In the long term, even at 10 years, there was still very high mortality, but at the dispersed oil site, after 10 years, growth was similar to the reference site, with some reduction in leafing (Dodge et al, 2014). Mangrove, primarily Rhizophora mangle, repopulation was impeded and substrate erosion was observed at the non-dispersed site. The mangrove forest lost a significant portion of trees which were slowly replaced by "waves" of seedlings that never reached maturity, presumably due to the long-term exposure to toxic petroleum compounds that remained trapped in the substrate.

Dispersants appeared to prevent long-term contamination to mangrove forests and the resulting absence of oil in the substrate, made possible by advantageous properties of dispersed oil, avoided long-term exposure to toxic compounds and provided the conditions for ecosystem and habitat recovery. Most significantly, after 25 years, components of non-dispersed crude oil, specifically aromatic hydrocarbons, remain in the mangrove substrate in the non-dispersed site, where chronic exposure continues to inhibit recovery and repopulation. The long-term (post-ten-year) effects on mangrove fauna at the sites have not been extensively studied (DeMicco et al, 2011).

Coral cover increased at all sites, culminating at approximately 200% over baseline at non-dispersed and dispersed sites, and almost 400% at the reference site. However, through 2001-2002, replacement by flora (seagrass and algae) was apparently a cause at the dispersed and reference sites, but not at the non-dispersed site. By contrast, percent flora in coral habitat increased at the non-dispersed and dispersed sites, while dropping to baseline at the reference site (DeMicco et al, 2011).

The dispersant treatment area for corals immediately following treatment found that coral coverage was greatly reduced, and that growth in one out of four species tested following dispersant use was reduced. However, the recovery and coverage to pretreatment levels happened after 10 years. With oil, there were no effects to coral because the oil traveled on the surface of the water (Dodge et al, 2014).

While dispersed oil has shown short-term effects to seagrass, mangrove and coral habitats, in contrast with considerable impacts of non-dispersed oil, we believe that while the use of dispersants may affect spiny lobster, queen conch, and their habitat, the impacts of dispersants and dispersed oil are expected to be local, short-term and minor in comparison to the impacts of non-dispersed oil.

B. Effects of In-Situ Burning

1. Sea Turtles and Critical Habitat

Green, loggerhead, leatherback, and hawksbill sea turtles:

Sea turtles surface to breathe so there is potential risk of injury from surfacing in the area of the burn. Given that the burn area is relatively small and an in-situ burn is of relatively short duration, the impacts are short-term. The location and timing of the in-situ burning, for example, can be controlled in order to minimize any exposure to wildlife, particularly listed species. The approximate levels of concern for sea turtle inhalation of smoke and gases from in-situ burning have not been determined. Effects from exposures to in-situ burn gas by-products are likely to include irritation of the eyes, nose, throat, as well as potentially life-threatening impacts to an individual due to impacts to air exchange from effects deep in the lung (pulmonary edema) or death due to oxygen displacement. Negatively buoyant residues and those that escape collection could pose some risk of exposure to sea turtles through ingestion. The effects of ingestion of these residues are not completely known. Even if they do cause some toxic effects, exposure is likely to be low considering the small volume of residues produced.

The overall impacts of combustion products, thermal effects, and floating burn residue are minimal. The location and timing of the in-situ burning, for example, can be controlled in order to minimize sea turtle exposure, which is part of the BMPs for protection of ESA-listed species. Therefore, we believe that the use of in-situ burn may affect, but is not likely to adversely affect ESA-listed sea turtles.

Green sea turtle critical habitat:

Designated critical habitat for green sea turtles is composed of seagrass beds and other habitats used by juvenile and adult green sea turtles around Culebra Island, Puerto Rico, and its surrounding islands and cays for refuge and foraging. Residues from in-situ burning that sink to the bottom in shallow, nearshore waters could result in some fouling of sea turtle refuge and foraging habitat in areas containing shallow seagrass beds and coral habitats.

However, given that in-situ burning is not proposed for areas close to shore and that BMPs have been developed to minimize the potential impacts of in-situ burning to ESA-listed species and their habitat, we believe the impacts from the use of in-situ burning on green sea turtle critical habitat around Culebra will be minimal.

Leatherback sea turtle critical habitat:

This critical habitat designation is largely to protect nesting and hatchling sea turtles in waters around Sandy Point, St. Croix. The final rule does not specify particular habitat characteristics. Because in-situ burning is not intended for use in nearshore waters or for areas with special protections, such as Wildlife Refuges, which is the case for Sandy Point, we do not anticipate an in-situ burn occurring within leatherback critical habitat. Residue from in-situ burning could sink to the bottom in leatherback critical habitat resulting in some fouling of habitat. Therefore, we believe the impacts from the use of in-situ burning on leatherback sea turtle critical habitat around Sandy Point will be minimal.

Hawksbill sea turtle critical habitat:

Designated critical habitat for hawksbill sea turtles is composed of reefs and colonized hard bottoms around Mona and Monito Islands, Puerto Rico, utilized as refuge and foraging habitat by adult and juvenile hawksbills. Residues from in-situ burning that sink to the bottom in shallow, nearshore waters could result in some fouling of sea turtle refuge and foraging habitat in areas containing shallow coral habitats. However, given that in-situ burning is not proposed for areas close to shore and that BMPs have been developed to minimize the potential impacts of in-situ burning to ESA-listed species and their habitat, we believe the impacts from the use of in-situ burning on hawksbill sea turtle critical habitat around Mona and Monito Islands will be minimal.

2. Marine Mammals

Sperm, blue, fin, sei, and humpback whales:

Because cetaceans must surface to breathe, there is potential risk of injury from surfacing in the area of the burn. The area where the burn is actually conducted is kept relatively small and an in-situ burn is of relatively short duration, typically only a few hours. The approximate levels of concern for mammals to smoke and gases from in-situ burning have not been determined. Little is known about what would be the smoke exposures levels and durations for animals that come to the water surface to breathe. Effects from exposures to in situ burn gas by-products are likely to include irritation of the eyes, nose, throat, as well as potentially life-threatening impacts to an individual due to impacts to air exchange from effects deep in the lung (pulmonary edema) or death due to oxygen displacement. Though most burn residues float and are collected, negatively buoyant residues and those that escape collection could pose some risk of exposure to cetaceans through ingestion or fouling of baleen. The effects of ingestion of these residues are not completely known. Even if they do cause some toxic effects, exposure is likely to be low considering the small volume of residues produced. Typically, only a small percentage of the original oil volume remains as residue following an in-situ burn. Any unrecovered residue would certainly pose lower exposure risk than the volume of originally released product.

The overall impacts of combustion products, thermal effects, and floating burn residue are minimal. The location and timing of the in-situ burning, for example, can be controlled in order to minimize whale exposure, which is part of the BMPs for protection of ESA-listed species. Therefore, we believe that the use of in-situ burning may affect, but is not likely to adversely affect ESA-listed whales.

3. Fish

Scalloped hammerhead shark and Nassau grouper:

Limited testing of the aquatic toxicity of samples from an in-situ burn have been done using fish species and urchins. Samples were found to be nontoxic to the test species indicating that the in-situ burn residue is no more toxic than weathered oil (Blenkinsopp et al. 1997). Because water temperatures a few centimeters below an in-situ burn remain unaffected and Nassau grouper and scalloped hammerhead sharks do not surface to breathe, the impacts to these species from in-situ burning should be minimal. The only potential impact is associated with burn residue in the form of tarballs though there is no evidence indicating that these species confuse these with prey items. Therefore, we believe the use of in-situ burning may affect, but is not likely to adversely affect scalloped hammerhead shark and Nassau grouper.

4. Corals and Habitat

Elkhorn, staghorn, pillar, rough cactus, lobed star, boulder star, and mountainous star coral:

Residues from in-situ burning that sink to the bottom in shallow, nearshore waters could result in some fouling of ESA-listed corals. It is not expected that residues from in-situ burning are more toxic than weathered oil to corals. In-situ burning is not proposed for areas close to shore and BMPs have been developed to minimize the potential impacts of in-situ burning to ESA-listed species and their habitat. Therefore, we believe the use of in-situ burning may affect, but is not likely to adversely affect elkhorn, staghorn, pillar, rough cactus, lobed star, boulder star, and mountainous star coral.

Elkhorn and staghorn coral critical habitat and Magnuson-Stevens Act managed coral habitats:

Designated critical habitat for elkhorn and staghorn coral is defined as those areas containing the essential feature which is consolidated hard bottom or dead coral skeletons free from fleshy macroalgae or turf algae and sediment cover. There are separate critical habitat units for Puerto Rico, St. Croix, and St. Thomas/St. John. The use of in-situ burning is not expected to result in an increase in sediment cover or macroalgal growth on hard bottom habitats or dead coral skeletons though there could be some fouling of habitat from residues of in situ burning. However, given that in-situ burning is not proposed for areas close to shore and that BMPs have been developed to minimize the potential impacts of in-situ burning to ESA-listed species and their habitat, we believe the use of dispersants will not affect the essential feature of elkhorn and staghorn coral critical habitat.

Similarly, we believe that the use of in-situ burning will not adversely affect FMP-managed coral habitat, including coral reefs, coral hardbottoms and octocoral reefs.

5. Reef Fish, Spiny Lobster, Queen Conch and Habitat:

As mentioned above, limited testing of the aquatic toxicity of samples from an in-situ burn have been done using fish species and urchins. Samples were found to be nontoxic to the test species, indicating that the in-situ burn residue is no more toxic than weathered oil (Blenkinsopp et al. 1997). Because water temperatures a few centimeters below an in-situ burn remain unaffected, and Magnuson-Stevens Act managed reef fish, spiny lobster and queen conch do not surface to breathe, the impacts to these species from in-situ burning should be minimal. The only potential impact is associated with burn residue in the form of tarballs, although there is no evidence indicating that these species confuse these with prey items. Therefore, we believe the use of in-situ burning will not adversely affect FMP-managed reef fish, spiny lobster, queen conch and their habitat.

C. Effects of Dispersants and In-Situ Burning Response Operations

1. Sea Turtles and Critical Habitat

Green, loggerhead, leatherback, and hawksbill sea turtles:

Sea turtles surface to breathe so there is potential risk of injury from surfacing in the area of the burn. However, because dispersants and in-situ burning operations require lesser amounts of vessels and equipment in comparison to physical response operations, it can be expected that exposure of sea turtles to vessels and other equipment involved in dispersant and in-situ burning response operations oil will be significantly reduced.

For in-situ burning operations, the burn area is relatively small and an in-situ burn is of relatively short duration, the impacts are short-term. The location and timing of the in-situ burning, for example, can be controlled in order to minimize sea turtle exposure, which is part of the BMPs for protection of ESA-listed species. Therefore, we believe that the use of in-situ burn may affect, but is not likely to adversely affect ESA-listed sea turtles.

The use of dispersants and in-situ burning are not intended for application in shallow nearshore waters, so the anticipated impacts of associated response operations on eggs and hatchlings are expected to be minimal.

For leatherback sea turtles, which are only found nearshore during nesting season, which is from February to August in the U.S. Caribbean, the potential interaction with associated response operations is expected to have little effect. Similarly, loggerhead sea turtles are uncommon in the U.S. Caribbean, with extremely infrequent reports of nesting and few reported in stranding data. Therefore, we do not expect dispersant and in-situ burning response operations to have significant effects to the Northwest Atlantic Distinct Population Segment.

For these reasons, we believe that the response operations associated with dispersant application and in-situ burning operations may affect, but is not likely to adversely affect sea turtles, especially if BMPs detailed in the previous section are employed during the response action.

Green sea turtle critical habitat:

Designated critical habitat for green sea turtles is composed of seagrass beds, embayments, and other habitats used by juvenile and adult green sea turtles around Culebra Island, Puerto Rico, and its surrounding islands and cays for refuge and foraging.

Given that the use of dispersants and in-situ burning are not proposed for areas close to shore, and that BMPs have been developed to minimize the potential impacts of dispersant and in-situ burning operations to ESA-listed species and their habitat, we believe the impacts from response operations associated with the use of dispersants and in-situ burning on green sea turtle critical habitat around Culebra will be minimal.

Leatherback sea turtle critical habitat:

This critical habitat designation is largely to protect nesting and hatchling sea turtles around Sandy Point, St. Croix. The final rule does not specify particular habitat characteristics. Because dispersant and in-situ burning applications are not intended for nearshore waters or for areas with special protections, such as Wildlife Refuges, which is the case for Sandy Point, we do not anticipate dispersant or in-situ burning response operations within leatherback critical habitat. Therefore, we believe the impacts from dispersant and in-situ burning response operations on leatherback sea turtle critical habitat around Sandy Point will be minimal.

Hawksbill sea turtle critical habitat:

Designated critical habitat for hawksbill sea turtles is composed of reefs and colonized hard bottoms around Mona and Monito Islands, Puerto Rico, utilized as refuge and foraging habitat by adult and juvenile hawksbills.

The application of dispersants and in-situ burning are not proposed for nearshore waters, which would exclude application in much of the designated critical habitat around Mona and Monito Islands. Furthermore, BMPs have been developed to minimize the potential impacts of dispersants and in-situ burning to ESA-listed species and their habitat. Therefore, we believe the impacts from response operations associated with dispersants and in-situ burning on hawksbill sea turtle critical habitat around Mona and Monito Islands will be minimal.

2. Marine Mammals

Sperm, blue, fin, sei, and humpback whales:

Because cetaceans must surface to breathe, there is potential risk of injury from surfacing in the area of dispersant and in-situ burning response operations. Collision with vessels poses a serious threat to some endangered species. Right whales are particularly susceptible to injury or death from vessel collisions because they surface skim-feed and often rest at the surface.

The implementation of BMPs, detailed in the previous section, including vessel speed restrictions when endangered species are present in the area of an oil spill, will ensure that response operations associated with dispersant and in-situ burning operations do not adversely affect ESA-listed coral species.

ESA-listed whale species are present in U.S. Caribbean waters mainly during their winter migration through the area. Humpback whales can be found in the nearshore environment, in particular when calving during their winter migration through the Caribbean. Dispersant and in-situ burning operations are not proposed for nearshore habitats, so the anticipated impacts of associated response operations on humpback whales while they are calving are expected to be minimal.

Because dispersant operations require lesser amounts of vessels and equipment in comparison to physical response operations, it can be expected that exposure of whales to vessels, booms and other equipment involved in dispersant response operations will be significantly reduced.

Likewise, in-situ burning operations are not intended for nearshore habitats. The area where the burn is actually conducted is kept relatively small, and an in-situ burn is of relatively short duration, typically only a few hours. The location and timing of the in-situ burning can be controlled in order to minimize whale exposure, which is part of the BMPs for protection of ESA-listed species.

For these reasons, we believe that the response operations associated with dispersant application and in-situ burning operations may affect, but are not likely to adversely affect ESA-listed whales, especially if BMPs detailed in the previous section are employed during the response action.

3. Fish

Scalloped hammerhead shark and Nassau grouper:

Because Nassau grouper and scalloped hammerhead sharks do not surface to breathe, direct impacts to these species from response operations associated with dispersants and in-situ burning should be minimal.

Given that scalloped hammerhead sharks are a coastal pelagic species that utilize a wide range of habitats, we do not expect them to be exposed to response operations associated with dispersants or in-situ burning for a long enough period of time to result in significant adverse effects.

Different life stages of Nassau grouper may be exposed to dispersant and in-situ burning response operations due to their use of nearshore nursery habitats through their juvenile life stage, followed by their migration to deeper reef areas where dispersant and in-situ burning applications might occur. However, dispersant and in-situ burning operations are not proposed for nearshore waters, and we do not expect Nassau grouper to be exposed to response operations associated with dispersants or in-situ burning in deeper, offshore waters for a long enough period of time to result in significant adverse effects.

For these reasons, we believe that response operations associated with dispersant and in-situ burning may affect, but are not likely to adversely affect scalloped hammerhead sharks and Nassau grouper, especially if BMPs detailed in the previous section are employed during the response action.

4. Corals and Habitat

Elkhorn, staghorn, pillar, rough cactus, lobed star, boulder star, and mountainous star coral:

Vessel traffic, anchor deployment, dragging lines, and physical contact by response workers can detrimentally impact coral reefs and associated habitats. The implementation of BMPs, detailed in the previous section, will ensure that response operations associated with dispersant and in-situ burning operations do not adversely affect ESA-listed coral species.

The application of dispersants and in-situ burning are not intended for nearshore waters, which would exclude application in areas where ESA-listed coral species are common. While some of the ESA-listed coral species, such as the star corals, have a wide depth range and may be present in areas where deep water response operations associated with dispersants and in-situ burning may occur, the potential impacts from vessels, fire booms and other response operations primarily occur at/or within the first few meters of the water’s surface.

Given the results of the ERA workshops and the dispersant use policy for the CRRT, as well as the required implementation of the BMPs detailed in the previous section during response actions, we believe that response operations associated with the use of dispersants may affect, but is not likely to adversely affect ESA-listed coral species.

In-situ burning is not proposed for areas close to shore, and BMPs have been developed to minimize the potential impacts of in-situ burning to ESA-listed species and their habitat. Therefore, we believe that response operations associated with the use of in-situ burning may affect, but is not likely to adversely affect elkhorn, staghorn, pillar, rough cactus, lobed star, boulder star, and mountainous star coral.

Elkhorn and staghorn coral critical habitat and Magnuson-Stevens Act managed corals:

Designated critical habitat for elkhorn and staghorn coral is defined as those areas containing the essential feature which is consolidated hard bottom or dead coral skeletons free from fleshy macroalgae or turf algae and sediment cover. There are separate critical habitat units for Puerto Rico, St. Croix, and St. Thomas/St. John.

The use of dispersants and in-situ burning are not proposed for areas close to shore. In addition, BMPs have been developed to minimize the potential impacts of dispersant and in-situ burning response operations to ESA-listed species and their habitat. Response operations associated with the use of dispersants and in-situ burning are not expected to result in an increase in sediment cover or macroalgal growth on hard bottom habitats or dead coral skeletons.

Therefore, we believe that dispersant and in-situ burning response operations will not affect the essential feature of elkhorn and staghorn coral critical habitat.

Similarly, we believe that response operations associated with the use of dispersants and in-situ burning will not adversely affect FMP-managed coral habitat, including coral reefs, coral hardbottoms and octocoral reefs.

5. Essential Fish Habitat

Based on the temporary nature of dispersant and in-situ burning response operations, and strict adherence to conservation and mitigation measures, impacts on EFH and managed fisheries as a result of dispersant and in-situ burning response operations are expected to be minor. Any impacts on marine habitats are expected to be localized, temporary and minor. And as previously noted, since the utilization of dispersants and in-situ burning are not proposed for nearshore waters, we do not believe that estuarine habitats will be affected by related response operations.

VI. CONCLUSIONS

The intended purpose of dispersants and in-situ burning, used separately or in conjunction with other open-water spill response techniques, is to quickly remove spilled oil from the water surface, thereby reducing exposure to wildlife and preventing contamination of sensitive nearshore and shoreline habitat. Under appropriate conditions, dispersants and in-situ burning can reduce environmental impacts from oil spills, including injury to listed species and critical habitat.

The CRRT believes that the use of dispersants may affect, but is not likely to adversely affect, the listed species or critical habitat present in the zones identified in the CRRT Preauthorization Agreements, and that formal consultation under Section 7 of the Endangered Species Act is not necessary. Additionally, the CRRT believes that the use of dispersants may affect, but is not likely to adversely affect, the listed species or critical habitat present in other waters within its jurisdiction in which dispersant use may be considered.

Similarly, the CRRT believes that the use of in-situ burning, used alone or in conjunction with other open-water spill response techniques, can quickly remove spilled oil from the water surface, thereby reducing exposure to wildlife and preventing contamination of sensitive nearshore and shoreline habitat. Under appropriate conditions, in-situ burning can reduce environmental impacts from oil spills, including injury to listed species and critical habitat. The CRRT has determined that the use of in-situ burning may affect, but is not likely to adversely affect, listed species and critical habitat present in the zones identified in the CRRT Preauthorization agreements, beyond the potential effects of the spilled oil, or add to the cumulative environmental stresses currently acting on the species.

The use of dispersants and in-situ burning offers strong potential for net environmental benefit during an oil spill by allowing for increased protection of biological resources and habitat, as outlined in this biological assessment, and provides for methods to remove oil products from the environment which are more difficult or elusive by other response methods. The CRRT's policies and procedures for the use of dispersants and in-situ burning incorporate measures to minimize overall harm to EFH. Based on the information presented in this assessment, the CRRT has concluded that while dispersant and in-situ burning operations may adversely affect EFH because of direct and indirect impacts, the impacts would be local, short-term and minor.

We request that you concur with these conclusions. Consultation will be re-initiated if additional information not previously considered becomes available, indicating adverse effects to listed species, designated critical habitat or EFH from the identified action.

List of Acronyms

ASTM	American Society for Testing and Materials
ATSDR	Agency for Toxic Substances and Disease Registry
BMP	Best Management Practice
BOEMRE	Bureau of Ocean Energy Management, Regulation and Enforcement
CDC	Centers for Disease Control
CFMC	Caribbean Fishery Management Council
CFR	Code of Federal Regulations
CRRT	Caribbean Regional Response Team
DOC	Department of Commerce
DOI	Department of the Interior
DOR	Dispersant-to-oil ratio
DOSS	Sodium dioctyl sulfosuccinate
DPS	Distinct Population Segment
DWH	Deepwater Horizon
EC50	Half Maximal Effective Concentration
EEZ	Exclusive Economic Zone
EFH	Essential Fish Habitat
EPA	Environmental Protection Agency
ERA	Ecological Risk Assessment Consensus Workshop
ESA	Endangered Species Act
ESCA	Endangered Species Conservation Act
FEIS	Final Environmental Impact Statement
HC5	Hazardous Concentration-5
HPAH	High molecular weight polycyclic aromatic hydrocarbons
FMC	Fishery Management Council
FMP	Fishery Management Plan
FOSC	Federal On-Scene Coordinator
GNOME	General NOAA Operational Modeling Environment
ITOPF	International Tanker Owners Pollution Federation
LC50	Lethal Concentration
LOA	Letter of Agreement
LPAH	Low molecular weight polycyclic aromatic hydrocarbons
MMS	Minerals Management Service
MSFCMA	Magnuson-Stevens Fishery Conservation and Management Act
NCP	National Oil and Hazardous Substances Pollution Contingency Plan
NMFS	National Marine Fisheries Service
NRC	National Research Council
NOAA	National Oceanic and Atmospheric Administration
NOBE	Newfoundland Oil Burn Experiment
OECD	Organization for Economic Cooperation and Development
OSC	On-Scene Coordinator
PAH	Polycyclic aromatic hydrocarbons
RRT	Regional Response Team
SEA	Scientific and Environmental Associates Inc.

SEAMAP	Southeast Area Monitoring and Assessment Program
SFA	Sustainable Fisheries Act
SPAG	Spawning Aggregation
SSD	Species Sensitivity Distribution
TL	Total Length
TOXNET	TOXicology Data NETwork
TPH	Total Petroleum Hydrocarbons
TROPICS	Tropical Investigations in Coastal Systems
USCG	US Coast Guard
USFWS	US Fish and Wildlife Service
USVI	US Virgin Islands
WAF	Water-Accommodated Fraction

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