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Toxicity of Oil to Reef-Building Corals: A Spill Response Perspective

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NOTICE

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TABLE OF CONTENTS

TOXICITY OF OIL TO REEF-BUILDING CORALS:
A Spill Response Perspective

INTRODUCTION

Evaluation of oil toxicity in general is not a simple or easy task. What we consider as "oil" includes qualitatively and chemically very different kinds of materials. Moreover, different species and even different life stages within the same species can react in dramatically different ways to oil exposure. Because the conditions encountered at each spill present a unique set of physical, chemical, and biological circumstances, it is a daunting undertaking to provide general oil toxicity guidance for responders. In the case of oil toxicity as it relates to corals and coral reefs, it is further complicated because there is limited information on the fundamentals of toxicity and toxicology. There is also reasonable doubt about how well information for one area or one species extends across other areas and the hundreds of species currently delineated.

For spill responders and resource managers faced with making choices during oil spills in coral reef areas, the major dilemma is how to use the available science in a meaningful and appropriate fashion. That is, just because someone has studied oil and corals either in the laboratory or in the field does not necessarily mean that the results of that work will help us understand what will occur during a specific spill or cleanup.

The several reviews on oil toxicity and effects to corals are excellent syntheses of the state of knowledge, particularly at the time of their preparation—which is to say, some of these reviews are dated. Loya and Rinkevich (1980), for example, remains a good, well-referenced summary of oil impacts. Knap et al. (1983) provided background information and a summary of study results. Ray (1981) also included a summary of actual incidents along with laboratory results, and then extracted lessons for response and mitigation. A nicely illustrated overview with a solid foundation in the literature was produced for the International Petroleum Industry Environmental Conservation Association (Knap 1992). Neff and Anderson (1981) not only reviewed the literature on oil impacts to a wide range of organisms that included corals, but also performed a series of their own experiments to augment the knowledge base. More general in scope, but useful in providing a broader frame of reference against which oil effects can be evaluated, are the reviews of Peters et al. (1997), Dubinsky and Stambler (1996), and Brown and Howard (1985). Fucik et al. (1984) approach the topic more pragmatically by focusing on damage assessment, recovery, and rehabilitation of corals impacted by oil.

Those reviews attempting to synthesize the many results into coherent lessons or themes about the effects of oil in corals often express a level of frustration over the many challenges related to the task.

> The recognition of a large range of stress responses shown by coral is complicated by the range of oils, oil fractions, and bioassay methodologies used in laboratory studies to date. Consequently, identification of trends or patterns of acute or sublethal responses of corals exposed to oil is difficult
>
> Fucik et al. (1984)

> …extrapolation of…experimental results to the field is hampered by the unnatural spatial scale and duration of the manipulations, and because laboratory and field experiments were not designed to mimic real oil spills.
>
> Keller and Jackson (1993)

The universe of available studies can also be confusing because they often are contradictory: some research indicates serious impacts to corals while others conclude few, if any, consequences of oil exposure. Some coral toxicity reviews critique other reviews. For example, one summary by Johannes (1975) asserted that "…there appears to be no evidence that oil floating above reef corals damages them," but Marshall et al. (1990) pointed out that the studies that led Johannes to this conclusion were largely anecdotal and did not include consideration of sublethal or longer-term effects.

How, then, do we sort through the literature and available science to glean lessons of relevance and utility for spill response? One strategy is to overlay the framework of each study on expected scenarios for actual spills. For example, one parameter that could be evaluated in this way is the nature of oil exposure to test corals in various studies. How did the oil come into contact with the animals? Corals, of course, reside (mostly) in the subtidal, while oil (mostly) floats. What is the best way to simulate the exposure corals might experience during an oil spill? As Ray (1981) commented in his review of oil effects in coral, "Oil preparation and exposures ran the entire gamut...Some of these techniques may simulate actual field exposures, others are unrealistic." Therefore, the *pathway* becomes an important consideration in the process of understanding coral toxicity.

EXPOSURE PATHWAYS

The means by which corals can be exposed to oil has a direct bearing on the severity of resultant impacts. Three primary modes of exposure can be envisioned for coral reefs in oil spills. In some areas (especially the Indo-Pacific), 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 aggregations, this potential 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:

- Are corals in the affected area intertidal?
- Does the spilled oil have a lighter, more water-soluble component?
- 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 make the oil sink?

Areas where corals are intertidal in distribution could be considered to be 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. In this kind of exposure the dose is very high and impacts from physical coating are an added mechanism of toxicity.

Coral exposure via the water column could be a serious route under some circumstances. Because many of the components in oil have 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 be harmed by exposure to enhanced concentrations of dissolved and dispersed oil. The absolute levels of exposure would be expected to be much lower than those experienced by direct contact with intertidal slicks, since only a small fraction of the total oil can be placed into the water column either in solution or physically suspended. However, the components of the oil most likely to enter the water column are those generally considered to have the highest acute toxicity. Corals may therefore be exposed to "clouds" of dispersed oil driven into the water column under turbulent conditions, with impacts dependent on exposure concentrations and length of exposure.

Heavier fuel oils contain fewer of the light fractions identified with acute toxicity than refined and crude oils (although these bunker-type oils are sometimes "cut" with lighter materials to meet customer specifications for viscosity). If they remain on the water surface, spills of heavier fuel oils are less of a concern from a reef perspective, and perhaps more of a concern for protection of other habitats like mangrove forests where they can strand and persist for long periods of time. However, the heavy oils can also weather or be mixed with sediments and increase in density to the point where they may actually sink—providing a direct route of exposure to subtidal habitats and corals. Although acute toxicity characteristics of heavy fuel oils may be lower, the potential for significant physical effects from smothering is greatly increased.

Examining how laboratory exposure methods compare to those likely to be encountered in situ reveals that a fundamental limitation—for our purposes here—of many of the available research studies on the effects of petroleum compounds on corals is that they rely on *nominal* rather than *measured* exposure concentrations. What this means is that the researchers frequently have reported oil exposure concentrations based on the proportions of oil added to water only, and did not actually measure how much of that oil was ultimately mixed into the water as a source of exposure to tested coral colonies.

Why is this problematic? There is likely to be a vast difference between a nominal concentration of oil in water of, say, 100 parts per million (ppm) and the amount that is actually dissolved or accommodated into the water column after mixing: the latter is probably lower by 1-2 orders of magnitude or more. What this means is that in studies where only nominal concentrations are reported, the actual effects levels are much lower than those reported. While nominal concentrations still permit reasonable comparison of effects at different levels *within* a given study, they make it almost impossible to generalize about or compare effects levels *across* studies. This significantly reduces our ability to extrapolate experimental results to a spill scenario.

It is not a new lament and, in fact, this weakness has characterized oil toxicity studies in general for many years. In their review of oil effects studies in the reef environment, Knap et al. (1983) list it as one of the primary difficulties in interpreting the results of available biological investigations.

Further complicating the task of understanding or predicting oil spill impacts in coral reefs based on the experimental literature is the fact that typical real-life exposures are unlikely to be constant. That is, most oil and coral studies are based on exposures to a certain concentration for a certain period of time. This is a standard approach in toxicology. In contrast, an oil spill, whether or not it is dispersed, probably would be characterized with highest exposure concentrations of hydrocarbons at the very beginning of the incident when the product is relatively consolidated in one location and relatively unweathered. This peak exposure could then be expected to decline rapidly and steadily as the spill spread laterally and weathering processes began to change the composition of the mixture. Of course, in a real spill there could be special circumstances (e.g., stormy conditions, intertidal stranding of large amounts of oil, sinking oil) that would alter the expected behavior and exposure of the oil; but the pulsed character of exposure is a reasonable scenario for a generalized spill.

This type of exposure scenario is more difficult to simulate in the laboratory than a constant exposure over a given experimental period. Some researchers model it by removing oil from the experimental system after an amount of time defined to be consistent with the amount of time a slick might reside on the water in an affected area. Others, especially those studying the effects of dispersed oil, introduce a pulse of oil into a flow-through system that is then allowed to dissipate with time. There is no question that methods have become more refined over the years to more realistically portray conditions in a spill, but results must still be interpreted and compared with caution.

This being said—even with the limitations of nominal concentrations and variable and unrealistic exposure conditions that were especially common in earlier coral and oil toxicity investigations, the available results still provide a relatively good survey of the kinds of impacts that might be expected when corals are exposed to oil. Although absolute threshold levels often cannot be derived from the studies, a suite of fundamental oil effects to corals emerges that can serve as the basis for anticipating potential impacts during a spill.

In this report, we intend to present an overview of known toxicity information for oil and corals. Although it will become apparent that a wide range of impacts has been documented over the years, certain patterns and consistencies emerge that we highlight as toxicity "common threads." The focus of our discussion is intentionally narrow, i.e., oil effects on corals themselves and not on the associated reef community of plants and animals. This was necessary to establish some reasonable bounds on the survey and synthesis effort, although we recognize and acknowledge that doing so arbitrarily and artificially limits the assessment of spill effect.

Many others have commented on the interrelated nature of reef communities. In their review of oil spill damage, recovery, and rehabilitation in coral reef systems, Fucik et al. (1984) point out that the discussion of criteria for the assessment of damage and recovery in coral systems must draw the distinction between the coral component alone and the total reef ecosystem. Coral reefs are almost universally recognized as highly productive and sensitive systems. Fucik and colleagues noted that although the hermatypic corals ultimately provide the fundamental structural framework for the entire reef, the coral organisms themselves do not necessarily dominate the biomass, productivity, or calcification. Brown and Howard (1985), in discussing assessment of stress in reef systems, pointed out some of the problems in isolating the corals themselves from the reef dwellers during such a process. They noted that, following a hurricane in St. Croix, scleractinian coral diversity decreased in shallow waters, but diversity of the community as a whole actually increased due to the colonization of new substrate by a wide range of organisms. Their conclusion: "Clearly quantitative measurements on coral reefs affected by disturbance should include some account of all major components of the reef community."

In an ideal world, we would take that advice (of course, in an ideal world, there would be no oil spills). It seems logical that impact assessment in coral reef systems should ideally include the associated community of plants and animals, to acknowledge the ecological linkages and portray a realistic set of spill-related conditions. Realistically, however, that comprehensive approach would enormously complicate the task at hand. Because of the basic role the corals play in the overall reef systems, and because a large-scale impact to the coral component would subsequently affect all associated plants and animals, Fucik et al. suggested that the initial focus in assessing oil spill impacts should be on the corals themselves. Therefore, for this and the pragmatic reason above, we have intentionally chosen to restrict ourselves to corals only in this review.

Similarly, we have not included a comprehensive review and discussion of the literature on chemical dispersants, chemically dispersed oil, and corals. There have been a number of efforts to compare the relative toxicities of oil, dispersants, and dispersed oil to corals. These are summarized elsewhere (see, for example, Hoff 2001). In our review here, we focus on the toxicity information for oil alone, isolating these results from others where possible and appropriate. In some cases, where oil and dispersant data are integrated or the oil-only data difficult to extract from a study, the dispersant results are included as well.

Fucik et al. provided a framework for assessing oil-related damage in a coral reef system(Figure 1). They noted the inherent problems in providing input into this framework, such as community level impacts are most severe but the most difficult to quantify accurately, while organismal impacts can be easier to measure but provide the least information on assessing the overall significance of perturbations on an ecosystem. Nevertheless, their decision tree provides a relevant example of how scientific support personnel in a spill setting could use on-scene observations along with the known science to provide at least some level of guidance. To the protocol Fucik et al. suggested in 1984, we might add additional considerations of sub-organismal levels of consideration such as cellular or even genetic impacts. Recently developed approaches, which will be alluded to later, may provide additional ways to assess impact to reef systems and isolate spill-induced damage from other stressors.

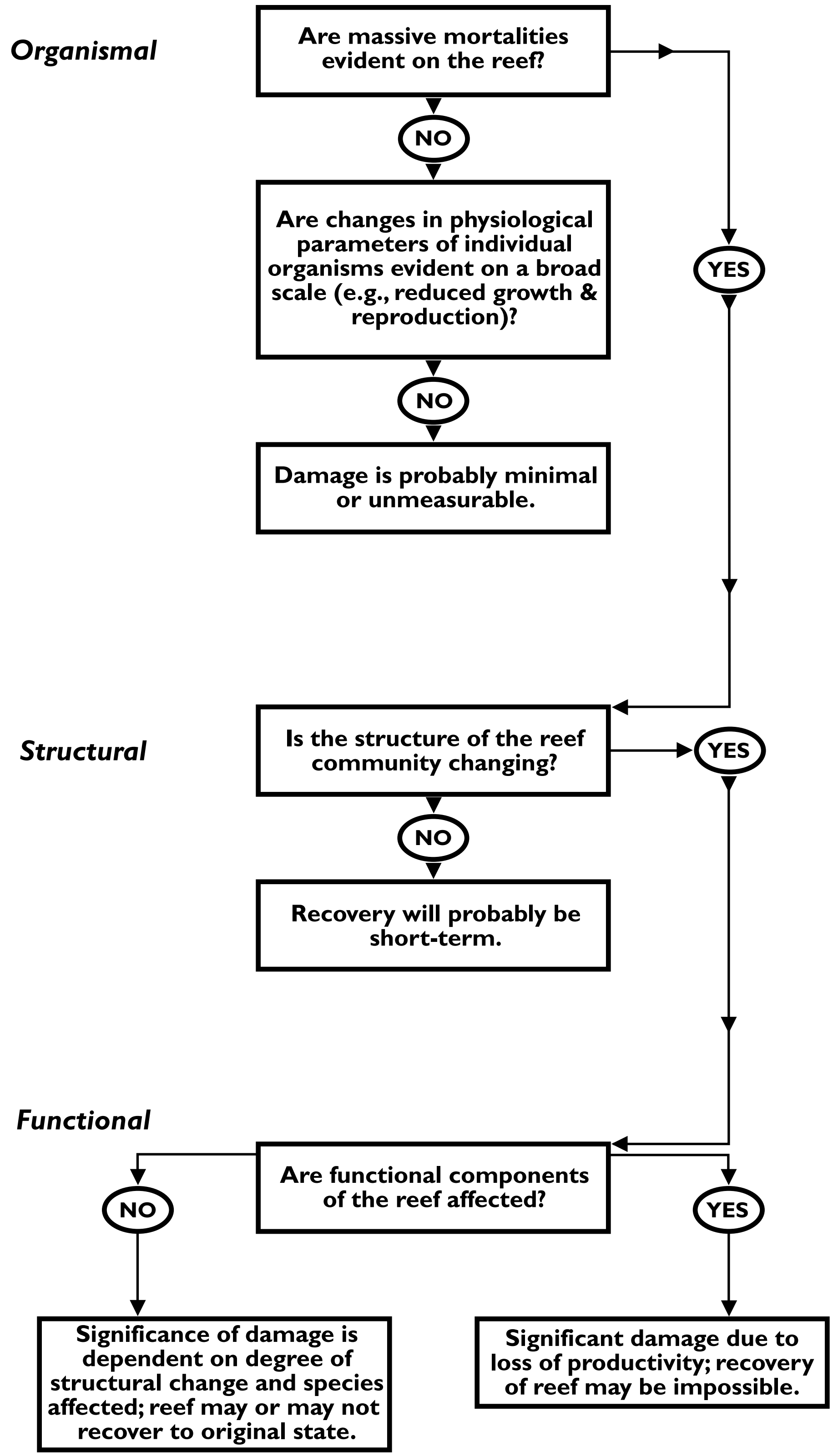

Figure 1. A suggested protocol or framework for initiating studies to measure oil spill damage on corals (from Fucik et al. 1984).

Our discussion of oil toxicity to corals is organized in the following manner. First, we summarize the known spills in coral reef areas and describe larger-scale field experiments. Next, we review documentation of acute toxicity. A section follows this on chronic toxicity effects, a generally well studied subject area that we have chosen to subdivide into categories of effect for purposes of organization. Next, we present individual summaries of information on bioaccumulation, indirect impacts, the influence of reproductive strategy, variability in effect with time and location, species differences, and synergistic effects. These are interpreted with a spill- response orientation to highlight relevant issues during an incident. Finally, we end with some "bottom line"-type insights and advice for responders and resource managers.

CORAL REEF SPILLS, ASSOCIATED FIELD STUDIES, AND LARGE-SCALE EXPERIMENTS

In theory, studies of actual spills in coral reefs should provide the best information on oil impacts in this habitat. In reality, these have been less than illuminating because they are infrequent and often confounded by other sources of disturbance in the reef environment. Of eleven coral reef spill studies cited here, four concluded that little damage occurred, two concluded that major damage occurred, and five neither explicitly nor adequately described effects to the coral reefs themselves.

In theory, large-scale field studies should offer the next-best setting in which to assess oil impact. In reality, these are difficult and expensive to undertake, and thus, even more scarce than spill studies. The two field experiments referenced here indicated little short- or long-term impact of exposure conditions typical of spills.

Case studies of actual oil spill events offer the best opportunity to investigate the effects of oil in a coral reef environment. However, these are also limiting, for at least two reasons: first, despite the seemingly increased risk posed by proximity to major shipping lanes, a relatively small number of major spills have taken place around coral reefs; and second, those spills known to have occurred near coral have not been well-studied. This is especially true for earlier (1960s and before) incidents.

In the absence of an actual oil spill incident, a large- or realistically scaled experimental spill offers an excellent—and, in some ways, better—opportunity to study oil spill impacts to coral reefs. The advantages are the ability to adequately prepare for impact assessment and the enhanced ability to control potentially confounding external parameters. Disadvantages include the reluctance of most environmental regulators to permit the intentional release of a pollutant, regardless of containment; and a significant degree of logistical support (translation: $$) necessary to implement a well-designed and monitored experiment of this kind. Though these studies are even rarer than coral reef spill assessments, two of them are described below.

ACTUAL SPILLS

Argea Prima, Puerto Rico July 1962

On the evening of July 16, 1962, the Italian tanker *Argea Prima* ran aground off Guayanilla Harbor on the southern shore of Puerto Rico. In an attempt to lighten the vessel to free it from the rocks, the captain decided to pump about 72,200 bbl (11,481,000 L) of crude oil into the sea. The oil was blown ashore and transported by currents far to the west. Diaz-Piferrer (1962). is the only published account of this spill that we could find. Diaz-Pifferer had maintained a study site at Guánica before the spill, and so was well positioned to describe changes following the spill. As reported by Diaz-Pifferer, those changes were significant: the physical character of the beaches shifted dramatically and, in some places, all sand was washed away when it formed aggregations with oil and was washed away in the surf*; mangroves habitat in the affected zone was "virtually destroyed" by large amounts of oil; and a widespread and heavy mortality of nearshore animals was described, including adult and juvenile lobsters, crabs, sea urchins, starfishes, sea cucumbers, gastropods such as king helmets and queen conchs, octopi, squids, a variety of fishes (particularly clupeoids), and sea turtles. Large areas were denuded of algae, and sea grass beds (*Thalassia*) were "badly affected."

Although Diaz-Piferrer noted that coral reefs west of the grounding site were initially thickly covered with oil, there was no subsequent discussion of either short-term or long-term impact to corals.

Brother George, Dry Tortugas, Florida January 1964

In January 1964, the tanker *Brother George* spilled 3,610 bbl (573,944 L) of unidentified oil near Bird Key Reef, in the Florida Keys. Jaap et al. (1989) noted that only cursory studies of reef damage took place at that time and thus it is unknown if the oil damaged colonies of *Acropora palmata* there. This short account was the only published documentation of this incident found, and no further details are available.

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* An oil geomorphologist who reviewed this manuscript argued that this erosional mechanism attributed to oiling is not possible.

R.C. Stoner, Wake Island
September 1967

On September 6, 1967, the 18,000-ton tanker *R.C. Stoner* attempted to moor to two buoys when strong winds drove the vessel aground 200 m southwest of the harbor entrance at Wake Island. The only documentation of this incident is found in Gooding (1971).

The tanker was fully loaded with over 142,857 bbl (22,712,500 L) of refined fuel oil products. This included 83,500 bbl (13,275,400 L) of JP-4 aviation jet fuel; 42,500 bbl (6,756,960 L) of A-1 commercial aviation fuel; 10,000 bbl (1,589,870 L) of 115/145 aviation gasoline; 4,000 bbl (635,949 L) of diesel oil; and 3,300 bbl (524,658 L) of Bunker C. There was an immediate release of fuel after the grounding, believed to be primarily aviation gas, JP-4, and A-1 turbine fuel. On the following day, a "considerable quantity" of Bunker C was also observed, and gasoline vapor odor was detected through September 8.

The heavy cargo load and rough seas hampered efforts to refloat the vessel, and on September 8 the stern of the ship broke off. An estimated 14,286 bbl (2,271,250 L) of the mixed fuels covered the surface of a small boat harbor, up to 20 cm thick. The strong southwest winds concentrated the spilled oil in that harbor and along the southwestern coast of Wake Island, which consists of three islets forming an atoll enclosing a shallow lagoon. Large numbers of dead fish were stranded along this shoreline. Oil recovered from the small boat harbor by pumps and skimmers was moved into pits near the shore and burned each evening; over 378,541 L were disposed of in this fashion. The oil was blocked from entering the central lagoon area of the Wake Island group by an earthen causeway.

The Federal Aviation Administration (FAA) cleared dead fish from the shoreline area closest to the spill location. In addition to the massive fish kill (approximately 1,360 kg collected), dead turbine mollusks, sea urchins, and a few beach crabs were also reported. About 2.4 km of shoreline beyond the FAA-cleaned zone was also oil contaminated, and Gooding estimated that another 900 kg of dead fish were not removed. Other dead invertebrates (cowries, nudibranchs, grapsoid crabs) were also observed.

In this assessment, corals in the area were mentioned only in passing, and apparently were not surveyed either formally or informally for impact. Discussion of corals was completely in the context of the associated fish communities. Given the mixture and quantities of fuel spilled, and the massive mortalities manifested in fish and reef-associated invertebrates, there almost certainly was an impact to the coral animals themselves. Gooding did note that, on a survey conducted 11 days after the grounding, the only remaining visible impact in the inner harbor was black oil impregnated in coral. He stated that only cursory observations were made on reef invertebrates and, given external challenges to impact assessment described in the account (typhoons, tropical storm, harassment by black-tipped sharks, skin irritation to divers from exposure to fuels in water), effects to coral were presumably not included in survey objectives.

Ocean Eagle spill, Puerto Rico
March 1968

Morris J. Berman, Puerto Rico
January 1994

These two spills are grouped here and summarized primarily because they were similar in two respects: their location, along the northern shoreline of Puerto Rico, and the lack of reported impact to coral reefs. The two considerations are linked. The northern coast of Puerto Rico has many large hotels and recreational beaches, but few coral reef areas.

The circumstances of the two spills were otherwise rather different. The *Ocean Eagle* grounded on March 3, 1968 in San Juan Harbor, after which it broke in two and spilled 83,400 bbl (13,259,500 L) of Venezuelan light crude oil (NOAA 1992). The *Morris J. Berman* was a barge laden primarily with No. 6 fuel oil and drifted ashore about 300 m off Escambron Beach after its towing cable parted on January 7, 1994 (NOAA 1995).

Impacts of concern, as noted above, were similar. Nearly all of the large tourist hotels and beaches in San Juan are concentrated along the north-central shoreline of Puerto Rico, where much of the oil in both incidents came ashore. Although many of these recreational areas were heavily impacted, sensitive natural resources such as coral reefs and mangroves that would be a major concern elsewhere in Puerto Rico, are not abundant here. Widespread mortalities, primarily among fish and benthic invertebrates, were noted during both the *Ocean Eagle* (Cerame-Vivas 1968) and the *Morris J. Berman* (NOAA 1995) spills; however, there is no mention of adverse effects to corals in either case.

SS _Witwater_ spill, Panama
December 1968

On December 13, 1968, the 35,000 bbl (5,564,560 L) tanker *SS Witwater* broke apart on the Caribbean coast of Panama and released around 20,000 bbl (3,179,750 L) of Bunker C and marine diesel oil. The spill occurred within 5.5 km of the then-new Smithsonian Tropical Research Institute laboratory at Galeta Point.

The incident highlighted the dearth of baseline information on Caribbean intertidal reef flat communities, and the Smithsonian lab invested a substantial effort to compile those data. These data provided background information for experimental tests of effects of oil, reported in Birkeland et al. (1976).

Rützler and Sterrer (1970) did report on damage to tropical communities, including corals, from the spill. They referred to the recent, incomplete baseline surveys of unimpacted communities, but based their assessment of actual and potential damage on comparisons with data from other areas in the tropical Atlantic.

The authors reported that coral reefs, consisting mainly of *Porites furcata*, *P. asteroides*, *Siderastrea radians*, and *Millepora complanata* (a hydrocoral), seemed to be the least affected of all the communities studied. At the time of the survey (2 months after the spill), no ill effects were observed. Rützler and Sterrer attributed this lack of impact to the subtidal nature of the reefs (i.e., lack of direct contact) and a higher than normal low tide caused by high winds.

Tarut Bay, Saudi Arabia
April 1970

During a storm on April 20, 1970, a pipeline broke on the northwest shore of Tarut Bay in Saudi Arabia. An estimated 100,000 bbl (15,898,700 L) of Arabian light crude oil entered the shallow bay. The spill was briefly documented in Spooner (1970), who described the cleanup actions (which included the use of the chemical dispersant Corexit 7664), and the initial mortalities of intertidal invertebrates (crabs and mollusks), fish, and one rat. Oiling was noted on dwarf mangroves and three months later some of these trees died. Spooner did not describe any impacts of the spill on coral reef areas, although she did comment that in a shallow (2.5-4.6 m) area of Tarut Bay directly east of the pipeline break area, an area of *Acropora* sp. was found "…growing healthily with abundant and diverse associated fauna…in an area which had been subjected to potential oil pollution from effluent and from terminal accidents for the past 25 years." This seemingly implied that the coral was not particularly sensitive to either the acute exposure of the pipeline spill or the chronic exposure of proximity to normal bulk loading.

T/V *Garbis* spill, Florida Keys
July 1975

On July 18, 1975, the tanker *Garbis* spilled 1,500 to 3,000 bbl (238,481 to 476,962 L) of crude oil into the waters approximately 48 km SSW of the Marquesas Keys, Florida. The oil was blown ashore along a 56-km stretch of the Florida Keys, east of Key West. The only published description of this spill and its impacts are found in Chan (1977), although the 1976 M.S. thesis of that author at the University of Miami in 1976 further detailed effects and recovery. The source of the spill, the *Garbis*, was identified after the publication of both documents (E. Chan, pers. comm., 2000).

In addition to documenting early impacts, Chan established a series of sites to be monitored over a longer period of time (a year). Since no pre-spill information was available, effect and recovery were judged through comparison with unoiled, biologically similar locations.

Several habitats were impacted and described. Some of the impacts were severe, e.g., mortalities in echinoderms and pearl oysters. However, a notable lack of spill effect was found in coral reef areas. Reefs were surveyed by divers immediately following the spill and subsequently in August and November 1975 and January 1976. Chan attributed this lack of impact to the fact that the reefs were completely submerged during the spill, and to calm seas that minimized water column contact with the oil.

***Peck Slip*, Puerto Rico**
December 1978

On December 19, 1978, the barge *Peck Slip* was damaged in heavy weather in the Pasaje de San Juan along the eastern side of Puerto Rico. Approximately 10,476-10,952 bbl (1,665,580-1,741,290 L) of Bunker C oil were discharged while the barge was towed to Puerta Yabucoa, its original port of departure. Gundlach et al. (1979) and Robinson (1979) reported that four habitats were surveyed for possible damage from the spill: sand beaches, gravel/cobble beaches, mangrove forests, and offshore lagoons and coral reefs. Although 26 km of shoreline was oiled, most of the oiled areas were sand beaches. Diver surveys of coral reef areas did take place; despite some evidence of oil in bottom sediments, Gundlach et al. reported no observed biological impacts to this habitat.

Robinson (1979) provided more detail on the assessments that took place, including those targeted on coral reef habitats. Diving surveys were conducted in one area of offshore coral reefs (Río Mar). The reef types were classified as nearshore fringing reefs and offshore patch reefs. Constituent corals were identified as *Montastrea annularis*, *M. cavernosa*, and *Diploria strigosa*. Robinson attributed the generally low diversity of corals in these habitats to high turbidity.

Aerial observations of the reef areas showed the presence of oil and sheens on the water. In addition sediment cores taken at offshore stations revealed "very light oil" had been incorporated into most of the bottom sediments in the Río Mar area. However, no oil was seen on the surface of bottom sediments or on corals, and the observed organisms "exhibited no abnormal behavior."

Bahía Las Minas, Panama
April 1986

On April 27, 1986, about 240,000 bbl (38,157,000 L) of medium-weight crude oil (70 percent Venezuelan crude, 30 percent Mexican Isthmus crude) spilled from a ruptured storage tank at a petroleum refinery at Bahía Las Minas, on the central Caribbean coast of Panama. Of this amount, Keller and Jackson (1993) estimated that at least 60,000-100,000 bbl (9,539,240 -15,898,700 L) spilled into the waters of Bahía Las Minas. According to Jackson et al. (1989), this was the largest recorded spill into a sheltered coastal habitat in the tropical Americas. It also happened to occur near the Smithsonian Tropical Research Institute's Galeta Marine Laboratory. As a result, the Bahía Las Minas spill was extensively studied, and the results from those investigations constitute a large part of the available field effects literature for oil impacts to coral reef (and other tropical) communities.

The area where this spill occurred was not pristine before the 1986 incident. Guzmán et al. (1991) pointed out that a wide range of human activities spanning more than a century has extensively modified and degraded Bahía Las Minas. . Nevertheless, the team of researchers studying the effects of the Bahía Las Minas spill concluded that the incident had major biological effects in all environments examined, including the coral reefs and reef flats. Widespread lethal and sublethal effects were noted. In coral reefs, the cover, size, and diversity of live corals decreased substantially on oiled reefs after the spill. Apparent sublethal impacts included decreased growth, reproduction, and recruitment. Keller and Jackson (1993) summarize and synthesize the results from the large team of investigators; results from coral researchers are cited more specifically in the discussions to follow.

Although Corexit 9527 oil dispersant was used during the initial response to this spill, Keller and Jackson termed the overall dosage of dispersant as "low," and concluded that the limited use of the chemicals could not explain the widespread subtidal biological impacts reported.

Gulf War spill, Arabian Gulf
January 1991

During the waning days of the Gulf War conflict in 1991, the Iraqi military deliberately discharged oil, causing the largest oil spill in history, variously reported to be between 6.3 million and 10.8 million bbl (1,001,620,000-1,717,060,000 L). Between 19 and 28 January, 1991, oil was released from two major sources: three Iraqi tankers anchored in the Kuwaiti port of Mina Al-Ahmadi; and the Mina Al-Ahmadi Sea Island terminal area (Tawfiq and Olsen 1993). Aerial fallout from the 730 oil wells destroyed by the retreating Iraqi forces indirectly contaminated the nearby marine waters (Saenger 1994).

Given the magnitude of this release and the previous coral reef impacts noted at other tropical spills, there were dire expectations of severe impacts to nearshore and offshore reefs in Kuwait and Saudi Arabia. However, to date the extent of coral reef damage directly attributable to the Gulf War spill has been remarkably minor.

Downing and Roberts (1993) surveyed the nearshore and offshore reefs in 1992 and noted rather equivocal indications of effect in ostensibly heavily impacted areas. For example, a reef at Qit'at Urayfijan was very likely covered by oil released from at least one tanker and the Mina Al Ahmadi terminal. While the reef is never exposed to the atmosphere, Downing and Roberts stated that crude oil probably flowed over it for days. This reef was clearly impacted, mostly in shallower water, with mortalities noted in large colonies of *Platygyra* as well as in most of the *Porites* (Downing and Roberts did not identify corals observed to species). New growth, however, was observed in nearly all dead portions of coral.

In contrast, Downing and Roberts also reported on conditions at Getty Reef, close to a visibly oiled beach and directly downstream from known release points. Here, there was no evidence of recent coral kills or even stress among *Porites*, *Platygyra*, *Cyphastrea*, *Leptastrea*, *Psammocora*, *Favia,* and *Favites*, and the associated fish community was especially robust.

The authors do not dismiss the possibility of a spill impact on the coral reefs, but suggest that it may be obscured by the effects of other environmental factors. They also discussed potential indirect impacts of the conflict, such as reduced water temperature and ambient light due to smoke cover, lasting several days, from oil fires.

Vogt (1995) established six 50-m study transects nearshore and offshore the Saudi Arabian shoreline to document effects and recovery from the 1991 Gulf War oil spill. On the basis of video recordings made along these transects between 1992 and 1994, Vogt concluded that live coral cover had significantly increased and that the corals offshore from Saudi Arabia had survived the largest spill on record "remarkably unscathed."

Saenger (1994) summarized many of these same results and commented on the contrary nature of the findings relative to expectations of damage in the wake of the massive release. In addition to discerning no demonstrable direct effects of oil, Saenger further noted that spawning activities of staghorn corals (*Acropora* sp.) were not impaired in either 1991 or 1992. He suggested three possible reasons for the accelerated "self-purification processes:"

- Acclimatization and proliferation of efficient, oil-degrading microorganisms due to long periods of exposure to natural seeps and releases;

- Exceptionally high ambient temperatures, which increase both rates of volatilization of lighter fractions of oil and of intrinsic biodegradation;
- Enhanced rate of photo-oxidation due to lack of cloud cover and shallow depths of the Arabian Gulf.

Other Gulf War spill researchers have suggested alternate explanations for the minimal impact reported to coral reefs. For example, Michel (pers. comm., 2001) noted that Kuwaiti crude oil has one of the highest rates of emulsification of any crude and combines with water to form a very stable emulsion (cannot be broken with chemicals, heat, or re-refining). The rapid and persistent emulsification may well have prevented most of the oil from entering the water column to harm corals. Moreover, the Arabian Gulf is a low-energy system, so there would be little mixing of oil into the water column.

Whatever the causes or conditions responsible for the noted lack of adverse effect, Saenger cautioned that these and other unique conditions found in the Gulf region suggested that extrapolating oil impact (or lack thereof) to other areas, such as the Great Barrier Reef, was probably not appropriate.

Rose Atoll National Wildlife Refuge, American Samoa October 1993

On October 14, 1993, the Taiwanese fishing vessel *Jin Shiang Fa* ran aground on the southwestern side of Rose Atoll, a remote coral reef in eastern American Samoa. In 1974, Rose Atoll was designated as a National Wildlife Refuge because of its fish and wildlife resources, which include giant clams and green sea and hawksbill turtles. The grounding spilled 2,381 bbl (378,541 L) of diesel fuel and other materials (lube oil and ammonia) onto the reef. Previously, this had been considered one of the most remote and pristine coral reefs in the world. Green et al. (1997) consolidated information and performed impacts assessments for the U.S. Fish and Wildlife Service, which administers the National Wildlife Reserve.

Rose Atoll is a unique coral habitat in Samoa, in that crustose coralline algae (identified as primarily *Hydrolithon onkodes* and *H. craspedium*) dominate instead of the hermatypic corals. The reef-building corals are present; Green et al. list *Favia*, *Acropora*, *Porites*, *Montipora*, *Astreopora*, *Montastrea*, and *Pocillopora* as being the common genera.

All of the petroleum products and ammonia were released into the marine environment over a period estimated as six weeks. Wave action reportedly mixed oil and oily debris from the wreck down onto the reef structure.

Although the injury to the corals from the grounding was judged to be "moderate to high," it was not possible to ascertain causal factors in a more specific way. Several possible injury pathways were identified:

- Fuel and other contaminant toxicity;
- Mechanical damage from the grounding and subsequent debris impacts;
- Anoxia due to mortalities in the reef community;
- Smothering and scouring from sediments created by the wreck;
- Competition from opportunistic algae; and,
- Bleaching from direct and indirect impacts of the incident.

While the physical effects of the grounding were obvious and long-term, the authors contend that the most widespread and severe injuries to the atoll seemed to be due to the release of diesel fuel. A massive die-off of coralline algae and many reef-dwelling invertebrates was observed after the release; blue-green algae blooms were recorded where they are typically not found; and the structure of algal communities had shifted substantially. Four years after the grounding, the affected areas remained visibly impacted—particularly with respect to coralline algae cover—and Green et al. cast some doubt as to whether Rose Atoll would ever return to its former pristine condition.

Review of these case studies does not yield a preponderance of evidence either for or against a finding of oil spills consistently causing damage to coral reefs. Some of the early spill accounts might be discounted as coral effects benchmarks, since the corals themselves did not appear to be high priorities for injury assessment when those studies took place. However, the more recent and intensively studied spills in Panama (Bahía Las Minas) and the Arabian Gulf (Gulf War) yielded one conclusion of moderate to severe damage, and one of little to no damage, respectively. Perhaps the one truism that emerges is that the unique characteristics of oil spills do not permit extrapolation very far beyond the individual circumstances of each incident.

LARGE-SCALE FIELD EXPERIMENTS

Tropical Oil Pollution Investigation in Coastal Systems (TROPICS), Panama
December 1984; September-November 1994

In 1984, the American Petroleum Institute (API) sponsored a multi-year experiment in which a representative tropical system (comprised of mangrove, seagrasses, and coral) was exposed to oil and chemically dispersed oil. The experimental design was intended to simulate a severe, but realistic, scenario of two large spills of crude oil in nearshore waters. Ballou et al. (1987) detailed the original experiment and its findings.

In 1994, the Marine Spill Response Corporation (MSRC) and API sponsored a revisit to the experimental site by much of the original research team. The findings from the ten-year follow-up studies were published as Dodge et al. (1995).

Although both efforts encompassed oil and chemically dispersed oil effects studies in mangrove, seagrass, and coral systems, only oil in corals will be discussed here. The reader is directed to the referenced documents for information and details on the other study components.

The sites selected for the experiment were located on the Caribbean coast of Panama, in northwestern Laguna de Chiriqui. The area was extensively surveyed, and experimental sites were chosen based on suitability of the three habitat types to be studied. Porites porites and *Agaricia tennufolia* dominated coral reefs. Ultimately, three sites (oil, dispersed oil, and untreated reference) were selected. The oiled site was treated with 953 L of Prudhoe Bay crude oil, which was released onto a boomed area of the water surface and allowed to remain for about two days. Tides and winds distributed the oil over the study area. After the exposure period free-floating oil was removed with sorbents.

Chemical monitoring of water exposure was extensive, using continuous fluorometry, fixed-wavelength ultraviolet fluorometry, and discrete water samples analyzed using gas chromatography (GC), and mass spectrometry (MS). The continuous fluorometry indicated mean exposure (hourly averages) of 1.4 and 2.5 parts ppm under the slick. Discrete water samples taken during the oiling treatment and analyzed by GC and GC/MS showed low exposure levels ranging from 0.01 to 0.09 ppm. Targeted analyses for low molecular-weight hydrocarbons ranged between 33 and 46 parts per billion (ppb), which was about two orders of magnitude less than levels at the chemically dispersed study plots.

For coral reefs, detailed transects measured abundance of epibiota living on the reef surface. Four measurements were taken: total organisms, total animals, total corals, and total plants. Growth rates of four coral species (*P. porites*, *A. tennufolia*, *Montastrea annularis*, and *Acropora cervicornis*) were also measured.

Of all the parameters listed above, the only statistically significant effect documented over the first 20 months at the oiled site was a decrease in coral cover. No significant changes in growth rates of the four targeted corals were noted.

The follow-up survey in 1994 showed no significant consequences to coral cover or coral growth. The authors contrasted the finding of no impact from oiling alone to that described by Guzmán et al. (1991) at Bahía Las Minas, where significant effects of oil alone were found in several of the same species studied at TROPICS. Dodge et al. implied that the greater damage may have been due to the size of the spill and continued chronic exposure at Bahía Las Minas.

Arabian Gulf field experiment
September 1981-September 1982

LeGore et al. (1983; 1989) described a large-scale field experiment carried out on Jurayd Island, off the coast of Saudi Arabia in the Arabian Gulf. The authors acknowledged the inconsistency in the scientific literature regarding oil impact on corals, and pointed out that much of the literature consisted of opportunistic observation of uncontrolled oil spills or controlled laboratory experiments under artificial and unrealistic conditions. They offered their experimental design and its results as a response to the stated need for more realistic toxicity testing and field verification of lab test results.

The study was designed to determine responses of corals to dispersed oil under realistic spill conditions, but the design included exposure to crude oil only (Arabian light) among its four exposure scenarios. Two exposure time periods were selected: 24-hr, and 120-hr. Study plots were established over coral reefs comprised mostly of *Acropora* spp. (more than 95 percent), with scattered colonies of *Platygyra* sp., *Goniopora* sp., and *Porites* sp. The plots measured 2 m x 2 m, located over approximately 1-m depth at low tide, and anchored in place. These were surrounded by oil-containment boom measuring 7.5 m x 7.5 m. Two plots were established for each treatment.

The stated intent of the experiment was to simulate conditions of a typical Arabian Gulf oil spill and not to overwhelm the corals with "extraordinary and catastrophic stresses." As such, oil was added to test plots to produce a slick of 0.25 mm thick, a total of 14 L in the 24-hr oil only treatment; and 0.10 mm and 5.63 L in the 120-hr experiment. Water concentrations of hydrocarbons were measured by infrared methods, and all measurements were below detection limits in the oil-only plots.

The oil-only plots were visually inspected at the end of the 24-hr and 120-hr exposures, and they appeared normal. These areas were monitored for one year, and no extraordinary changes occurred relative to the unoiled plots (seasonal changes in degree of bleaching, however, were noted across all monitored plots). While dispersed oil appeared to delay the recovery from seasonal bleaching, this was not observed in the oil-only plots.

Growth rates, expressed as skeletal extension along branch axes, showed no correlation to treatment in the 24-hr exposure. There was some indication that growth rates were depressed with 120-hr exposure, but LeGore et al. cautioned that these were not definitive.

The authors made the following conclusions:

- No visible effects were exhibited over a 1-year observation period by Arabian Gulf corals exposed to floating crude oil corresponding to a slick thickness of 0.25 mm.

- Corals exposed for 5 days to floating crude oil corresponding to a slick thickness of 0.10 mm exhibited no visible effects during the 1-year observation period.
- Coral growth and colonization of study plots appeared unaffected by exposure to crude oil in both the 1-day and 5-day experiments.
- There was a similar lack of impact in dispersant and dispersed oil experiments, although corals exposed to dispersed oil for 5 days showed some delayed stress reactions and possibly some synergistic impacts from cold winter temperatures.

There were few observed effects in the two large-scale field experiments that have examined oil and corals. With only two studies of this type available, generalizations can be made only cautiously. What these experiments do show is that it is possible to have realistic exposure scenarios with oil on the water over coral reefs and have few, if any, demonstrable impacts. However, they do not rule out the potential for impacts to corals from oil, and the LeGore et al. studies suggest an enhancement of impact with the use of chemical dispersants.

ACUTE EFFECTS

Contrary to some earlier research findings and review conclusions, exposure to oil and oil spills has been shown to cause acute oil toxicity. Some studies have involved somewhat extreme exposure scenarios, but other, more realistic experiments have demonstrated relatively rapid toxic impact. Other studies have shown that a brief exposure may not result in immediate death, but does so after an extended period of time.

A review of laboratory and field studies on acute effects of oil to corals can be confusing. Widespread coral mortalities following actual spills have been reported only infrequently, even when (as reported by Ray 1980) associated reef dwelling organisms have perished. Fucik et al. (1984) suggested that acute toxicity impacts were probably not a good indicator of oil effect, and stated it is more likely that adverse effects to the coral would be manifested in sublethal forms.

There were no reported mortalities of corals after the unprecedented Gulf War spill. As previously detailed in the summaries of larger-scale experiments, the 1984 TROPICS experiment in Panama (Ballou et al. 1987; Dodge et al. 1995) showed no short- (0-20 months) or long-term (10 years later) effect to corals in an intentionally oiled zone. LeGore et al. (1989) found a similar lack of effect in their field experiment.

Shinn (1989), arguing that oil is not among the biggest threats to coral survival, related results of a simple qualitative experiment he performed. He placed pieces of staghorn and star corals in plastic bags of Louisiana crude oil and seawater and left them exposed for 90 minutes before returning them to clean seawater. Shinn said that corals appeared normal the next day, as well as fourteen days later. He attributed this apparent lack of effect (and that from a subsequent half-hour immersion in pure Louisiana crude) to the protective qualities of mucus.

However, there are many notable exceptions to a conclusion of little apparent acute oil toxicity in corals. These include the previously mentioned 1986 Bahía Las Minas spill in Panama, whose effects were extensively documented. In that incident, researchers found widespread coral mortality attributed to the spill, as will be detailed below.

Results from laboratory experiments investigating acute toxicity of oil to corals are somewhat equivocal, and have shown a range of impacts. While it is not possible (nor is it our intent) to reconcile the apparent contradictions, some of the variation may derive from differences in exposure methods. That is, laboratory studies on acute toxicity of oil have involved several different means of exposing test corals to oil. These have included complete immersion in refined and crude products, coating with oil, and mixing oil into water and using only the water for experiments. In an actual spill, reef corals would be expected to be directly exposed to oil infrequently, if not rarely. Nevertheless, the more direct (one might say extreme) exposures provide a useful endpoint for understanding the acute toxicity of the tested oils on the tested corals.

Elgershuizen and deKruijf (1976) examined the acute toxicity of four crude oils (Nigerian, Forcados, Tia Juana Pesado, and Forcados long residue) to the hermatypic coral *Madracis mirabilis*. Oil and water test solutions were prepared in two ways: water soluble fraction (WSF) of oil floating on seawater, in which a known quantity of oil was floated on the surface of seawater for 24 hrs before it was removed and the remaining seawater used for dilutions; and as an oil-seawater mixture, in which a known quantity of oil was added to seawater and stirred for 24 hrs (dispersant/seawater and oil/dispersant/seawater mixtures were also tested; those results may be found in the original reference). Toxicity endpoints for the experiment were RD_{50} and LD_{50}, with those terms being defined somewhat unconventionally in this study: RD_{50} concentrations were the 50 percent response doses after the 24 hr test period; LD_{50} concentrations were the 50 percent mortality doses after an additional 24 hrs of recovery in running seawater.

Nearly all *M. mirabilis* colonies exposed to solutions derived from oil on the surface recovered, and thus no mortality curve could be generated. Recovery was also quite high for solutions of Nigerian and Forcados crudes mixed with seawater. Elgershuizen and deKruijf found that oils mixed with seawater were more toxic than solutions from oils floated on the surface. Test solutions from Forcados long residue and Tia Juana crude were more toxic, but lower concentrations did not induce permanent damage. It is, however, unclear how the authors defined "permanent" within the context of this experiment.

Elgershuizen and deKruijf concluded that the oils alone were of low acute toxicity to coral colonies. In this case, the authors were much more concerned about the toxic impacts of chemical dispersants, alone and in combination with oils.

Johannes et al. (1972) conducted oil exposure experiments on 22 species of coral at Eniwetok Atoll. Because the upper portions of coral reefs in the Pacific region are sometimes exposed during low spring tides, the researchers were interested in studying how oil affected the corals under these conditions. Two specimens of each of the coral species were floated in frames with a portion of each exposed to the air and the remainder submerged. Santa Maria crude oil was then poured into the water around one frame containing the corals, but not directly on them (the other frame was left as a control). Natural wave action coated the exposed surfaces with oil. Exposed corals were left in the frame for about 1.5 hrs, at which point all specimens were placed in clean water. They were then observed over the next four weeks.

Branching corals, such as those in the genera *Acropora* (the most abundant genus in the Indo-Pacific region) and *Pocillopora*, were most susceptible to oil coating and retention. Corals such as *Fungia* sp. and *Symphyllia* sp., which are characterized by large fleshy polyps and abundant mucus, retained almost no oil after immersion for a day and showed no subsequent damage. Members of the genera *Turbinaria*, *Favia*, *Plesiastrea*, *Favites*, *Psammocora*, *Astreopora*, *Symphyllia*, *Montipora*, and *Porites* showed intermediate oil affinities. In locations on the colonies where oil adhered in patches greater than a few mm in size, "complete breakdown" of tissue occurred. Areas where oil did not adhere appeared healthy. Control colonies remained healthy throughout.

It seems clear from this study that direct contact with crude oil kills coral when the oil adheres. This contrasts with the several other studies that indicated that proximity of submerged corals to surface oil generally resulted in few discernible acute impacts. For example, Johannes (1975) described unpublished experiments in which he and others floated five types of oil over groups of the Hawaiian corals *Porites compressa*, *Montipora verrucosa*, and *Fungia scutaria* for 2.5 hrs. No visible evidence of injury was found over 25 subsequent days of observation.

The observation by Johannes et al. that branching corals were more susceptible to oil exposure is consistent with the findings of others who have made field studies in oil affected areas. Guzmán et al. (1991, 1994) noted that in the Bahía Las Minas spill, nearly all branching corals were killed and thus, longer-term studies could be performed only on massive species of coral. Similarly, Hudson et al. (1982) found that, in areas around oil production wellheads, massive species like *Porites lutea* preferentially survived over branching genera like *Pocillopora* or *Acropora*. The latter showed an estimated 70-90 percent reduction.

The differential susceptibility to oil seems to be more closely linked to physical form than taxonomy. This distinction can be confusing, since the literature (e.g., Dodge et al.1995; LeGore et al.1989) describes field evidence supporting the notion of *increased* tolerance to oil for some species of *Acropora*—the same taxon described as being *less* tolerant above. This may be explained by the fact that species of *Acropora* assume many forms, from massive to arborescent (branching). Moreover, colonies within the same species can assume different forms, depending on environmental or biological conditions (Veron 1986).

In a sparsely described scoping experiment, Grant (1970) placed two of each specimen of *Favia speciosa* in three 15-gallon aquaria. In two of the tanks, he floated about three pints of Moonie crude oil, and in one of these he varied the water level daily to simulate tidal change. In this way, oil was permitted to contact the corals for about five minutes over five days. At the end of eight days, oil was removed from the two exposed aquaria. The corals were maintained in tanks for another 16 days and Grant related that all were alive and "apparently unaffected." It is worth noting that he qualified the results heavily and concluded, "The experiment described above cannot be regarded as definitive: it is indicative only, and calls attention to a need for substantially expanded inquiries…"

Cohen et al. (1977) studied the effect of Iranian (Agha Jari) crude oil on colonies of the Red Sea octocoral *Heteroxenia fuscescens* under both static and continuous flow assay conditions. For static tests, coral colonies were placed in 3 L of aerated seawater and oil was introduced at nominal concentrations of 1, 3, 10, and 30 ml/L (oil was simply added to the water surface at the various calculated concentrations after a 3-hr acclimation period). Exposure time was 96 hrs. LC_0, LC_{50}, and LC_{100} concentrations were then calculated.

Flow-through assays were also conducted in 1500-L fiberglass tanks. The highest oil concentration used was 10 ml/L, added as a single dose on the surface of the water. The exposure period was 168 hrs.

Table 1 summarizes acute lethality results from both experimental setups. The 96-hr. LC_{50} concentration was determined to be 12 ml/L. The results reflect the fact that differences in experimental exposure (static or flow-through) do affect toxicity. This becomes more evident at longer exposure times, with lower toxicity results in the flow-through setup.

Table 1. Concentrations of Iranian crude oil in ml/L added to medium at start, fatal to 0, 50, and 100 percent of *Heteroxenia* colonies under static and continuous flow conditions, from Cohen et al. (1977).

	Static Continuous				
	24 hr.	48 hr.	72 hr.	92 hr.	168 hr.
LC_0	30	3	1	1	10
LC_{50}	>30	>30	17	12	>10
LC_{100}	>30	>30	>30	30	>10

Although the results suggest a relatively low overall toxicity of crude oil to *Heteroxenia fuscescens*, the previously discussed consideration of nominal rather than measured exposure concentrations should be mentioned again: the water column concentration resulting from the specified nominal concentrations of oil simply added to water would be much lower than those listed.

Reporting an actually measured concentration would result in a much higher toxicity result. As a consequence, interpreting and applying the findings to situations that might be encountered during a spill is quite difficult. Because of the uncertainties inherent with nominal exposure concentrations, the apparent differences with time and with the nature of experimental exposure (static or continuous) are more interesting. Cohen et al. showed that toxicity of a given concentration of oil increased with longer time of exposure, and that continuous flow conditions result in lower toxicity values relative to static.

One conclusion we can draw from this study is that experimental setup type can affect toxicity results. In a broad analysis of aquatic toxicity, Mayer and Ellersieck (1986) compared 123 paired static and flow-through toxicity results and found that for the group of chemicals they studied, static test most often (53 percent) resulted in lower acute toxicity than flow-through tests. The flow-through was less toxic in only 10 percent of the pairs. . Mayer and Ellersieck suggested that degradation or hydrolysis products of the given contaminant that accumulated in static test vessels might have been an influence in the latter cases; this might be the case for petroleum hydrocarbons and corals. Cohen et al. themselves attributed the lower toxicity in flow-through exposures to "…the continuous removal (of oil) in the medium and probably to a reduced biomass per unit volume in tanks as compared to jars."

Birkeland et al. (1976) performed a series of three experiments with hermatypic corals from the eastern Pacific and from the Caribbean. Results were also reported in Reimer (1975). Coral species used were *Pocillopora* cf. *damicornis*, *Pavona gigantea*, *Psammocora stellata*, and *Porites furcata*. These experiments involved some of the most extreme oil exposure scenarios encountered in our review. While it seems unlikely that natural reef corals would experience these kinds of exposures in an actual spill, the results help to establish one end of the range of acute effects.

In their first experiment, two colonies of *Pocillopora* cf. *damicornis* were completely submerged in marine diesel oil for 30 minutes. The corals were then rinsed thoroughly and placed in an aquarium. Control colonies were similarly treated, with seawater substituted for the marine diesel.

> Results: The exposed colonies initially survived a 30-minute exposure to pure marine diesel, despite tissue rupture especially at the edges of the colonies. After 17 days, however, 70 percent of the polyps were dead and those still living had mouths open and mesenterial filaments extended (controls showed none of these responses and lived for 35 days without significant tissue death). The authors noted a "massive" initial extrusion of zooxanthellae.

In the second study, two colonies of *P.* cf. *damicornis* were placed in marine diesel, three in Bunker C oil, and three in seawater, for exposure periods of 1 minute. Branches and small colonies of *Pavona*, *Psammocora*, and *Porites* were also subjected to similar experimental conditions to evaluate species differences.

> Results: All the colonies had similar degrees of tissue death for the first week; but after 13 days obvious differences between oiled and unoiled *P.* cf. *damicornis* were noted. For example, within 13 days colonies exposed to marine diesel lost nearly all living tissue and those exposed to Bunker C lost 70-84 percent. After 16 days, both oil-exposed groups had lost nearly all living tissue. In contrast, control colonies sustained over 95 percent.

In the third experiment, an individual branch on a given colony was exposed to diesel, Bunker C, or seawater, for 30 seconds. Another branch on the same colony was left untouched.

In all cases, oil was removed from experimental colonies by submerging them in seawater and removing the resulting surface film with an absorbent tissue. They were then rinsed in running seawater for 30 minutes.

> Results: Somewhat equivocal, few differences over a 1-month period; some sharper differences appeared after 71 days, but no trends were evident that could be related to treatments. After 109 days, all colonies treated with Bunker C as well as one control were dead; the remaining controls and the marine diesel colonies had low cover of living tissue. Several colonies showed extensive bleaching within 5-13 days after the experiment commenced, but all but one had recovered by day 27.

Reimer (1975) reported a fourth experiment in this series, one in which colonies of the four species were exposed to 1-4 ml marine diesel added to the surface of 50 ml or 250 ml finger bowls, for periods of either 30 minutes or 4 hrs.

> Results: Oil on the surface of the water caused synchronized contractions in *P.* cf. *damicornis* colonies, along with mouth opening. When corals were exposed for 4 hrs, 32 to 50 percent of the colonies died over a 93-hr. post-exposure period. For the other three species, a similar mouth-opening response was observed after exposure. This reaction was transient, with duration after return to clean seawater lasting from 15 minutes in *Pavona* to 4 days in *Porites*.

Rinkevich and Loya (1977) have performed a number of studies with Red Sea corals, especially *Stylophora pistillata*. Although most have examined chronic or longer-term consequences of oil exposure, they also reported on the acute effects of water-soluble fractions of Iranian crude oil on coral planulae. The test mixture was prepared by mixing 1 part crude oil with 99 parts seawater, with the resulting aqueous solution considered as 10 ml/L (the authors acknowledged that only a minor portion of the oil would have actually dissolved in the seawater).

At 144 hours, none of the planulae had died in the control exposure, while more than 50 percent had died in the 1, 5, and 10-ml crude oil/L seawater aqueous fractions.

Rinkevich and Loya also tracked *S. pistillata* colony mortality rates in the field, at a chronically polluted reef near the oil terminals of Eilat, and at an unoiled reference reef 5 km to the south. They checked 59 healthy colonies in the polluted area and 39 in the control area for viability every four months. After one year, 42.3 percent of the oiled area colonies had died, compared to 10.3 percent of the controls. This represented a significant ($p < 0.01$) difference in mortality.

There was no mention in the referenced article of exposure documentation, which would significantly bolster the inferred link between chronic oil contamination and coral viability.

Te (1991) studied the toxicity of gasoline, motor oil, and benzene to Hawaiian reef coral planulae (*Pocillopora damicornis*). Te performed open- and closed-system (i.e., sealed to limit volatilization) bioassays with gasoline:oil mixtures and with benzene. For the gas:oil mixture exposures, three replicates each of 15 planulae were exposed to nominal concentrations of: 5, 10, 50, and 100 ppm mixtures of gasoline and motor oil in 50 ml petri dishes (open system); and 1, 5, 20, and 100 ppm in 200 ml sealed bottles (closed system). Hourly observations were made for the first 6 hrs, 6-hr intervals for three days, and

12-hr. intervals for the final 13 days. For benzene studies, a closed system setup as detailed for gasoline was established.

Te did not document any mortality from gasoline:oil mixtures in the open system experiments, although total mortality occurred with the 100-ppm concentration in the closed system. No mortality was observed in the benzene exposures. Te concluded that in contrast to many organisms that readily showed mortality at even very low concentrations, *P. damicornis* planulae seemed to be resistant to oil exposure.

A field study of the Bahía Las Minas oil spill in Panama (Jackson et al. 1989) reported an extensive mortality of both intertidal reef flat corals (*Porites* spp.) and subtidal reef corals (*Diploria clivosa*, *Porites astreoides*, and *Siderastrea siderea*) that was attributed to the spill. *S. siderea* was found to have been particularly vulnerable, with new partial mortality disproportionately common on heavily oiled reefs one year after the spill. Burns and Knap (1989) commented that these findings of acute impact from a spill stand in sharp contrast to the conclusions from laboratory dosing experiments and small-scale field studies suggesting only transient effects.

A longer-term summary of impacts in Bahía Las Minas can be found in Guzmán et al. (1994). They assessed acute (recent mortality) as well as sublethal (growth) impacts to the species listed above. At heavily oiled reefs, percentages of recently injured (as identified by bare white or lightly overgrown skeleton exposure) corals were higher for all three species. However, there were peaks immediately after the spill and also during another period spanning 3-5 years post-spill. The latter impacts were attributed by Guzmán et al. to be linked to a series of diesel fuel spills at the electrical generation plant in Bahía Las Minas.

This comment by Guzmán et al. suggests one of the major difficulties in trying to isolate impacts attributable to a specific spill incident from other possible sources of impact: the fact that a myriad of other natural and human-induced influences can affect a community. This is the downside to the real-world example embodied in an actual spill.

Harrison et al. (1990) describes a set of experiments performed in Australia using a Great Barrier Reef coral species (*Acropora formosa*) . This laboratory study is particularly interesting because it showed both acute and chronic toxicity impacts from oil in the water (water-accommodated fraction, WAF) at measured concentrations which might be encountered during a spill. Branches of *A. formosa* tolerated 6-hr. exposures to 5-10 ppm (measured in the water) marine fuel oil, but with 12 to 24 hr. exposure to the same concentrations, the colonies became stressed (increased mucus production, expelled zooxanthellae) and died. After 48 hrs, virtually all of the tissue in the 5 and 10-ppm treatments had disintegrated. The chronic and potential indirect toxicity results will be discussed in subsequent sections.

Gardiner and Word (1997) and Gardiner et al. (1998) also documented acute and chronic toxicity effects in a branching coral (*Acropora elsyii*) exposed to water-accommodated fractions (WAFs) of fresh and artificiallyweathered Campbell condensate and Stag crude oil. Artificial weathering consisted of a distillation process in which the source oil products were heated to drive off the more volatile fractions and simulate the changes that occur to oil in the environment. The water-accommodated fractions were prepared using modifications to a standard protocol developed by Environment Canada (Blenkinsopp et al., 1996), and the 100 percent WAF concentrations of total petroleum hydrocarbons were estimated to fall between 2 mg/L and 20 mg/L. Table 2 shows (concentrations of the 100 percent WAFs measured by infrared spectrophotometry. For toxicity tests, 100, 50, 10, and 0 percent concentrations of the stock WAF mixture were used.

Table 2. Measured (by infrared spectrophotometry) replicate concentrations of 100-percent solutions of water-accommodated fractions of test oils used in coral toxicity experiments. Concentrations in µg/L (ppb). From Gardiner and Word (1997).

Oil/Treatment	Concentrations (ppb)	
Fresh Stag	219	273
Stag/200°C	138	140
Stag/250°C	83	102
Fresh Campbell	7900	540
Campbell/150°C	3430	949
Campbell/200°C	638	126

Table 2 shows a generally consistent trend of decreasing solubility and/or accommodation of hydrocarbons into the water with increased weathering. This makes intuitive sense, since the more volatile fractions lost during weathering are also the more soluble. The table also shows the distinct differences in solubility characteristics with different oil products (condensate and crude). Finally, the replicate measurements for the Campbell condensate in particular illustrate the great variability that can occur in the preparation of nominally identical mixtures.

Gardiner and Word (1997) did not elicit an acute toxicity response in 144-hr tests except for the fresh Campbell condensate at 100 percent strength. Coral fragments exposed to the full-strength fresh Campbell WAF experienced 100 percent mortality, which occurred in the first hours of exposure. Sublethal exposure experiments are discussed below in the chronic toxicity section.

A petroleum product that is attracting an increasing amount of interest from power generating entities worldwide is a material known primarily by its trade name, Orimulsion. Orimulsion is a natural bitumen in a freshwater emulsion, stabilized by the addition of non-ionic surfactants. The bitumen is designated as Cerro Negro and is produced in the Orinoco Belt in eastern Venezuela. It is being marketed as a cheaper alternative fuel for power generation, but environmental concerns have complicated its ready acceptance and approval in some countries (e.g., the U.S.).

Brey et al. (1995) reported on toxicity evaluations they conducted for Orimulsion and its major constituents, and for No. 6 fuel oil. Among the tests performed were acute toxicity studies using the coral *Tubastrea aurea*. The methods for these tests were not described in great detail. It appears that exposure concentrations for petroleum products were measured, but only as a gross analysis of oil and grease. In the case of the No. 6 fuel oil solutions, the stock mixture was held under refrigeration for one week before tests were performed.

Acute toxicity results (96-hr. LC_{50}) in *Tubastrea aurea* were reported to be 112 mg/L (ppm) for Orimulsion; values for bitumen and No. 6 fuel oil were not calculated "because no mortality was observed at the highest concentrations used" (25.8 and 43.97 mg/L, respectively).

Although Brey et al. commented that *T. aurea* seemed to be more sensitive to Orimulsion than to the other petroleum products, the uncertainties related to methodologies suggest cautious interpretation of these results. It is unclear whether concentrations used for calculating LC_{50} values were nominal or measured. Water-accommodated fractions were apparently chemically analyzed using a gross oil and grease methodology. Moreover, stock solutions, especially that for the No. 6 fuel oil, were not prepared in a conventional way (i.e., held under refrigeration for an extended period). These considerations complicate interpretation of the results within the study and severely limit comparison to other toxicity results.

CHRONIC EFFECTS

Chronic effects of oil exposure have been consistently noted in corals and can be substantial, ultimately killing the colony. A number of chronic impacts have been described, including histological, biochemical, behavioral, reproductive, and developmental effects. Cumulative impacts resulting in mortality are also suggested. Field studies of chronically polluted areas and manipulative studies in which corals are artificially exposed to oil also suggest that some coral species are more resistant to the detrimental effects of oil than other species.

It may be stating the obvious that oil spills can take many forms, and that a catastrophic release of oil such as a spill or blowout may also result in chronic, or long-term sublethal impacts to an area. In addition, chronic effects to sensitive resources like coral reefs may also occur without an identifiable incident, i.e., via non-point source contamination. In either case, ample research evidence demonstrates that oil can cause a number of sublethal but serious impacts to coral. It is a fairly robust literature, and is characterized by a qualitatively broad range of impact.

Coral researchers such as Guzmán et al. (1994) have suggested that oiled corals perform a tradeoff between functions related to exposure response (e.g., cleaning and damaged tissue regeneration) and normal energy expenditures (e.g., growth and reproduction). The literature on chronic and sublethal effects of oil on corals supports this, and the resulting studies focus on the questions of whether oil increases the stress responses or decreases normal physiological functions. It is reasonable to presume that the reallocation of energy in the face of stress imposed by a spill would ultimately reduce the fitness of the affected corals, as would be expected for any organism responding to any stress.

Field studies examining chronic effects are less common than laboratory experiments because of the length of time necessary to study longer-term effects of oil exposure to corals, the lack of control over environmental conditions that may influence results, and the generally more subtle measurements necessary to document a sublethal impact. . Those field efforts that do take into account longer-term effects often rely on gross or more integrative measures of health, such as areal cover or simple presence and absence. An example of such a study is described by Bak (1987), who compared coral reef status in a chronically contaminated embayment on the island of Aruba in the southern Caribbean Sea. Between 1929 and 1985, a large oil refinery (heavy Venezuelan crude) operated in this location with, as listed by Bak, "…all accompanying sources of pollution such as spills, refinery waste water discharge and eventually cleanups with dispersants (Corexit)." A continuing chronic source of contamination—a sheen at the harbor entrance—was noted a year after the refinery was closed.

Bak surveyed 24 species of coral in the study, and concluded that there appeared to be a clear relation among the condition of the reef structure and the coral communities, the location of the oil refinery, and the current pattern. That is, the major deterioration of the reef occurred in front of the refinery and immediately downcurrent. Coral species such as *Montastrea annularis* and *Agaricia agaricites* were absent in these areas but became abundant upstream of the facility. In contrast, *Diploria strigosa* showed a completely opposite distribution, suggesting an increased tolerance to oil (the possibility that *D. strigosa* is more resistant to oil exposure was supported by the laboratory studies of Dodge et al. 1985).

Some reviewers have asked how well laboratory studies of chronic effects relate to actual field conditions. That is, do the artificial strictures of the laboratory unrealistically skew results so that they have minimal relevance to a real spill scenario? As we have noted, field and laboratory studies of oil toxicity to corals often yield conflicting or confounding results. In an effort to address this, Rinkevich and Loya studied the impacts of chronic exposure to oil in the field (1977) and then compared these results to those obtained in long-term laboratory exposures (1979). Loya, Rinkevich, and their colleagues have researched oil effects to Red Sea corals (the branched hermatypic coral, *Stylophora pistillata* in particular) for the last 20 years. They have used a number of sublethal responses as endpoints for their studies and, at least for *S. pistillata*, have generated a substantial body of information for chronic impacts of oil exposure. Rinkevich and Loya obtained results that were consistent with both the field and laboratory conclusions, which provides at least one set of experiments suggests the relevance of laboratory results to field situations.

An interesting approach to combining the most desirable aspects of lab experiments (e.g., controlled exposure conditions) with those of the field (realistic environmental conditions) was that of Dodge et al. (1984) in which corals were exposed to oil in the laboratory but then moved to a field setting for subsequent long-term observation (around a year). Their results will be discussed later.

We reviewed many studies of chronic effects of oil exposure to corals. A number of studies used multiple endpoints for exposure impact, and it became apparent that we would need to organize or group the results in some fashion for summary and synthesis. Fucik et al. (1984) created a list of laboratory studies documenting sublethal oil impacts. Their table, updated to include additional endpoints and more recent studies, is reproduced below as Table 3.

The table is instructive in that it shows the broad range of impacts from oil exposure that researchers have identified; it also links the researchers with endpoints and serves as a quick reference if a reader has an interest in a specific kind of effects endpoint. The list here, however, is not necessarily comprehensive with respect to studies reviewed for this report.

We have chosen to group the reported endpoints into the following categories: behavioral, fecundity and reproduction, larval; histological, calcification and growth, surface cover, photosynthesis, and mucus and lipids. This list is also not a comprehensive one, but it seems to encompass most of the reported effects in the literature. Any that did not fit into the eight categories will be discussed separately.

<u>Behavioral Endpoints</u>

Lewis (1971) exposed four species (*Porites porites*, *Agaricia agaricites*, *Favia fragum*, and *Madracis asperula*) collected on the west coast of Barbados to an unspecified crude oil by soaking strips of filter paper in the petroleum and then submerging them near but not in physical contact with test corals. The exposure period was 24 hrs. Lewis used three behavioral endpoints as effects indicators: tentacle extension, feeding, tentacle retraction upon stimulus ("tactile," in table below), and development of ruptures in the oral disks through which septal filaments were extruded ("septal filaments absent," in table). The results of the oil exposures are summarized below in Table 4.

Table 3. Stress responses shown by corals exposed to oil and oil fractions (adapted from Fucik et al. 1984).

Response	**References**
Tissue death	Johannes et al. (1972); Reimer (1975); Neff and Anderson (1981); Wyers et al. (1986)
Impaired feeding response	Reimer (1975); Lewis (1971); Wyers et al. (1986)
Impaired polyp retraction	Cohen et al. (1977); Elgershuizen and de Kruijf (1976); Neff and Anderson (1981); Knap et al. (1983); Wyers et al. (1986)
Impaired sediment clearance ability	Bak and Elgershuizen (1976)
Increased mucus production	Mitchell and Chet (1975); Peters et al. (1981); Wyers et al. (1986); Harrison et al. (1990)
Change in calcification rate	Birkelund et al. (1976); Neff and Anderson (1981); Dodge et al. (1984); Guzmán et al. (1991, 1994)
Decreased growth (wet-weight biomass) rate	Gardiner and Word (1997)
Gonad damage	Rinkevich and Loya (1979b); Peters et al. (1981)
Impaired fertilization	Negri and Heyward (2000)
Premature extrusion of planulae	Loya and Rinkevich (1979); Cohen et al. (1977)
Larval death	Rinkevich and Loya (1977)
Impaired larval settlement	Rinkevich and Loya (1977); Te (1991); Kushmaro et al. (1996); Epstein et al. (2000); Negri and Heyward (2000)
Coenosarc tissue damage	Peters et al. (1981)
Expulsion of zooxanthellae	Birkelund et al. (1976); Neff and Anderson (1981); Peters et al. (1981)
Decrease in chlorophyll a	Gardiner and Word (1997)
Change in zooxanthellae primary production	Neff and Anderson (1981); Cook and Knap (1983); Rinkevich and Loya (1983)
Muscle atrophy	Peters et al. (1981)

Table 4. Range of oil concentrations (ppm, nominal) at which impairment in specified parameter was noted in 50 percent of test colonies, by species (Lewis 1971).

TENTACLE EXTENSION			
Porites porites	*Madracis asperula*	*Favia fragum*	*Agaricia agaricites*
100-200 ppm	10-50 ppm	10-50 ppm	200-500 ppm
FEEDING			
Porites porites	*Madracis asperula*		
100-200 ppm	10 ppm		
TACTILE			
Porites porites	*Madracis asperula*		
>1000 ppm	10-50 ppm		
SEPTAL FILAMENTS ABSENT			
		Favia fragum	*Agaricia agaricites*
		500-1000 ppm	>1000 ppm

From our current perspective, the major shortcoming of this study stems from the use of nominal oil exposure concentrations only. The nominal oil concentrations make it difficult to compare or extrapolate these results to other situations. If we assume an internal consistency in the way the corals were exposed to oil (perhaps a big assumption) then patterns across species emerge: *Madracis asperula* was the most sensitive to oil exposure, and *Agaricia agaricites* least sensitive. The other two species were of intermediate susceptibility. Lewis suggested that branching forms such as *Porites* and *Madracis* were more affected by exposure than the other two encrusting species. Johannes (1972) also noted this link between form and susceptibility for acute effects. (Lewis also examined the toxic effects of exposure to an oil dispersant but those results are not presented here).

The problem in assuming that oil exposure was consistent with consistency in methodology is that others have made the comparison (see Table 2) and shown that actual exposures presumed to be equal can vary considerably, over an order of magnitude. Experiments relying on nominal oil concentrations have to be interpreted with great caution.

In addition to acute lethality, Cohen et al. (1977) used simple behavioral endpoints in determining effects of Iranian crude oil to *Heteroxenia fuscescens*. As described in the preceding section on acute toxicity experiments, the researchers used both static and flow-through setups to assess effects of exposure to Iranian crude oil at four nominal concentrations (1, 3, 10, and 30 ml/L). They used two behaviors—abnormal pulsation and abnormal response to mechanical stimulation—and found that oil-exposed colonies showed a reduced pulsation rate and an uncoordinated response to stimulation.

Cohen et al. concluded that, while their exposures did not result in substantial acute toxicity, they also suggested that sublethal exposures could produce latent adverse effects that would not be reflected in short-term assays.

Wyers et al. (1986), Knap et al. (1983), and Cook and Knap (1983) were companion studies designed to examine toxicity of chemically dispersed oil to the brain coral, *Diploria strigosa*. As a part of these studies, the effects of physically dispersed oil (i.e., no chemical dispersant) were also assessed. The experimental design was intended to inject a degree of spill realism into coral and oil studies, by using a flow-through system that mixed Arabian Light crude oil with water (also with dispersants as well). *D. strigosa* colonies were exposed for a 24-hr. period. Wyers et al. performed an initial study, presumably for scoping purposes, targeting 1,5, and 20 ppm total oil concentration; because most of the colonies exposed at 1-5 ppm appeared unaffected, subsequent experiments (including those of Knap et al.) used 20 ppm as their targeted exposure concentration. As illustrated in Figure 2, measured concentrations (hexane extracts by fluorescence spectrometry) showed a fair amount of variability but were close to the desired level.

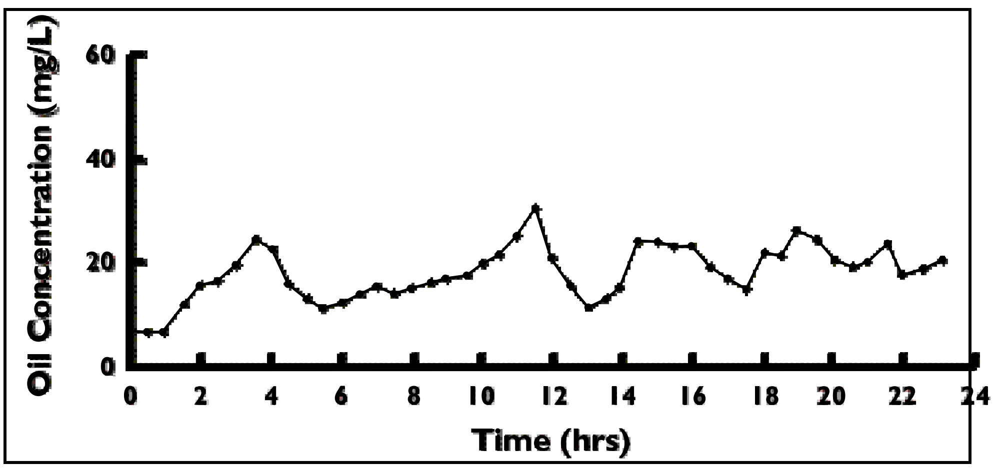

Figure 2. Fluctuations of oil concentrations measured by fluorescence spectrometry during 24-hr dosing for physically dispersed oil; nominal concentrations were 20 ppm. From Knap et al. (1983).

Wyers et al. used tentacle extension as the response measure for exposure, along with other parameters such as the presence of adherent mucus/oil, mesenterial filament extrusion, loss of pigmentation, and tissue swelling. They showed that while exposure to 20-ppm oil in the water column induced stress-related symptoms, normal appearance was usually resumed within 2 hrs to 4 days following the 24-hr exposure period. They concluded that the observed effects were unlikely to impair overall coral viability. Wyers et al. did indicate that tissue rupture and associated lesions could be an exception due to the possibility of increased infection or predation. However, these lesions were found only after exposure to the highest tested concentration, 20 ppm, and not with 1-5 ppm.

Knap et al. continued the work of Wyers et al. and found that the oil-exposed corals were characterized by a decline in tentacle expansion. One week after the 24-hr exposure, expansion behavior typical of the controls was recorded. No longer-term differences in skeletal growth were observed with treatment, although the authors noted that high intercolony variability made differentiation of potential subtle impacts difficult.

Bak and Elgershuizen (1976) studied the abilities and patterns of 19 species of hermatypic corals to reject, or cleanse themselves, of oiled sediment. The oiled sediment results were referenced against patterns for clean sediment. Corals were collected from the fringing reefs of the southwest coast of Curaçao. The oils used were four of the same oils employed by Elgershuizen and deKruijf (1976)—Nigerian, Forcados, Tia Juana Pesado, and Forcados long residue—with the addition of Lagomar short residue.

Bak and Elgershuizen could find no evidence of oil adsorption to coral tissues, and no sign of active ingestion of oil droplets. Oil introduced into and onto the corals was actively cleared by the colonies. If the species was a mucus secretor, oil could be incorporated into mucus for up to 5 hrs. There was no difference in clearance rates or patterns between oiled sediment and clean sediment.

<u>Fecundity and Reproduction</u>

In a laboratory experiment, Rinkevich and Loya (1979) split *S. pistillata* colonies, with one portion being held as a control and the other exposed to oil. The exposed colonies were held in tanks and Iranian crude oil was introduced on the surface of the water once each week over periods of two and six months. While oil did not contact the colonies directly, they were exposed to the water-soluble fraction of the crude floated on the water's surface.

The researchers found that after two months, 75 percent of the exposed *S. pistillata* colonies showed a significant decline in the number female gonads per polyp. After six months, Rinkevich and Loya documented a significantly higher mortality among exposed colonies than in the controls (no mortality had been noted after two months). They suggested that this reflects a latent cumulative effect of chronic oil exposure.

The same researchers (Rinkevich and Loya, 1977) performed field studies of the effect of chronic oil contamination on *S. pistillata* colonies. They noted a significant difference in the number of colonies with gonads in polyps between an unoiled reference area and a reef located near oil terminals in the northern Gulf of Eilat (Red Sea). One hundred three mature colonies from the oil port area and 98 from the control reef were studied. About 75.5 percent of control colonies contained gonads in polyps, compared to 44.6 percent in the oil terminal colonies.

Guzmán and Holst (1993) studied the major reef-building coral species, *Siderastrea siderea*, in the Bahía Las Minas (Panama) area, focusing on potential reproductive impacts from chronic oil exposure. They measured coral fecundity, as reflected by the number of gonads per polyp and gonad size, in heavily oiled and unoiled reefs. The study took place 39 months after the major 1986 spill, summarized earlier in the case studies section.

Guzmán and Holst found no difference between oiled and unoiled areas in the number of coral colonies with gonads at any stage of development and also in the number of gonads per colony. However, gonad size did vary significantly with oiling, with larger gonads occurring at unoiled reefs. The authors suggested this reflected a stress-induced lowering of fecundity at the oiled sites, and that gonad size might be used as a sensitive indicator of coral viability.

In a more direct look at the effects of oil on critical reproductive processes in corals, Negri and Heyward (2000) performed a series of studies to determine the effect of water-accommodated fractions of a heavy Australian crude oil (Wandoo) on fertilization success for the broadcast spawning coral, *Acropora millepora* (they also tested WAFs from production formation waters, chemical dispersant mixed with the crude, and neat dispersant). Egg and sperm were isolated from Great Barrier Reef colonies and used for the fertilization assays. The test mixtures of egg, sperm, contaminant WAF, and filtered seawater were held for 4 hrs before termination and preservation.

Negri and Heyward estimated the total hydrocarbon concentration in their stock crude oil WAF mixture (100 percent) to be 1.65 ppm by means of UV fluorescence. In the absence of dispersant, the WAF of crude oil alone failed to inhibit fertilization up to a concentration of 0.165 ppm total hydrocarbon, or a ten- percent dilution of the stock solution. Dispersant alone was less toxic (significant fertilization inhibition at 10 ppm), but one and ten percent mixtures of dispersant and crude oil were more toxic (inhibition at 0.225 and 0.0325 ppm, respectively). Dispersed crude oil completely inhibited fertilization at 0.325 ppm, equivalent to a 1- percent dilution of crude-dispersed stock mixture.

<u>Effects to Larvae and Larval Development</u>

Rinkevich and Loya (1977) documented the numbers of planulae released from *S. pistillata* colonies in chronically oiled and control reefs. The two areas were defined by proximity to an oil port facility, and apparently no chemical measurements were made of exposure conditions. They performed this study by enclosing 35 large colonies in each area with plankton nets, and found that most (68.6 percent) in the control reef released more than five planulae per coral head, while most in the contaminated reef (85.7 percent) released less than five. Rinkevich and Loya concluded that coral fecundity in the control reef was four times greater than in the chronically contaminated area (this was based on the total number of planulae collected at the two colony groups, 181 in oiled vs. 772 in control).

The authors also studied the settlement and viability of planulae in the two areas, and found that colonization onto artificial settlement plates was significantly higher in control reef areas.

In another series of experiments, Loya and Rinkevich (1979) exposed *S. pistillata* colonies to water-soluble fractions (WSFs) of Iranian crude oil (1:100 stock mix). Actual concentration of oil in the water was not determined. They found that the crude oil WSF induced the coral to prematurely expel larvae into the water, and additionally large numbers of symbiotic zooxanthellae were also shed (in the absence of disturbance, *S. pistillata* usually sheds larvae only at night). The authors also observed abnormal movement in the larvae, even at the lowest dilution exposure. Loya and Rinkevich suggested that the incompletely developed planulae released by the stressed corals would have a low probability of survival.

In their experiments with *Heteroxenia fuscescens*, using Iranian crude oil, Cohen et al. (1977) found that exposure to the oil resulted in greater numbers of larvae expelled relative to unoiled controls. Actual numbers were not reported, although the differences in numbers of larvae were noticed after 72 hrs exposure and appeared to diminish after 96 hrs of recovery in clean water.

Rinkevich and Loya (1977) exposed planulae of *S. pistillata* to WSF of Iranian crude oil (concentrations of mixtures reported as nominal 0.01, 0.1, 1, 5, and 10 ml/L, as previously described). They found that at the highest concentrations of 1, 5, and 10-ml/L settlement of planulae was significantly lower than in the controls.

Epstein et al. (2000) used the earlier experiments of Bak and Rinkevich, and Rinkevich and Loya, as a basis to examine the effects of both oil and dispersant exposure to the planulae of *Stylophora pistillata* and the soft coral *Hetertoxenia fuscescense*. For oil effects, they prepared a WSF mixture of Egyptian crude oil by shaking a 1:200 mix of oil:water overnight and used this as a stock solution for dilutions. Over a 96-hr exposure period, no *S. pistillata* mortalities were observed in any dilutions from 0.1-100 percent of the stock WSF mix. However, significantly fewer settlements of *S. pistillata* planulae occurred. There were no observed abnormalities in settled polyp morphology or in larvae swimming behavior.

Kushmaro et al. (1996) used the planulae of *Heteroxenia fuscescense* to study the effect of crude oil (unspecified) on metamorphosis. The crude oil was both floated on the water surface and used as a coating on the test vessels. Only nominal, not measured, exposure concentrations of oil were provided. In both exposure scenarios the nominal concentrations ranged as high as 5000 ppm.

With increasing concentration of oil on the water's surface, planulae increasingly lost the ability to undergo metamorphosis to polyps. At 10-ppm nominal crude oil, for example, only 50 percent metamorphosis occurred (compared to 97 percent in controls). The remaining planulae survived and appeared normal but did not metamorphose. Acute mortality of planulae increased at 500 ppm and above.

Oil coating on the experimental containers' surface also inhibited metamorphosis, at concentrations as low as 0.1 ppm. Planulae held in 0.1 ppm oil showed a 50 percent metamorphosis rate; at 100 ppm, only 3.3 percent metamorphosed. An increase in morphological deformities was also observed.

Kushmaro et al. felt that mortality was not a sensitive indicator of crude oil toxicity in *H. fuscescense*, and recommended consideration of sublethal effects at lower concentration ranges.

As previously described, Te (1991) exposed the planulae of *Pocillopora damicornis* to different nominal concentrations of gasoline:oil mixture and to benzene. Acute toxicity was noted only at the highest exposure concentration (100 ppm) in the closed system experiment. Clear correlations between concentration and settlement rate (corallite formation) were not evident with the gasoline:oil mixture. Te commented that the actual concentration per container may have varied significantly from one another—as we have naggingly noted, one of the primary problems with the use of nominal concentrations.

Results in the benzene exposures were similarly variable and difficult to interpret. In fact, Te suggested that the settling response in corals may not be suitable as a bioassay for exposure to petroleum hydrocarbons. He further recommended that future studies incorporate quantitative measurements of exposure, as opposed to the nominal or calculated equivalents.

In addition to the fertilization assays detailed previously, Negri and Heyward (2000) assessed the effects of crude oil and dispersant WAFs to *Acropora millepora* recruitment as reflected by inhibition of metamorphosis. In contrast to the findings of Te above, Negri and Heyward determined that settlement/metamorphosis to be a useful assay for exposure to oil. The results indicated that the larval metamorphosis assay was more sensitive to crude oil WAF than was the fertilization test. Crude oil significantly inhibited larval metamorphosis at 0.0824 ppm total hydrocarbon concentration, and completely inhibited it at 0.165 ppm. The dispersed oil mixtures (1 and 10 percent dispersant:oil used for stock WAF preparations) significantly inhibited metamorphosis at 0.225 ppm and 0.0325 ppm, respectively.

Histological Changes

Peters et al. (1981) performed a three-month laboratory exposure of the shallow-water Caribbean hermatypic coral species, *Manicina areolata*, to No. 2 fuel oil. Over the experimental period, none of the corals died. However, other physiological and histological changes were noted. At two, four, and six weeks into the study, corals showed increased mucus secretory cell activity. Both numbers and size of these cells increased until the eighth through twelfth weeks. At this point, the secretory cells atrophied and in some cases disappeared altogether. Other histological and cellular changes were observed as well, including loss of symbiotic zooxanthellae from the gastrodermis and mesenteries.

The authors suggested that because corals have a high lipid content, oil may partition into cells or membranes to disrupt vital bioenergetic processes and the symbiotic relationship between coral host and zooxanthellae.

Calcification and Growth (Extension) Effects

Researchers such as Birkeland et al. (1976) stated that in the absence of acute mortality effects in coral from oil, evaluating the rate of growth provides probably the best quantitative, objective measure that integrates a variety of physiological effects. To that end, they collected heads of *Porites furcata* in Caribbean Panama and performed a series of oil exposure experiments.

Birkeland et al. exposed heads of *P. furcata* to Bunker C oil for periods of 1 and 2.5 hrs, after which the colonies were placed back in the field for 61 days (controls were also established in which colonies were held in clean seawater with no oil added, for the same exposure time periods). As has been common for many of the earlier coral and oil toxicity studies, the actual exposure concentration was not measured. In this study, even the nominal concentration was not specified: only that 100 ml of Bunker C was added to buckets containing coral colonies and "just enough seawater to cover them." The resultant oil film on the surface of the water was noted at 2.4 mm in thickness.

Twenty-four hours following placement in the field, all colonies were examined and appeared healthy. After the two-month post-exposure periods, growth measurements were taken on the calcareous skeletons of control and treated corals .

Interestingly, the authors noted no visible qualitative differences between control and Bunker C exposed corals (i.e., no apparent damage), but the difference in growth increments was significant. Further, a significantly greater proportion of branches among the oiled corals failed to grow at all.

Neff and Anderson (1981) examined several sublethal endpoints with a range of oils and constituents (No. 2, South Louisiana crude, phenanthrene, naphthalene) on four species of scleractinian corals (*Madracis decatis*; *Oculina diffusa*; *Montastrea annularis*; *Favia fragum*) and one species of hydrocoral (*Millepora* sp.). Although only a few of the experiments included a flow-through exposure design, actual exposure concentrations were measured or estimated by IR spectrometry or conversion of radioactivity.

Among the investigations on corals, Neff and Anderson studied the effect of water-soluble fraction of No. 2 fuel oil on calcium deposition in *Oculina diffusa*. They found a high degree of variability in the depositional rates, but noted a trend of decrease in rates with decreasing WSF concentration. Overall, Neff and Anderson found that exposure to 0.45 and 0.87 ppm total hydrocarbons resulted in approximately 60 percent reduction in ^{45}Ca deposited by *O. diffusa* in 3 hrs.

They also performed experiments with five species of Bermuda reef corals and WSF of No. 2 fuel oil and South Louisiana crude oil. Results were quite variable. In some cases, (*Millepora* sp.), there was a trend (but not significantly so) toward increased ^{45}Ca deposition rates with increased concentration; *Favia fragum* was unaffected; *Madracis decatis* showed trends both toward both increased and decreased Ca deposition in different experiments, but significant results were obtained only in isolated instances. One experiment was performed with *M. decatis* and WSF of South Louisiana crude and the rate of calcium deposition was found to increase significantly with increasing WSF concentration. In *Monastrea annularis*, ^{45}Ca deposition increased slightly with exposure to No. 2 fuel oil, but only one significant result was obtained in the immediately post-exposure studies and with a 72-hr recovery period.

No. 2 fuel oil WSF significantly depressed calcium deposition in *Oculina diffusa* at the 2.6-ppm concentration but not at lower levels. Again, however, variability was high.

Exposure to phenanthrene resulted in both increased and decreased Ca uptake in *Millepora* sp., depending on concentration: lower levels (25 ppb) resulted in a great increase, while 100 and 500 ppb caused a slight decrease; variability was high.

Gardiner and Word (1997), and Gardiner et al. (1998) exposed colonies of *Acropora elsyii* to WAFs of a condensate and crude oil for 144 hrs, as detailed above under acute effects (refer to Table 2 to obtain approximate hydrocarbon concentrations for WAF mixtures of the two products). They obtained what can best be described as a variable set of growth (wet-weight biomass increase) responses under the different conditions of exposure: growth relative to controls was suppressed by one-third and two-thirds in the full-strength solutions of 150°C and 200°C weathered Campbell condensate, and "variable" among the diluted concentrations of the weathered condensate WAFs (recall that the full-strength fresh condensate WAF was acutely toxic to the corals). Growth was *enhanced* in the 10 percent WAF of the 200°C weathered Stag crude oil; *inhibited* in the 100 percent WAF from fresh Stag and 10 percent WAF of 200°C weathered Stag; and *unaffected* in the 50 or 100 percent WAFs of the 200°C Stag.

Dodge et al. (1984) attempted to simulate spill-like conditions by exposing colonies of *Diploria strigosa* collected in Bermuda to various concentrations (1-50 ppm, measured in water by fluorescence spectrometry) of Arabian Light crude oil for 6-24 hr periods in four laboratory and two field experiments. Following the exposure, the corals were moved to/left in place in the field for approximately one year. At that time, extension (growth) of skeletons was measured. This experimental design was intended to assess the long-term effects of a brief low-level exposure, as would be expected during an oil spill.

Dodge et al. found no effects of the oil exposure to *D. strigosa*. However, this should be tempered with the observation that substantial natural variability in growth parameters existed for the corals, which compromised the ability to detect small changes.

As a component of post-spill effects monitoring, Guzmán et al. (1991) and Guzmán et al. (1994) reported the results of coral growth studies ("sclerochronological analysis") after the Bahía Las Minas spill in Panama in four native coral species. Initially, Guzmán et al. (1991) found reductions in growth for three of the species (*P. astreoides*, *D. strigosa*, *and M. annularis*) and no effect in one (*S. siderea*). The lowest annual mean growth rates were measured for 1986, the year of the spill. Five years later, Guzmán et al. (1994) examined growth rates in *S. siderea* and *P. astreoides*. Growth rates for both were lower during the three years after the spill than before. At heavily oiled reefs, growth after the spill declined significantly for *S. siderea* but not for *P. astreoides*.

A more general study of coral growth by Lough and Barnes (1997) provides excellent background on the use of skeletal extension and density banding as a means to document environmental changes over time. They discuss both the advantages of using massive corals in this way as well as the drawbacks and limitations. Their review and research make it clear that oil spills are but one kind of environmental change that can affect coral growth. In studying massive *Porites* colonies (the skeletal cores could not be identified to species) on the Great Barrier Reef, Lough and Barnes found a particular sensitivity to sea surface temperature, with higher temperatures resulting in higher calcification and lower temperatures in lower calcification. However, they cite other research demonstrating growth impacts from bleaching, sewage, offshore drilling, dredging, eutrophication, and oil spills. They noted that because the links between environmental factors and density banding have been mostly through correlation rather than mechanistic studies of how environmental conditions affect growth, those links have been uncertain and "fuzzy."

Among oil spill researchers, Birkeland et al. cautioned that differences in spatial location of colonies, and time of year, each affected growth rate to a greater extent than did the exposure to oil. Neff and Anderson, despite concluding that ^{45}Ca deposition was a sensitive indicator of contaminant-induced stress in corals, noted that the high variability they experienced during the course of their experiments was a problem. However, they suggested that increasing the sample size and standardizing other experimental parameters would partially compensate for this and permit the use of the parameter.

Given the fact that several of the researchers using coral growth as an indicator of effect commented on interpretive complications and limitations due to natural variability and the influence of other environmental stressors and influences independent of oil, this may not be the best standalone endpoint for evaluation of spill impacts on a coral community. However, because growth integrates several parameters related to the overall health of a colony, it would seem reasonable to include it as part of a suite of endpoints in a multivariate assessment.

Surface Cover

Changes in coral cover over time represent another way to assess longer-term conditions in a reef. Similar to the use of coral growth results to infer oil spill impacts, relying exclusively on coral cover may not provide spill-specific insights. However, it should reflect an integrated response to environmental conditions that can include an oil spill.

Guzmán et al. (1991) compared cover of common coral species (noted elsewhere) at six reefs before (1985) and after (3 months post) the oil spill at Bahía Las Minas. At one heavily oiled reef, total coral cover decreased by 76 percent in the 0.5-3 m depth range and by 56 percent in the >3-6 m range. The decrease in cover was less at moderately oiled reefs and either increased or did not change at the unoiled reference reefs. The branching species *Acropora palmata* nearly disappeared at the heavily oiled site, but increased by 38 percent at the unoiled reefs.

In this same survey, Guzmán et al. also documented changes in average size of colonies and diversity based on cover. They found that colony size and diversity decreased significantly with increased oiling.

The TROPICS experiment (Ballou et al. 1987; Dodge et al. 1995) in Panama used changes in coral cover as one measure in assessing short- and long-term effects of oil and dispersant use in the three tropical habitats of mangroves, seagrass, and coral reefs. A slight, but statistically significant, decrease in coral colony cover was found over the period immediately following the oil exposure through 20 months, but when the site was revisited ten years later the difference in this measure relative to unoiled control had disappeared.

Photosynthesis

The symbiotic relationship between reef-building corals and their associated dinoflagellate algae (zooxanthellae) is well known, and the expulsion of the symbionts under stressful environmental conditions (bleaching) has been the source of much recent concern. As summarized by Benson and Muscatine (1974), the presence of zooxanthellae is not insignificant, either physically or functionally: in the coral *Pocillopora damicornis*, the algae constitute 45-60 percent of protein biomass of the colony; and, they provide the animal with photosynthetic products such as glycerol, alanine, and glucose. In turn, the coral supplies the algae with ammonia and protein.

Given the importance of the symbiotic relationship to the health of corals and the reef community, the effect of oil on photosynthesis represents a key series of endpoints for consideration. Cook and Knap (1983) studied the effect of Arabian Light crude oil and the dispersant Corexit 9527 both individually and in concert on the brain coral, *Diploria strigosa*. In this case, fluorescence spectrometry measured exposure concentrations in the water; average concentrations were 18-20 ppm and the dosing curve was similar to that illustrated in Figure 2. These, according to the authors, represented realistic upper limits to concentrations that might be encountered on a reef during a real spill in which dispersants were applied. Photosynthesis (carbon fixation) was studied using radioactive carbon.

Exposure to Arabian crude oil alone did not affect carbon fixation, expressed as total carbon fixed, in the distribution of labeled carbon in different chemical fractions of coral tissue, or in the distribution of photosynthetic carbon into lipid classes. Although some short-term (1-3 hrs after dosing) significant impacts to photosynthesis were noted with oil + dispersant, the reader is referred to the Cook and Knap article for detailed discussion.

Cook and Knap concluded that their results suggest that, without the application of dispersants, oil pollutants in realistic environmental concentrations had little effect on coral photosynthesis.

Cook and Knap compared their results to those of Neff and Anderson (1981) and commented that experimental approaches differed in important ways, such as longer dosing periods in static systems in the latter study, different oils and concentrations, and different methods of tissue digestion. Despite the methodological differences, Cook and Knap concluded that both studies found little substantive impact to coral photosynthesis from oil exposures at "realistic environmental concentrations." However, Neff and Anderson did determine that exposure of fire coral (*Millepora* sp.) to the water-soluble fraction of No. 2 fuel oil for 72 hrs resulted in a highly significant reduction in the rate of photosynthetic carbon fixation by zooxanthellae. They found a linear relationship increasing WSF concentrations and decreasing ^{14}C uptake rate, with exposure to 2.6 ppm total hydrocarbons resulting in an approximately 50-percent decline. If corals were permitted to recover for 72 hrs, the effect appeared to be transient and no differences could be detected between control and exposed specimens.

Interestingly, qualitatively different results were found for the species *Madracis decatis*. That is, after 72-hrs exposure to WSF of No. 2 fuel oil, no difference was found in the rates of carbon fixation. However, after another 72 hrs of post-exposure recovery, there was a significant and inverse relationship between previous exposure concentration and ^{14}C fixation rate: previous exposure to 2.6 ppm total hydrocarbon produced a 60-percent decrease in carbon fixation rate.

In 24 hr exposure tests with No. 2 fuel oil and South Louisiana crude oil, Neff and Anderson found no effect in the zooxanthellae of *Favia fragum* and *Montastrea annularis*. Inhibition of carbon fixation was found when *Millepora* sp. was exposed for 24 hrs. to phenanthrene, with a maximum decrease of 54 percent correlated with an exposure of 500 ppb.

Despite the findings of some impact to photosynthetic carbon fixation by coral zooxanthellae from exposure to oil, Neff and Anderson concluded that relative to oil effects on free-living marine algae, the coral zooxanthellae were "not greatly affected" by sublethal exposure.

Gardiner and Word (1997) determined that exposure of *Acropora elsyii* to two of the three full-strength WAFs of fresh and weathered Campbell condensate and all three WAFs of Stag crude oil resulted in substantial reductions in levels of chlorophyll a within coral tissues. However, only one of the dilutions (50-percent concentration of fresh Campbell condensate) caused a reduction.

Rinkevich and Loya (1983) also studied the effect of crude oil (Iranian) on photosynthesis of the branching coral, *Stylophora pistillata*. For this series of experiments they prepared a stock water-soluble fraction mix of the crude oil by mixing 300 ml of the Iranian crude with 700 ml of filtered seawater. This was stirred for 24 hrs, allowed to stand for 30 minutes, after which the water was drained and designated as 100 percent water-soluble fraction. Dilutions of 2.5-20.0 ml WSF/L were prepared and used. Corals were incubated with $Na_2{}^{14}CO_3$ in 3 m of water in the Red Sea.

No mortality was observed to corals or zooxanthellae. However, photosynthesis was affected: a *decrease* in higher (12.0 ml/L) WSF concentrations, and an *increase* at lower concentrations. Rinkevich and Loya interpreted the latter phenomenon (increase in photosynthesis at low-level exposures to WSF of crude oil) as a stress reaction called hormesis. The authors also suggested that these results reflect the inadequacy of traditional dose-response models due to the "toggling" of stimulation and inhibition with dose.

<u>Mucus and Lipids</u>

We might think of mucus as a physiological by-product rather than an important constituent in reef ecosystem energetics, but several researchers have studied the role of coral mucus and the impacts oil exposure might have on production and energy transfer in the reef. There is, in fact, a surprising amount of information on coral mucus and oil effects in the literature.

Benson and Muscatine (1974) researched the basic chemistry of coral mucus and also studied feeding behavior of reef fishes on the material. They found that coral mucus was rich in wax esters and triglycerides. In addition to observing fish feeding directly and extensively on mucus, they also found that artificially dispensing coral mucus resulted in an aggregation and feeding by fish. That is, reef fish *like* coral mucus. Benson and Muscatine suggested that ingestion of coral mucus by reef fishes was a likely route for the energy-rich products of coral metabolism to be transferred into the larger reef system.

Mitchell and Chet (1975) exposed healthy corals heads of the genus *Platigyra,* collected in the Red Sea, to crude oil and other chemicals. Unfortunately, little specific information is provided about the specifics of crude oil type and the manner in which the coral was exposed to oil. With this qualification in mind, they observed that addition of crude oil caused a dramatic increase in the production of mucus: 100 ppm crude oil increased production from 25 to 500 µg/10 ml over the 24 hr. test period, while 1000 ppm caused an increase in mucus production from 30 µg to 600 µg in the first day. In the latter exposure, mucus production subsequently declined to 100 µg after 4 days and remained at that level until the corals died on day 6. Exposure to other chemicals suggested that increase in mucus production was a generalized reaction to pollution stress.

Harrison et al. (1990) exposed a staghorn coral, *Acropora formosa,* collected from the Great Barrier Reef, to 5 and 10 ppm water-accommodated fraction of marine fuel oil (among other stress treatments, including chemicallydispersed oil and dispersants alone). Exposure concentrations were measured gravimetrically (i.e., water samples were filtered, extracted three times in dichloromethane, evaporated, and residues were weighed). In both oil treatments, response of the coral was similar: "massive" amounts of mucus were immediately discharged during the first hour of treatment.

Branches were able to withstand a 6-hr exposure, but some died after a 12-hr exposure. After 48 hrs, the authors noted a marked increase in the concentration of pigmented bacteria on the mucus, which they attributed to increased amounts of mucus and possibly to bacterial utilization of oil components.

Cook and Knap (1983) summarized the role of symbiont photosynthesis in corals and in reef ecology by noting that the primary production by zooxanthellae probably provides most of a coral's energy requirements. They linked primary production, mucus, and energy dynamics in the larger reef community by the same route described by Benson and Muscatine above.

Burns and Knap (1989) found indications that corals heavily stressed by oil had altered protein to lipid ratios and they discussed the adverse implications of impacts to lipid metabolism. As an example, they noted that a large portion of the energy fixed in algae/coral photosynthesis was channeled into mucus production. Mucus, with its high-energy lipid-rich content, was acknowledged to be a key component in reef food web bioenergetics, and Burns and Knap suggested that impairment of lipid metabolism in corals induced by oil exposure could easily cascade into the larger reef ecosystem.

Neff and Anderson (1981) made a qualitative observation of increased mucus production in *Montastrea annularis* with exposure to South Louisiana crude oil. Although they did not quantify the amount of production, they did use UV spectrophotometry to target naphthalenes in mucus. Naphthalenes were detected. Similarly, after exposure to ^{14}C-naphthalene, Neff and Anderson detected ^{14}C in mucus produced by *M. annularis.* They suggested that the results indicate that:

- Coral mucus can bind or adsorb aromatic hydrocarbons;
- Surface mucus may protect coral tissues from aqueous hydrocarbons;
- Mucus production may be a mechanism of hydrocarbon clearance from contaminated corals.

To explore the idea that coral mucus may act to transfer both energy as well as hydrocarbon exposure to the larger reef ecosystem, Neff and Anderson also studied the reef-dwelling butterfly fish (*Chaetodon* sp.). The fish was observed to actively ingest coral mucus. Corals exposed to WSF of South Louisiana crude for 24 hrs were placed in aquaria with butterfly fish, which fed on the mucus produced by the corals for several hours. Although low concentrations were found in several organs of the fish and in coral tissues, relatively high levels were found in gall bladder (5.77 ppm wet weight), head (4.51 ppm), heart (2.47 ppm), and brain (1.62 ppm) tissues.

These studies indicate that the relationship between coral mucus and oil exposure is a complex one. Researchers have shown that mucus production varies with oil exposure and that it may be a vehicle for removal from coral colonies. However, the high lipid content of mucus and its role as a food source to other organisms would suggest that in the event of oil exposure, mucus would also be a ready pathway to transfer not only energy, but also contamination.

OTHER EFFECTS OF OIL: BIOACCUMULATION

Oil quickly and readily bioaccumulates in coral tissues and is slow to depurate. This may be linked to the high lipid content of the tissues. Uptake into the symbiotic zooxanthellae also occurs. Researchers have found that petroleum hydrocarbons are deposited into the calcareous (aragonite) exoskeleton of corals, which introduces the possibility of using coral skeletons as historical records of hydrocarbon contamination in an area.

Bioaccumulation—defined as the concentration of a chemical in an organism through uptake from water or ingested food—has been studied in the context of oil and corals by many researchers.

We alluded to the lipid constituent in coral mucus above. Harriott (1993) reviewed studies of the role of lipids in coral histology and physiology and noted that lipid deposits were likely to be important energy reserves for the animals. The known affinity of petroleum hydrocarbons for lipid-rich tissues would imply the likelihood of bioaccumulation occurring when corals are exposed to oil.

Harriott had hoped to utilize the lipid content of corals from the Great Barrier Reef in an index for general condition (she had noted that lipid decline had been previously used to monitor changing condition of marine organisms, including corals). However, she commented that the method required a considerable refinement to achieve consistent results; and even when this took place, there was considerable variation among samples from the same colony. In addition, the efficacy of lipid extraction varied between species (*Pocillopora damicornis* and *Acropora formosa)*. Harriott concluded that the approach was problematic, not necessarily from a conceptual perspective, but rather from an applied standpoint.

In the chronic laboratory exposure of *M. areolata* by Peters et al. (1981) discussed above, the No. 2 oil was mixed with seawater with targeted water-accommodated fraction (WAF) of hydrocarbons of 0.1 ppm and 0.5 ppm. Measured concentrations for the two targets were 0.07 ± 0.04 ppm and 0.15 ± 0.10 ppm, respectively.

Chemical analysis by GLC/FID showed probable increases in coral tissue hydrocarbons from exposure to the WAFs after two weeks in the higher exposure concentration and six weeks in the lower. After one to two weeks of elimination* time in clean flowing seawater, all corals retained oil-associated hydrocarbons in their tissues. The lighter fractions, however, appeared to have volatilized or solubilized.

In addition to studying the lethal and sublethal effects of Iranian crude oil to the octocoral *Heteroxenia fuscescens,* Cohen et al. (1977) also evaluated uptake of petroleum hydrocarbons by the test colonies. While they demonstrated through examination of gas chromatograms that the corals were incorporating petroleum-derived hydrocarbons into tissues, the actual amount of uptake was not specified; it was noted as being much lower than "normal hydrocarbon background."

Knap et al. (1982) used a radioactive form of the polynuclear aromatic hydrocarbon (PAH) phenanthrene ([9—^{14}C] phenanthrene) to document uptake and elimination in the coral *Diploria strigosa* collected from the northern fringing reefs of Bermuda. Knap et al. used an exposure concentration equivalent to 33 ppb and found that, after a one-day exposure, about 17 percent of the radioactivity in the water were incorporated into the coral. The corals were then allowed to eliminate the radioactive phenanthrene in clean seawater. A sharp decrease in radioactivity was observed after the first two days of elimination, which was followed by a much slower rate through the rest of the 14-day experiment. At day 10 of elimination about 25 percent of the initially accumulated radioactivity was still present.

Other researchers have observed this kind of uptake and elimination pattern in other species of coral. Kennedy et al. (1992) tracked the fate of the PAH [^{3}H]benzo[a]pyrene (BaP) in two scleractinian coral species, *Favia fragum* and *Montastrea annularis,* collected from Biscayne National Park, Florida. They confirmed that corals accumulated BaP in their soft tissues, and were also able to show that the symbiotic zooxanthellae sequestered it as well.

They found that both *F. fragrum* and *M. annularis* readily incorporated the radioactive BaP from water, with a strong linear relationship shown between uptake rate and concentration of BaP. Normalized to skeletal surface area, *M. annularis* showed an uptake rate that was roughly twice that for *F. fragrum.* Similar to the findings of Knap et al. (1982), elimination of BaP from both species was slow, with a more rapid rate in the first 50 hrs than the last 94 hrs. At the end of the 144-hr experiment, *F. fragrum* had eliminated over half of its accumulated BaP, but *M. annularis* still retained about 60 percent.

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* We follow the suggested practice of Meador et al. (1995), in which the word depuration is reserved for passive processes of contaminant reduction in an organism, such as diffusion; and use elimination for the combined processes of metabolism, excretion, and diffusive loss of contaminants.

BaP-derived radioactivity was found in both the coral animals and the zooxanthellae fractions. The proportion in coral tissue vs. zooxanthellae fluctuated considerably with time, and differences were also noted between species.

Kennedy et al. found a relatively low proportion of the radioactivity in both coral tissue and zooxanthellae in an aqueous-soluble phase, which they say suggested only a limited ability to biotransform and detoxify BaP. Although this might reduce the toxic effects attributable to reactive BaP metabolites, it also would suggest a prolonged exposure to the parent compound.

In field studies stemming from the 1986 Bahía Las Minas oil spill in Panama, Burns and Knap (1989) confirmed laboratory observations of coral bioaccumulation in the two species *Siderastrea siderea* (a massive species) and *Agaricia tenuifolia* (thin, plate-like colonies).

Based on the residue patterns from the gas chromatograph, the corals appeared to take up hydrocarbons from the water column, as opposed to sediments. Coral mortality data, as shown by decrease in coral cover, correlated with hydrocarbon concentrations from surviving coral tissues.

Neff and Anderson reported less conclusive evidence of hydrocarbon (naphthalene) uptake in three species of corals from Florida (*Montastrea annularis, Acropora cervicornis*, and *Acropora palmata)*. They exposed colonies to a surface slick of South Louisiana crude oil for up to three days. There was no direct contact, only exposure to dispersed and soluble fractions of the crude at naphthalene concentrations that ranged between 0.006 and 0.29 ppm in a flow-through exposure setup. Neff and Anderson did not find a significant accumulation of naphthalene in the exposed corals, and no aromatic hydrocarbons were found by gas chromatograph at or exceeding 0.1 ppm. Oil-exposed corals did contain higher concentrations of total hydrocarbons, with the difference attributed by Neff and Anderson to increased accumulation and production of paraffin-type compounds in oil-exposed colonies. They suggested that oil-induced stimulation of wax-rich mucus may have been a contributing factor in this increase.

In another experiment by Neff and Anderson, *Oculina diffusa* from Texas was exposed to a 10-percent solution of South Louisiana crude oil spiked with ^{14}C-naphthalene estimated to provide an exposure concentration of naphthalene of about 42 ppb. Although initial concentrations of naphthalene in the exposure medium declined rapidly (50 percent loss of the spiked material in 3 hrs, 74 percent loss after 7 hrs), uptake of the naphthalene into coral tissues was rapid (Figure 3). The maximum radioactivity was estimated to equal about 0.27 ppm naphthalene/protein N. An elimination curve indicated that accumulated naphthalene would be cleared in 14 days; the half-time ($t_{1/2}$) of naphthalene elimination was about 24 hrs.

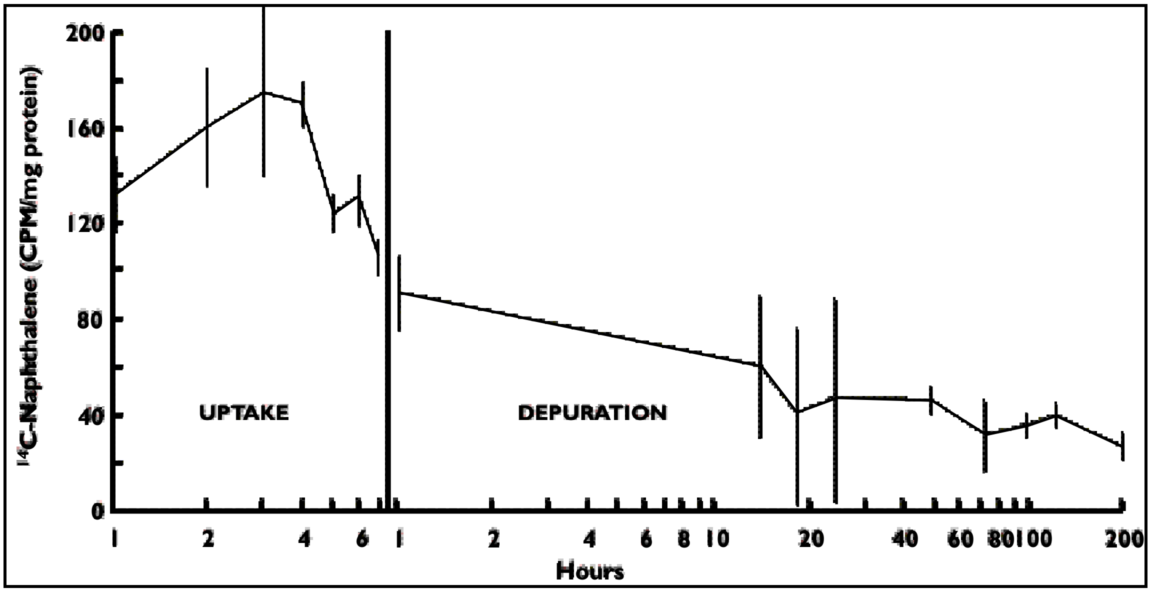

Figure 3. Accumulation (during 7-hr exposure to spiked WSF of South Louisiana crude oil) and elimination (following return to clean seawater) of ^{14}C-naphthalene by *Oculina diffusa*. From Neff and Anderson (1981).

Neff and Anderson (1981) also studied the differences in ^{14}C-naphthalene uptake in *Madracis decatis* under light or dark conditions. The coral accumulated significantly more naphthalene in the light than in dark, although they noted a high level of variability especially under light conditions.

Although we typically think of bioaccumulation as occurring in soft tissues, it can also include the bony or skeletal portions of an organism as well. Readman et al. (1996) studied corals in the northwestern Arabian Gulf (Kuwait and Saudi Arabia) after the 1991 Gulf War and found that the coral *Porites lutea* incorporated oil into the calcareous exoskeleton. The coral materials were sectioned and dated using microscopic and x-ray inspection. Chemical analysis for aliphatic hydrocarbons was accomplished by GC-FID, while the aromatic hydrocarbons were analyzed by GC/MS. Combining the dating of coral layers with the detailed chemistry within layers, Readman et al. were able to confirm that the colonies did incorporate hydrocarbons that fingerprinted back to the oil types known to have spilled during the Gulf War. Although they state that the occlusion process, as they termed the incorporation of hydrocarbons into the aragonite exoskeleton, is not fully understood, the elevated concentrations found within the skeleton likely reflect enhanced exposure at the time of deposition. However, they noted that selective degradation processes also seemed to occur, and that further work was necessary to confirm whether occlusion is proportional to exposure. This would determine the usefulness of corals as historical recorders of organic markers.

Guzmán and Jarvis (1996) conducted a similar study to that of Readman et al. above, but used the presence of the element vanadium in coral (*Siderastrea siderea)* skeletons as a proxy for oil exposure (vanadium is found as a trace compound in crude oils). Guzmán and Jarvis used the skeletal incorporation of vanadium as a potential indicator of chronic oil pollution attributable to the operation of an oil refinery in Panama, and found a good correlation between increase in concentration and the beginning of refining activities. They suggest that vanadium could also be used to document the longer-term history of regional oil contamination.

In addition to actually incorporating compounds to which they are exposed into their skeletons, corals record physical evidence of injury and recovery into their growth rings. Ruesink (1997) used these "scars" and "sclerochronology" (defined as the study of scleractinian coral growth rings), in the skeletons of two scleractinian species, *Siderastrea siderea* and *Porites astreoides,* to evaluate susceptibility to injury and ability to recover from it. The colonies had been collected to assess the effects of the Bahía Las Minas spill, but the results of Ruesink's study were not directly applicable to impact assessment from that particular incident. In fact, although Ruesink concluded, based on her sclerochronology study, that *P. astreoides* sustained less injury to internal regions of the colony and recovered more quickly than *S. siderea,* she was able to discern no obvious trends in injury rates over the preceding several decades, even at the site of several oil spills. Ruesink noted that the significant drawbacks to sclerochronology as a means to document disturbance in the coral reef system derive from what is not preserved in the skeletons: i.e., small, rapidly recovering injuries, and total colony death. This, unfortunately, may preclude two of the major effects observed as oil spill effects in corals and thus probably limits the utility of the approach in a disturbance assessment context.

INDIRECT IMPACTS

The indirect impacts of exposure to oil or to oil spill activities can take many different forms, only a few of which we will detail here. It should be clear by now that oil directly affects corals in a number of ways, but it can also initiate a sequence of events that ultimately can result in an endpoint of damage to the coral reef. We can find indications of this from laboratory work as well as field observations.

As detailed previously, Mitchell and Chet (1975) documented that exposure to crude oil caused corals to increase their production of mucus. They also studied the role of bacteria in determining the severity of impact from the production of the excess mucus. Interestingly, they found that coral mortality occurred only in the presence of bacteria and, when microbial growth was inhibited or prevented through the addition of antibiotics, the corals survived exposure to the chemical contaminants. Mitchell and Chet suggested three different microbial mechanisms for death of the coral:

1. Predatory bacteria were initially attracted to the excreted mucus, which then attacked the coral tissue;
2. Bacteria of the genus *Desulfovibrio,* attracted to reduced redox conditions produced by the growth of other bacteria, produce hydrogen sulfide, which possibly killed the corals;
3. Infected corals were visibly covered with a filamentous bacterium called *Beggiatoa,* a heterotrophic genus that may have fed on the coral tissue itself.

Mitchell and Chet concluded that pollutant concentrations insufficient to kill corals directly can cause death indirectly by stimulating adverse microbial processes.

Harrison et al. (1990) observed a similar "bloom" of bacteria on the increased mucus production from *Acropora formosa* corals exposed to WAF of marine fuel oil. Although they cautioned that the extent to which the bacteria caused the destruction of coral tissue was not known, they felt it likely that the increased microbial activity contributed to the collapse of the tissue. However, they also postulated that this bacterial action may have been an artifact of the experimental setup. That is, in normal reef environments the mucus and bacteria would be carried off the coral or ingested by reef dwellers; in a tank with static or insufficient flow-through conditions, the bacterial population could rapidly rise above normal levels and result in additional coral stress.

Kinsey (1973) studied another phenomenon associated with oil spills that could conceivably affect corals in an indirect way, the effect that crude oil slicks might have on gaseous exchange over a reef. The author asserted that—at least in some reefs, such as the Great Barrier—corals have an oxygen requirement so critical and sensitive to change that extreme tides in sheltered reefs can sometimes result in completely anoxic conditions with resultant mortalities to much of the community.

Kinsey used a Moonie crude oil to create slicks with approximate thicknesses of 0.1 and 0.7 mm. He found that while the oil caused a noticeable calming of the water surface, there was no interference with normal gas exchange (specifically, oxygen and carbon dioxide) between the water and atmosphere, and hence, no adverse impact.

Recent work has elevated the level of concern related to photoreactivity of petroleum hydrocarbons to greater prominence. Although at this writing, no research specific to corals has taken place, results to date suggest an area for future work and the potential for significant adverse impact during oil spills. Pelletier et al. (2000), for example, found that maternal transfer of the PAH fluoranthene (a common component of petroleum products) from adult bivalves to pelagic larvae occurred and then rendered the larvae susceptible to photoxicity effects when they were in the water column. The demonstrated ability of corals to accumulate PAHs and, perhaps more importantly, the strength and duration of sunlight exposure in the clear, shallow waters typifying coral reef environments should flag possible photoreactive toxicity effects as a concern during spills.

Finally, there is an indirect consideration that has received little attention in the coral and oil literature but is well known to spill responders. A sensitive resource like a coral reef can be damaged by the response and cleanup itself. Ray (1981) noted this in closing his review of oil spill impacts, and singled out resuspension of sediments by propellers and mechanical damage from boats, anchors, and personnel as potential sources of damage that could take years to recover. In many cases, cleanup-related impacts can be minimized or reduced with awareness and planning, but rarely are they completely eliminated.

VARIABILITY IN IMPACT DUE TO REPRODUCTIVE STRATEGY

The vulnerability of gametes and early life stages of coral to oil suggests that there is a critical link between timing of a spill and the nature of coral reproduction to determine impact. In many, if not most, organisms, the earlier developmental stages often do not have fully functional detoxification mechanisms and are less capable of dealing with toxic exposure than are more mature forms of the same organism. Many of the studies previously reviewed documented effects of oil on coral fecundity or planulae. As pointed out by Guzmán and Holst (1993), the gametes of most spawning marine species tend to rise to the surface of the water after spawning. Negri and Heyward (2000) noted that coral oocytes are rich in hydrocarbons and highly buoyant. In mass spawnings such as those typical of the Great Barrier Reef, these oocytes can form their own slicks in calm weather and there is a high probability of coral gametes and petroleum products coinciding in the water column. Moreover, the larval stages of coral spend one to several weeks as plankton before developing to the point of settling competently. Therefore, the timing of an oil spill relative to the reproductive cycles of coral species of concern could be expected to play a key role in determining the impact to the population. Presumably due to their inherent tendency to associate with the uppermost layers of the water column, free-floating larvae stand a greater chance of contacting elevated concentrations of oil than would larvae that are not dispersed from the colonies. Empirical information (Pain 1994) from Bahía Las Minas reflected this, as those coral species with dispersed planktonic larvae were found to be much less robust than brooding species.

Peters et al. (1997) concurred with this idea. They noted that spills occurring near or at peak reproductive season (e.g., late August in the Caribbean and Gulf of Mexico, April in the Great Barrier Reef region) could effectively eliminate an entire year of reproductive effort while continuing to reduce fecundity through partial mortality and impairment of gonadal development.

In an excellent overview of coral reproduction, Richmond (1987) related how corals can be hermaphroditic (simultaneous or sequential, and in the latter, protandrous or protogynous), gonochoric (dioecious), or sterile. Species may release brooded planulae, spawn gametes, or reproduce solely by asexual means. It is, however, difficult to generalize about mode of reproduction, even within the same species: Richmond noted that *Acroporis humilis* was reported to brood planulae at Eniwetok, but was found to spawn gametes on the Great Barrier Reef and in the Red Sea.

In light of these considerations, information on reproduction and recruitment in various coral species would be a useful reference for spill responders and resource managers in anticipating potential impacts of a spill incident. Reviews of this kind of information are uncommon. Fortunately, at least two exist: Richmond (1987) summarized reproductive data for Caribbean, Pacific, and Red Sea species, providing information for 92 of what was at that time estimated to be 400 known scleractinian reef coral species; Richmond and Hunter (1990) updated and augmented the data to include reproductive information for 210 of (what was then estimated to be) about 600 identified corals.

Although data for only about a third of the identified scleractinian coral species are available, use Appendix C as a quick reference for a broad geographic region to ascertain whether pelagic coral reproductive activities are peaking at a given time. See the appendix for more detailed information on how to use and interpret the information.

VARIABILITY IN IMPACT WITH TIME OF YEAR AND REGION

The time of year in which a spill event occurs overlays on reproductive strategy to give one predictor of impact. Time of year, independent of reproductive considerations, is an important determinant of effect for other reasons, and these are variably intertwined with seasonal weather, geographic location, and other considerations.

Wyers et al. (1986) found an apparent seasonal difference in the nature of sublethal oil toxicity in *Diploria strigosa.* They exposed the coral to a measured, physically dispersed Arabian oil concentration of 18-23 ppm for a 24-hr period during both summer and during winter months. They found a qualitatively different kind of impact in the two seasons: in winter, tissue rupture following exposure was detected in 83 percent of colonies; in contrast, summertime results included only 22 percent with detectable tissue rupture. Wyers et al. termed these differences in effect "minimal" when considered over the entire 4-week experimental periods, but also suggested lower winter water temperature might have contributed to the effects differences.

Johannes et al. (1972) described a combination of seasonal and regional factors that may subject a coral reef to greater risk from oil spills . They mentioned coral reef differences among geographic regions that bear factoring into response strategies in the event of a spill. For example, they note that Indo-Pacific reefs include corals that protrude above the water surface on low tides more commonly than Atlantic reefs. This consideration would certainly influence the impact that an oil spill of a given size might have on a reef, as a tidally exposed coral could be expected to suffer more severe impacts from direct contact with the oil than a coral exposed only to the water-column fractions of that oil.

During the early days of the Gulf War oil spill in 1991, concerns were raised about potential effects to the marine environment of the region. In a background document that detailed threatened resources, the World Conservation Monitoring Centre (1991) noted that the Gulf is an inherently stressful environment for corals because it is the northerly range limit for tropical corals and experiences a wide fluctuation in temperature and salinity. LeGore et al. (1989) similarly described a seasonal bleaching event in many of the corals of the Arabian Gulf due to lower ambient water temperatures in the winter months and suggested an additional susceptibility to oil exposure because of it.

Jaap et al. (1989) studied the Bird Key Reef coral community in the Dry Tortugas off Florida and found a similar limitation in reef robustness because of the low temperatures occasionally encountered during the winter in those waters. Besides sometimes causing outright mortality due to cold water, the temperature stresses also were thought to alter reproductive patterns by inhibiting sexual reproduction. Thus, response-related assumptions about corals in such an area may not be valid if they are based on general life history characteristics for a given taxon.

Other scientists and managers have recognized that distinct differences in tropical marine coastal ecosystems among regions are important. UNESCO sponsored an entire workshop on the subject in 1986 (Birkeland 1987). In the proceedings from this workshop, Sammarco (1987) compared ecological processes on coral reefs of the Caribbean and the Great Barrier Reef and, in particular, compared coral recruitment patterns in the two areas. He then discussed these differences in the context of reef recovery from a major perturbation.

Sammarco focused on corals of a single genus, *Acropora,* which occurs in both the Caribbean and in the Great Barrier Reef. *Acropora* is an important and sometimes dominant genus in both environments. However, distinct differences exist—especially with respect to recruitment patterns. In the Caribbean, individuals representing *Acropora* are rare in the newly settled community of juvenile corals derived from planulae. On the Great Barrier Reef, in contrast, newly settled spat from *Acropora* can account for 50-80 percent of juvenile corals. Sammarco suggested that, in the latter case, most species of *Acropora* rely on reproduction via planular settlement, whereas in the Caribbean, *Acropora* is more heavily dependent on asexual reproduction via branch breakage and recementation.

According to Sammarco, the implication, when contemplating recovery of that genus from a major environmental disturbance, is that the process could be expected to take longer in the Caribbean than on the Great Barrier Reef. This suggests that the ability to extrapolate coral biology and disturbance responses across regions is limited and should be done cautiously. It seems this is a consistent message across many aspects of coral biology and impact assessment, as it becomes clear that few hard and fast generalities exist.

VARIABILITY IN IMPACT WITH SPECIES

That inter-species differences exist among corals in toxicity of oil or severity of spill impacts should not be particularly surprising, since these kinds of variations in effect have been shown to occur for countless permutations of organism and contaminant (see, for example, Mayer and Ellersieck 1986). Some of the studies cited for this report have alluded to potential species differences in the way oil affects corals. Both Johannes et al. (1972) and Guzmán et al (1991, 1994) suggested that branching coral species were more susceptible to oil than non-branching species. Johannes et al. linked the sensitivity of the branching species—and the increased tolerance of the other massive species—to the protective qualities of mucus. There may also be other more subtle factors that contribute to apparent species differences.

Table 3 showed that Bak and Elgershuizen (1976) used impairment in the ability of corals to clear sediment as an endpoint for sublethal oil toxicity. Research Planning, Inc. (1986), in a review of the impacts from a ferry grounding near Isla de Mona, Puerto Rico, suggested that increased sedimentation from the grounding could affect coral health. They referred to studies indicating that some species of corals are more sensitive to the effects of sedimentation and turbidity than others, and identified differences in local species' tolerances as follows:

High Sensitivity

Acropora palmata
Acropora annularis
Porites astreoides

Intermediate Sensitivity

Diplora labyrithiformes
Diplora strigosa
Montastrea annularis
Madracis mirabilis
Agaricia agaricites
Porites porites

Low Sensitivity

Mussa
Eusmilia
Montastrea cavernosa
Gorgonians

These rankings are independent of any species differences in tolerance to oil, so it is possible that a vessel grounding could result in negligible detrimental effects from oil, but cause more severe consequences from the production and distribution of particulate material in the water column and on the bottom. It also raises the possibility of synergistic amplification of adverse impact if oil toxicity combines with indirect effects of the grounding (other synergistic interactions are discussed in the follow section).

Table 4, previously discussed under Chronic Effects, shows the results of Lewis (1971). Despite the limitations of the method for calculating oil exposure, there appeared to be distinct inherent differences in the sublethal responses of four species of corals, possibly related to the morphology of the species. Research Planning, Inc. (1986) also made the link between observed species differences in sensitivity to sediments and physical form by suggesting that coral species that are cylindrical or upright are less susceptible since flat, palmate growth forms are more likely to accumulate sediments. In Lewis' study, however, there were differences in tolerances to oil even between species of similar form.

Neff and Anderson (1981) had subjected five species of corals to similar oil exposures and found substantial differences in the resulting toxicities among the species. They found that *Madracis decatis* and *Montastrea annularis* were severely stressed by exposure to No. 2 fuel oil WSF, as reflected by polyp retraction, expulsion of zooxanthellae, and marked increase in calcium deposition. In contrast, *Oculina diffusa* and *Millepora* sp. showed no impact in polyp and zooxanthellae endpoints and a slight decrease in calcium deposition. The authors did not speculate as to the reasons for these apparent species differences.

SYNERGISTIC IMPACTS

Those who have studied oil spill impacts or other disturbances in natural systems understand that it is often difficult, if not impossible, to separate the effects of one perturbation event from others. That is, an area in which we may wish to study the effects of and recovery from a spill incident may also have been subjected to one or more natural or human-induced impacts so that we cannot quantify the relative contribution of each to the resultant environmental conditions observed.

Dubinsky and Stambler (1996) summarized the range of anthropogenic stresses to which reef systems are subjected. Their review provides an appropriately broad framework for considering the impact of oil on corals, which is but one of many human disturbances to these systems. Moreover, anthropogenic impacts are a subset of the environmental changes (including shifts in oceanic conditions and climatic patterns) which ultimately determine coral reef ecosystem health.

Shinn (1989) was more direct in asserting that anthropogenic influences, while possibly of local significance, were much less significant than large-scale influences like sea level change, hurricanes, global increases in carbon dioxide levels, and changes in water temperature. However, he acknowledged the danger that human activities may exacerbate the threat to corals both locally and on a global scale. In the Arabian Gulf off the coast of Oman, Al-Jufaili et al. (1999) documented significant impacts to coral reefs from natural and human sources. Predation on corals by the starfish *Acanthaster planci,* storm damage, coral disease, and temperature stress were the most prevalent natural impacts, while fishing gear damage (primarily lost or abandoned gill nets) caused significant damage on many reefs.

Messiha-Hanna and Ormond (1982) documented significant deterioration of coral reefs in the Gulf of Suez area of the Red Sea. Although this reduction in coral cover was ostensibly attributed to erosion by sea urchins (principally *Diadema setosum)*, the authors further linked the high numbers of urchins to reduced numbers of their fish predators, which in turn was blamed partly on fishing pressure. In addition, however, Messiha-Hanna and Ormond suggested that the observed chronic oil pollution of the area was also responsible for reduction in coral cover. They commented that regrowth and reproduction would, under normal circumstances, mediate predatory reduction in cover. However, the distribution of coral reef erosion suggested that oil impaired those processes . Fishelson (1973) had similarly observed that injured coral reefs do recover and regenerate; however, a necessary condition for those to occur was the prevention/elimination of pollutant stresses.

In his review of global and local threats to coral reefs, Wilkinson (1999) noted that before 1998, direct and generally localized anthropogenic impacts (increased sediment loading, pollution, over-exploitation) were regarded as the major threat to coral reefs. However, in 1997-1998, the significance of truly global coral reef threats was identified when unprecedented coral bleaching occurred in pristine reef areas and modeling exercises showed that increased ambient CO_2 concentrations were expected to be increasingly problematic.

Loya (1975) illustrated the interpretive dilemma in a coral reef system, the northern Gulf of Eilat region of the Red Sea extensively studied by this researcher over the years. Loya monitored two reef flats over the four-year period 1969-1973—one in a nature reserve located near a refinery and several other human activities, the other a control area removed from those influences. Both areas supported rich and diverse coral reef communities. In 1970, an unexpectedly low tide exposed the reefs in both areas to very high ambient temperatures for extended periods of time. The direct consequence of these conditions was a mass mortality of about 90 percent of the scleractinian corals in both the nature reserve and the control area. In the subsequent years, distinctly different patterns of coral recolonization in the two areas occurred. In 1973, the reef near the nature reserve was gray, unattractive, and dominated by algal species. In contrast, the control reef had rebounded in recolonization of corals to the point where numbers of colonies exceeded those in 1969 before the low tide. While no significant differences in community structure (number of species, number of colonies, living coverage, diversity per transect) between the nature reserve and control reef were found in 1969, all parameters were significantly higher at the control in 1973.

Loya speculated as to the reasons for these distinct differences, and suggested that oil spills, chronic oil pollution, phosphate eutrophication of a lagoon in the reserve, thermal pollution from a desalinization plant, and coral breakage from tourist activities all could have had potential roles in inhibiting recolonization. Fishelson (1973) similarly attributed coral reef decline in the area to proximity of the oil terminal (responsible, according to Fishelson, for 2-3 spills per month) and a phosphate loading operation. Unfortunately, there was no way to determine the relative contribution of these and other influences to the overall impact. Ray (1981) further noted that, while effects on this reef were probably due to the high level of chronic oil pollution, it would be difficult to extrapolate the findings generated there to other locations for a given spill.

Loya's work foreshadowed current concerns about incremental, cumulative, detrimental changes occurring to coral reef systems worldwide. It is clear that many coral ecosystems continue to be subjected to a range of stresses, the most obvious being bleaching attributed to ambient temperature increases. The additional stress imposed by an event such as an oil spill might have a far greater impact on a weakened reef and reef community than the single and isolated insult of a spill would otherwise be expected to represent. There is a general recognition of the cumulative nature of environmental stresses in corals in oil spill contingency planning, as shown by statements such as this from Nansingh and Jurawan (1999) in an environmental sensitivity mapping exercise in Trinidad:

> Salybia Reef on the northeast coast of Trinidad exists at the limit of environmental tolerance of corals due to seasonal low salinity and high turbidity conditions. This reef would be adversely affected by the additional stress due to oil contamination.

A qualitatively different kind of synergistic spill effect consideration that should at least be mentioned here is one we have explicitly avoided detailing here in order to simplify the complex task of discussing oil impacts: the potential synergistic increase in impact with the combination of oil and chemical spill dispersants. The example we can cite here is the research of Cook and Knap (1983), who 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; but these are detailed elsewhere (see, for example, the direct discussion of this subject in Hoff 2001).

SUMMARY

While a number of research efforts have been undertaken over the years to elucidate the effects of oil on corals, extracting general lessons from the resulting literature is difficult. Many, if not most, of the exposure methods poorly document the concentrations of oil experienced by test corals. This deficiency precludes comparing results across studies, and does not even offer strong support for internally consistent exposures due to variability in how oil mixes into water.

There is also the question of how realistically some research exposure compare to spill exposures. That is, in the interest of obtaining a strong response "signal," some studies may have relied on oil exposure scenarios that could be imagined only in the most extreme situations (which is not to say, based on previous experiences, that they could not occur).

LABORATORY STUDIES

Over the last thirty years, many researchers have studied the effects of oil to corals in the laboratory. Unfortunately, a large portion of the results can be interpreted and extrapolated to oil spill scenarios only in a most general way, primarily due to the way corals were exposed or to the way the oil exposure was quantified. That is, our real-world expectation of oil exposure to reef corals would be in the form of water-accommodated fractions of spilled petroleum mixed into the water column, with the highest concentration encountered early in the spill and steadily (probably rapidly) declining concentrations over time. The available oil toxicity studies in corals rarely come close to simulating this kind of exposure scenario, which renders the application and extrapolation of results difficult and possibly inappropriate. This does not mean that the body of research not portraying realistic spill parameters is not useful, however— a great deal of information about the range of effects resulting from oil exposure can be gleaned.

Some noteworthy laboratory studies based their experimental design on oil exposures and exposure scenarios that could reasonably be expected during actual spill events. These focused on oil exposure through water-accommodated fraction at realistic levels that were measured and not merely calculated. The exposures were pulsed or flow-through to approach conditions during a spill incident and avoid complications from scale effects and lack of circulation. Although many have commented that laboratory results are difficult to extrapolate to field conditions, the studies that incorporated the above considerations into their design and implementation likely provide the most appropriately applicable results. The laboratory studies that made good efforts to satisfy these conditions include:

Cook and Knap (1983)
Dodge et al. (1984)
Gardiner and Word (1997)
Harrison et al. (1990)
Knap et al. (1983)
Neff and Anderson (1981)
Peters et al. (1981)
Negri and Heyward (2000)

Based on what we and others have identified as some of the weak and strong points from oil toxicity studies in coral, suggested laboratory parameters for future studies are listed below. To the extent possible, we would recommend using the experimental setup procedures and protocols of CROSERF, the working group of the Chemical Response to Oil Spills Ecological Effects Research Forum. These were developed through extensive testing and discussions among an international group of toxicologists, chemists, and physical scientists. These protocols are summarized in Singer et al. (2001) and Singer et al. (in press). Some suggested parameters:

- Water-accommodated fractions as mechanism of exposure;
- At least three different oils: No. 2, No. 6, and at least one crude;
- Realistic exposure concentrations, measured by fluorescence or other means, spanning a range of up to 50 ppm;

- Spiked exposure for comparison to constant concentration exposure and to simulate spill conditions;
- Multiple species, with selection based on factors of particular interest such as form (e.g., branching or massive), geography (e.g., Pacific/Oceania, and Caribbean/Atlantic), reproductive strategy (brooder or spawner), or mucus cover;
- Sensitive and multiple toxicity endpoints, such as molecular biomarkers, as well as calculation of traditional acute endpoints such as LC_{50}.

Of these, we believe that measurement of exposure concentrations is the most critical, as it represents the most direct way to link laboratory experiments to field or oil spill situations.

FIELD STUDIES

Although field studies ostensibly offer the best opportunity to study and understand the effects of oil spills under realistic conditions, they are uncommon in a coral reef setting. Rinkevich, Loya, and their colleagues have studied and written about a heavily impacted spill area in the Red Sea, but interpreting these results is complicated not only by the layering of spill incidents with chronic oil pollution from refinery and terminal operations, but also from a host of other human-induced stresses unrelated to oil. Perhaps the best example of a field spill study is that of the 1986 Bahía Las Minas spill in Panama, which had the good/bad fortune to have occurred near the Smithsonian Tropical Research Station. The extensive series of studies across that tropical system documented a number of short- and long-term impacts attributed to the oil spill. The effects of the Bahía Las Minas spill are discussed in (among others):

Burns and Knap (1989)
Guzmán et al. (1994)
Guzmán et al. (1991)
Guzmán and Holst (1993)
Guzmán and Jarvis (1996)
Jackson et al. (1989)
Keller and Jackson (1993)

The most carefully designed and monitored attempt to perform a manipulative, large-scale field experiment with corals and oil was the 1984 TROPICS effort, sponsored by API. Intended to provide a degree of realism within a relatively more controlled setting, TROPICS examined short- and long-term effects of oil and dispersed oil to three common tropical habitats. Interestingly, of the three habitats studied, the coral reefs were least affected by exposure to oil alone. References for TROPICS are:

Ballou et al. (1987)
Dodge et al. (1995)

ACUTE EFFECTS

Many early studies of acute oil toxicity effects in coral involved what can only be described as severe exposure conditions: submerging corals in marine diesel for 30 minutes (Birkeland et al. 1976);holding corals under static conditions in small (50- or 250-ml) containers with relatively large amounts (1-4 ml) of oil (Reimer 1975); allowing crude oil to coat portions of coral colonies exposed to the air (Johannes et al. 1972).

In the latter example, the extreme dosing had a basis in a potential spill scenario, as the authors noted that some corals may be exposed during periods of low tides in the Indo-Pacific region.

That any of the test corals survived these exposures for any length of time is interesting, and in some cases even astonishing. It is notable that sometimes a colony was not killed outright after a "dunking" in pure product, but subsequently showed a steady decline in condition over a long (>100 days) period, ending in death. This might lead us to question the definition of "acute." Another interpretive difficulty arises when conclusions of apparent survival from high concentration exposure are compared to those of Harrison et al. (1990), whose methods satisfied the criteria established above for realism in a laboratory setting: Harrison et al. found that low-level exposures (relative to the submersion in pure oil product that was sometimes carried out) nearly completely disintegrated coral tissues after 48 hrs. While they had selected a coral species known for its sensitivity to stress (*A. formosa)*, these results taken at face value suggest that a brief exposure (1-30 minutes) to extreme concentrations of oil is far less acutely toxic than a longer (4-48 hrs.) exposure to low concentrations. Differences in species' tolerances are probably important here, with branching corals among the most susceptible and massive corals more tolerant of oil exposure.

The old notion that coral reefs do not suffer acute toxicity effects from oil floating over them is probably incorrect. Certainly, direct coating increases the severity of impact, but the water-accommodated fraction at concentrations that could be encountered during a spill also appears capable of causing rapid mortality (in addition to longer-term effects).

CHRONIC EFFECTS

In contrast to the situation with investigations into acute toxicity of oil to corals, all researchers studying chronic effects documented sublethal changes in exposed corals in some form. The diversity of impacts described in the literature at least in part reflects an increasing degree of toxicological sophistication and technical ability to document effects at suborganismal levels. The literature also points to a realization that more subtle sublethal manifestations of oil exposure may be most important in understanding how a spill will affect a reef system.

Sublethal oil exposure appears to affect many normal biological functions. From the available literature, it would seem that the function with the most potential to adversely influence the survival dynamics of the corals themselves would be those associated with reproduction and recruitment. A host of studies have shown that oil reduces fecundity, decreases reproductive success, and inhibits proper development of early life stages of corals. A spill occurring at just the wrong time in a given area, at the peak of reproductive activity, could cause immediate and long lasting harm to the communities of corals themselves.

Oil also impairs two fundamental bioenergetic components for the entire coral reef community: primary production by the zooxanthellae symbionts in coral, and energy transfer via coral mucus. While some of the referenced studies indicate that effects to these processes are transient and that corals can recover from them in the absence of oil, circumstances of individual spills will dictate whether these would be of concern to responders and resource managers.

<u>IMPLICATIONS FOR SPILL RESPONSE AND PLANNING</u>

1. **Oil is toxic to corals.** Although Wilkinson (1999), in his review and prediction of trends in coral reefs worldwide suggested that "…it is unlikely that major oil spills near coral reefs will cause significant damage," most of the literature points to a potential for impact that cannot be ignored. We can argue about how toxic oil is to corals and how that toxicity is expressed, but it is clear that exposure to oil can adversely affect corals. The difficult part is the quantification of exposure and effect for the purposes of making (and justifying) the inevitable tradeoffs during a spill. This exercise involves no easy answers, and the available science offers only the most general guidance.

   If we acknowledge the potential for adverse impact, the logical question for responders to ask is, " What level of exposure represents a threshold for significant effects?"

   - Direct contact between a living coral and oil, whether it is intertidally or subtidally, is likely to result in serious pathologies or death of some portion of the colony.

   - Based on the admittedly limited literature in which exposure concentrations were measured, a reasonable effects threshold in the water column is 20 ppm. This is the concentration noted by Cook, Knap, Dodge, and their colleagues as a level where sublethal impacts could be elicited, as well as one representing a water concentration possible in a spill scenario. Even at this relatively high water concentration, most of the noted impacts were transient after a certain period of recovery in clean water.

- Transient concentrations of oil in the water below 20 ppm are probably not likely to result in lasting harm to a coral reef. The key word here, outside of "probably," is "transient." This implies that such a concentration would be sustained and experienced for only a short time. The situation where such a level is a chronic parameter, where there is a continuing source, may result in more serious pathological effects to exposed coral reef communities.

- It would be reasonable to expect that much lower concentrations of oil in water could harm larvae of coral or impair normal reproductive processes. The work of Negri and Heyward, which at least measured concentrations of the stock solution used in preparing dilutions, identified a threshold for fertilization inhibition at 0.165 ppm.

2. **Time of year is critical.** A spill of a given size of a given oil in a given area may have dramatically different impacts depending on the time of year in which it occurs. While this is true of any environment anywhere in the world, it is especially true for coral reefs, where reproductive and early life stages are known to be particularly sensitive to oil. In other words, if we know who is reproducing when, it takes us a long way toward determining whether the spill has an enhanced potential for injury to the corals. Appendix C is intended to help define periods of increased vulnerability.

   In many areas of the world, spring and summer are peak reproductive periods. This is reflected in Appendix C. This seasonal timing for spawning has been noted especially in coral reefs like the Great Barrier Reef, where the synchrony of timing becomes even more narrowly focused to specific lunar phases and more broadly defined to include a wide segment of the reef community beyond corals. In such a system, an ill-timed spill (which is not to imply there is such a thing as a well-timed spill) has the potential to wipe out a sizable portion of a given year's recruitment for many community members.

3. **Expert knowledge should be used.** Coral experts who have knowledge of the reefs of concern should play a key role in shaping a response strategy. For example, local biologists may be able to tell whether a threatened area is dominated by branching corals, thought to be sensitive to oil, or massive corals, thought to be more tolerant. They may know about life histories, a key consideration as suggested above. They should have information about the presence of threatened or especially sensitive reef community members.

Resource managers in particular will be able to provide useful and relevant information about portions of a reef community that may be experiencing greater stress from (for example) a recent grounding or hurricane damage. In this way, such experts can help to identify areas for increased protection or areas where such efforts may not be worth the effort.

4. **Cumulative impacts complicate things.** Everything that we know about oil and corals may be moot if the reef in question is under serious stress from other sources before the spill. Currently, concerns about coral reef deterioration are nearly universal. Warming of the oceans has been identified as a key factor in widespread coral bleaching events; physical destruction from recreational boating or fishing activity has been locally significant; human pollution and land use policies have contributed to degradation. It is possible that an oil spill in any area subjected to substantial but unrelated stress may represent a synergistic "tipping point," an impact that ordinarily would not cause significant damage, but that against the background of other stresses causes a rapid cascade of seemingly irreversible decline in the reef system.

New tools that could help us determine the pre-existing status of a coral reef and also isolate effects of oil spills may be imminent. For example, Downs et al. (2000) describe a molecular biomarker system capable of discriminating among different sources of stress to identify (and even predict) corals at risk from bleaching. Their approach assays specific parameters of coral cellular function to gauge degree of stress; each parameter was chosen because it reflects specific cellular physiological functions. If this approach is validated and can be readily adapted to the field, it could provide an excellent way to ascertain status of a coral community and evaluate relative sources of overall stress in an impacted area.

Even with the promise of new technologies, spill responders will continue to labor under the burden of incomplete and not entirely relevant information. Despite any common ground that may exist from previous incidents, each spill represents a new and unique set of circumstances. The studies we have summarized here provide a background of research against which we can evaluate alternatives and extrapolate effects. There will, however, always be some degree of judgment call associated with the exercise.

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APPENDIX A

GLOSSARY/ACRONYMS

API: American Petroleum Institute

arborescent: like a tree in growth, structure, or appearance.

aromatic hydrocarbon: a class of unsaturated hydrocarbons characterized by rings containing six carbon atoms and three conjugated double bonds.

broadcasting: method of reproduction utilized by some corals in which gametes are released into the water column

brooding: method of reproduction in some corals in which reproduction occurs in place.

bunker oils: viscous fuel oils (usually designated by letters B or C, or numbers 5 or 6) used primarily in marine and industrial boilers.

coenosarc: the hollow stem and living basal parts of colonial hydroids, which houses a continuous gastrovascular cavity.

depuration: passive or diffusive loss of contaminants from an organism.

dioecious: having the male and female reproductive organs in separate individuals (most animal species are dioecious, as are some plants, such as asparagus).

dispersant; a chemical formulation containing surface active agents and/or solvents that lower the interfacial tension between oil and water and enable the oil film to break up more easily under natural wave action or mechanical agitation.

dispersion: the distribution of spilled oil into the upper layers of the water column by natural wave action or by application of chemical dispersants.

elimination: combined processes of metabolism, excretion, and diffusive loss of contaminants from an organism.

FAA: Federal Aviation Administration

gas chromatography: analytical method used to aid identification of oil constituents in which a mixture of volatile substances (e.g., oil) is carried by an inert gas through a tube containing a non-volatile liquid supported on an inert porous solid. Movement of the various components is selectively retarded, thus permitting their separation and subsequent identification.

GC-FID: gas chromatography/flame ionization detector

GC/MS: gas chromatography/mass spectrometry

GLC/FID: gas-liquid chromatography/flame ionization detector

gonochoric: animals having separate sexes.

hermaphroditic: an animal possessing both male and female functional reproductive organs, such as the earthworm; or a unisexual animal having male and female gonads as an aberration.

hermatypic: reef-building

hormesis: enhancement of a physiological process, also termed Arndt-Schult effect, in response to stress.

hydrocarbons: organic compounds composed only of the elements carbon and hydrogen; principal constituents of crude oils, natural gas, and refined petroleum products.

LD_{50}: concentration of a substance that causes death in 50 percent of a test population

massive: when referring to coral types, describes a skeleton formed as a solid block rather than being branched or plate-like.

MSRC: Marine Spill Response Corporation

octocoral: one of two subclasses of corals, colonial anthozoan coelenterates with eight-branched tentacles, including soft corals, horny corals, and sea pens.

planula: the ciliated, free-swimming larva of a coelenterate.

ppb: parts per billion

ppm: parts per million

protandrous: sequential hermaphroditism in which an organism changes sex from male to female.

protogynous: sequential hermaphroditism in which an organism changes sex from female to male.

RD_{50}: concentration of a substance that results in a given response in 50 percent of a test population scleractinian: any of the corals of the order Scleractinia; this still-abundant order first appeared in the Triassic and was the first to replace the tabulate and rugose corals, which had disappeared at the end of the Permian.

sclerochronology: the study of scleractinian coral growth rings

WAF: water-accommodated fraction, that portion of a substance dissolved or suspended in water.

weathering: the alteration of the physical and chemical properties of spilled oil through a series of natural biological, physical, and chemical processes beginning when the spill occurs and continuing as long as the oil remains in the environment; contributing processes include spreading, evaporation, dissolution, dispersion, photochemical oxidation, emulsification, microbial degradation, adsorption to suspended particulate material, stranding, or sedimentation.

WSF: water-soluble fraction, that portion of a substance which truly dissolves in water.

zooxanthellae: plantlike flagellate protozoans in the order Dinoflagellida, photosynthesizing symbiotes inside the cells of sponges, corals, and others.

APPENDIX B

CORAL SPECIES CITATION LIST
Cross-reference of coral species mentioned in referenced papers

Acropora cervicornia
Neff and Anderson (1981); Ballou et al. (1987)

Acropora elsyii
Neff et al. (1998)

Acropora formosa
Craik (1991); Harriott (1993); Harrison et al. (1990)

Acropora millepora
Negri and Heyward (2000)

Acropora palmata
Neff and Anderson (1981); Bak (1987)

Acropora spp.
Spooner (1970); LeGore et al. (1989)

Agaricia agaricites
Lewis (1971)

Agaricia tenuifolia
Ballou et al. (1987); Burns and Knap (1989); Dodge et al. (1995)

Diploria clivosa
Jackson et al. (1989); Guzmán et al. (1994)

Diploria strigosa
Bak (1987); Knap et al. (1982); Cook and Knap (1983); Dodge et al. (1984); Guzmán et al. (1994)

Favia fragrum
Lewis (1971); Neff and Anderson (1981); Kennedy et al. (1992)

Favia speciosa
Grant (1970)

Fungia scutaria
Johannes (1975)

Goniopora sp.
LeGore (1989)

Heteroxenia fuscescense
Cohen et al. (1977); Kushmaro et al. (1996); Epstein et al. (2000)

Madracis asperula
Lewis (1971)

Madracis decatis
Neff and Anderson (1981)

Madracis mirabilis
Elgershuizen and deKruijf (1976)

Manicina areolata
Peters et al. (1981)

Millepora sp.
Neff and Anderson (1981)

Montastrea annularis
Neff and Anderson (1981); Bak (1987); Ballou et al. (1987); Kennedy et al. (1992)

Montipora verrucosa
Johannes (1975)

Oculina diffusa
Neff and Anderson (1981)

Pavona gigantea
Birkeland et al. (1976)

Platigyra sp.
Mitchell and Chet (1975); LeGore et al. (1989)

Pocillopora cf. damicornis
Birkeland et al. (1976); Te (1991); Harriott (1993)

Porites spp.
Jackson et al. (1989); Lough and Barnes (1997)

Porites astreoides
Rützler and Sterrer (1970); Jackson et al. (1989); Guzmán et al. (1994); Ruesink (1997)

Porites compressa
Johannes (1975)

Porites furcata

Rützler and Sterrer (1970); Birkeland et al. (1976)

Porites lutea
Readman et al. (1996)

Porites porites
Lewis (1971); Ballou et al. (1987); Dodge et al. (1995)

Porites sp.
LeGore (1989)

Psammocora stellata
Birkeland et al. (1976)

Siderastrea radians
Rützler and Sterrer (1970)

Siderastrea siderea
Burns and Knap (1989); Jackson et al. (1989); Guzmán and Holst (1993); Guzmán et al. (1994); Guzmán and Jarvis (1996); Ruesink (1997)

Stylophora pistillata
Rinkevich and Loya (1977); Loya and Rinkevich (1979); Rinkevich and Loya (1979); Rinkevich and Loya (1983); Epstein et al. (2000)

Tubastrea aurea
Brey et al. (1995)

APPENDIX C

REPRODUCTIVE TYPE AND TIMING, BY REGION

(Adapted from Richmond and Hunter 1990; and Kenyon 1992)

These figures are intended to provide a quick reference about reproductive type (brood or spawn) and timing over the year for coral species for which information is available. Source information was extracted from tables in Richmond and Hunter (1990) and, for *Acropora* species in Hawaii, from Kenyon (1992). Refer to these papers for additional and more detailed data, as well as original data sources.

The information is organized by region (Caribbean, Great Barrier Reef, Hawaii, Okinawa, Central Pacific, and Red Sea). Coral species for which reproductive information was available are listed down the vertical axis, and months of the year are listed across the horizontal axis. If a species is known to spawn (i.e., broadcast zygotes into the water column) it is shown in red; if a species broods (i.e., retains gametes within a colony) it is shown in blue. If reproductive strategy is unknown, it is shown in gray. A white bar reflects that the species was reported to be possibly sterile. A dotted line indicates that timing of reproduction is unknown.

The graphic presentation of data is an easy way to identify periods of increased sensitivity or susceptibility for the coral species of a given region. For example, in examining the ample information for the Great Barrier Reef, it becomes apparent that November would be an especially bad time of the year to have an oil spill. A responder or resource manager may choose to structure a response or cleanup strategy around this fact by (for example) minimizing activities that could increase levels of oil in the upper portions of the water column where coral gametes and larvae would be expected.