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Annual Review

ECOTOXICOLOGY OF TROPICAL MARINE ECOSYSTEMS

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Abstract—The negative effects of chemical contaminants on tropical marine ecosystems are of increasing concern as human populations expand adjacent to these communities. Watershed streams and ground water carry a variety of chemicals from agricultural, industrial, and domestic activities, while winds and currents transport pollutants from atmospheric and oceanic sources to these coastal ecosystems. The implications of the limited information available on impacts of chemical stressors on mangrove forests, seagrass meadows, and coral reefs are discussed in the context of ecosystem management and ecological risk assessment. Three classes of pollutants have received attention: heavy metals, petroleum, and synthetic organics such as herbicides and pesticides. Heavy metals have been detected in all three ecosystems, causing physiological stress, reduced reproductive success, and outright mortality in associated invertebrates and fishes. Oil spills have been responsible for the destruction of entire coastal shallow-water communities, with recovery requiring years. Herbicides are particularly detrimental to mangroves and seagrasses and adversely affect the animal–algal symbioses in corals. Pesticides interfere with chemical cues responsible for key biological processes, including reproduction and recruitment of a variety of organisms. Information is lacking with regard to long-term recovery, indicator species, and biomarkers for tropical communities. Critical areas that are beginning to be addressed include the development of appropriate benchmarks for risk assessment, baseline monitoring criteria, and effective management strategies to protect tropical marine ecosystems in the face of mounting anthropogenic disturbance.

Keywords—Ecotoxicology Coral reefs Mangroves Seagrasses Chemical contaminants

INTRODUCTION

Islands and coastal waters of the tropics and subtropics possess unique, speciose, and highly productive ecosystems, including mangrove forests, seagrass meadows, and coral reefs. These economically and culturally valuable ecosystems had long been considered to be pristine and safe from the degradation of human activities. However, in the late 1960s and early 1970s, as natural resource exploitation spread from temperate regions to the tropics, environmental concerns led to numerous reports documenting the decline of these ecosystems [1]. Scientists now agree that shallow-water tropical marine ecosystems in many areas are degraded, if not destroyed, as a result of exposure to sedimentation, nutrient loading, and chemical contaminants as well as physical habitat destruction associated with human harvesting of wood, and food, mining of coral block and limestone, and dredging and filling for construction [2–6]. Destruction of these important ecosystems, which form and protect land masses from the open ocean, has serious consequences in areas where tropical storms, hurricanes, or typhoons occur. Disruption of ecological processes in these ecosystems affects not only the resident organisms but also the humans who depend on them for food and recreation.

At the interface between land and sea, wave action is dissipated by salt-tolerant mangrove forests. The roots of these trees trap and stabilize sediment as peat (a combination of silt, sand, and decomposing forest detritus), which prevents fine particles in coastal rivers from reaching offshore seagrass beds and coral reefs. The roots provide a substratum for encrusting algae, sponges, and molluscs, attracting crustaceans and fishes that feed on these resources and providing shelter for these organisms, many of which are juveniles of species found in seagrasses and on coral reefs as adults [7]. The leaf detritus and associated microbes form the basis of the mangrove community food web and are also a food source when transported to adjacent deeper waters [8].

Seagrasses inhabit mud or sand bottoms to a depth of 30 m or more throughout tropical and subtropical areas. Principal tropical seagrasses include the flat-bladed genera *Thalassia, Halodule,* and *Halophila* and round-bladed *Syringodium* [9]. The blades of the grasses can extend approx. 40 to 60 cm above the substratum and are anchored by an extensive system of rhizomes. Seagrasses grow rapidly and have high organic productivity, serving as food for herbivorous urchins, molluscs, fishes, and sea turtles, as well as producing large quantities of detrital material, which serves as another food source when exported to adjacent ecosystems. The dense leaves and roots also buffer the coastline, trapping sediment and reducing erosion while providing attachment for epiphytes and shelter for a variety of organisms [9,10].

Coral reefs are biogenic fringing, bank, barrier, or atoll structures that are the seaward component of much of the tropical shoreline, buffering it from wave action. Coral reefs are formed primarily by the calcification processes of coralline algae and scleractinian corals, producing structural habitat for filamentous and fleshy algae, invertebrates, and fishes. The unique symbiosis of the reef-building scleractinia with single-celled

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dinoflagellates, also known as zooxanthellae, enables these organisms to live in clear, oligotrophic waters with low nutrient levels [11,12]. Corals are very susceptible to changes in environmental conditions including light levels, nutrients, and temperature, outside the range that they normally experience. Such changes affect the coral–algal symbiosis and, ultimately, calcification and the entire reef community. The death of key organisms on the reef or the shift from an autotrophic to a heterotrophic (suspension/detritus-feeding) community changes the dominant ecological process from calcium carbonate deposition to erosion [13]. Many organisms (e.g., urchins, sponges, fishes) assist in the destruction of the reef and the formation of sand.

Like coral reef ecosystems, seagrass meadows and mangrove forests are also adversely affected by changes in water quality (increased or decreased temperature, salinity, oxygen; increased turbidity; presence of chemical contaminants) or substratum quality (alterations in grain size, composition, porosity), which can cause direct mortality or physiological alterations [see reviews in 1–4,14]. Sublethal effects of environmental changes are reflected in impaired reproduction and recruitment of key species, eventually resulting in a cascade of losses within the ecosystem [13].

Because concerns about the demise of coral reefs and adjacent ecosystems have increased, there have been numerous international efforts to conserve and protect them, primarily by designating marine parks (e.g., Great Barrier Reef Marine Park, Queensland, Australia; Florida Keys National Marine Sanctuary, Florida, USA). Efforts are also under way to educate people and change attitudes and practices that are adversely affecting the ecosystems, including those associated with fishing and land management. Managers of tropical marine ecosystems need to be able to synthesize information on stressors and develop methods to assess and predict risks to these ecosystems. To determine risks, however, there must be knowledge of the magnitude of stressors at which effects become apparent. Although much research has been conducted on factors affecting tropical marine ecosystems, particularly coral reefs, efforts to direct research to address management goals and decisions have been limited until recently.

The ''Framework for Ecological Risk Assessment'' [15] provides guidance to evaluate what is known about an ecological resource and its susceptibility to physical, chemical, or biological stressors with the intent of developing one or more estimates of risk involved from past or potential exposure to these stressors [16,17]. Statements of ecological consequences [17] help provide managers with options to more effectively protect resources at greatest risk while balancing costs and benefits to society. Although most ecological risk assessments have been site- or stressor-specific, efforts are under way to develop assessments on larger temporal and spatial scales. This is particularly important for ecosystems, watersheds, or landscapes with numerous ecological components involved in complex interactions and exposed to multiple and different types of stressors. Such an approach might also be appropriate for coral reefs and adjacent tropical marine ecosystems.

A recent review of pollution in tropical marine ecosystems discussed a variety of stressors and effects and proposed some preliminary tolerance levels for corals on the Great Barrier Reef [4]. This article focuses on the current status of knowledge on chemical contamination in tropical coastal ecosystems, how this information might be used in the context of ecological risk assessment, and the need for further ecotoxicology research. The valued ecological resources for these ecosystems are primary producers, those organisms that provide physical structure and food for the major communities on which other organisms, including humans, depend. The following resources, which might serve as index species or the focus of assessment endpoints in assessing ecological risk, are targeted: for mangrove forests, *Rhizophora* and *Avicennia* spp.; for seagrass beds, the turtle grass *Thalassia testudinum*; and for coral reefs, scleractinian corals. The three ecosystems differ in their structural basis (tree, macrophyte, and colonial invertebrate containing symbiotic algae, respectively) as well as in longevity (decades for mangroves, 1 year for seagrasses, centuries for corals). Tropical fishes are also discussed, although there is limited research on the effects of pollutants on these organisms. Many fishes migrate from the reef to seagrass beds and mangroves or spend part of their lives in the adjacent ecosystems, potentially transferring pollutants from one ecosystem to another. Fishes are important components of marine food webs and can pass bioaccumulative chemicals acquired from these primary producers and/or invertebrate prey to fishes at higher trophic levels and to coastal birds and mammals that prey on fishes.

Here we review the current data and make recommendations for further research that can benefit managers and users in protecting these important ecosystems [18]. In tropical marine communities, waterborne chemical threats can come from far as well as near, from both point and nonpoint sources. Through a better understanding of the tropical marine ecosystems and their responses to chemical stressors, managers will be able to more effectively address the goals of reducing negative effects and improving mitigation measures by designing appropriate policies and regulations.

STATUS OF KNOWLEDGE ABOUT CHEMICAL STRESSORS IN TROPICAL MARINE ECOSYSTEMS

The sources of chemical contamination in tropical marine environments are similar to those in temperate marine ecosystems (Fig. 1). Terrestrial runoff from rivers and streams, urban areas, and agricultural areas is probably the most important. In addition, some urban areas have sewage outfalls near coral reefs [19,20] or can contribute to the contaminant load through desalination plants [21]. Landfills can leach directly or indirectly into shallow water tables, streams, and the coastal zone, a particular problem on porous limestone islands. Many reefs are located near population centers and are the focus of tourism, which makes them vulnerable to chemical inputs from recreational uses and industries related to recreation (e.g., boat manufacturing, boating, fueling at marinas). Reefs are economically and socially important for their fishery resources, and chemical-based fishing methods (e.g., using bleach or cyanide) are not uncommon in developing regions.

Some shallow coastal zones are near shipping lanes where ships pump out contaminated bilge water or spill their cargo. Dredging of channels to allow ship traffic can increase chemical concentrations through resuspension of buried organics or heavy metals. Ocean disposal of waste chemicals or incidental chemical spills are also contributing factors. Industrial inputs from coastal mining and smelting operations [22] are sources of heavy metals; offshore oil and gas activities and coastal petroleum refineries contribute metals, phenols, and oil [20] from tanker loading, spills, well blowouts, and pipeline ruptures.

Because the transport of most point and nonpoint source

Fig. 1. View from the reef off Guayanilla and Tallaboa Bays, a heavily industrialized area on the southern coast of Puerto Rico. Similarly developed coastal areas occur in the vicinity of mangrove forests, seagrass meadows, and coral reefs around the world.

toxicants is primarily from inland to offshore, the ecosystem sections in this document are arranged accordingly, i.e., from inshore (mangroves) to offshore (coral reefs). The fate and transport of chemical pollutants, factors influencing bioavailability and toxicity, and ecosystem effects are discussed under the categories of metals, oil and dispersed oil, and pesticides and other organics. One general factor affecting the bioavailability of contaminants is that the organic carbon content of sediments decreases from mangroves to seagrass beds to coral reefs. An overview of the ecological significance of these contaminants and reports of trends in contaminant loading in these ecosystems are also included in the following discussion.

Mangrove forests

Mangrove forests are an important buffer for adjacent marine ecosystems, trapping sediments and nutrients, as well as many anthropogenic chemical contaminants. Different species of mangrove trees have different types of sediment-trapping root systems adapted for combatting the low-oxygen conditions in the peat, including the pores or lenticels on the branching prop roots of the red mangroves (*Rhizophora* spp.) or on the cylindrical pneumatophores of black mangroves (*Avicennia* spp.), which project above the water level. Species in these genera are the most salt-tolerant mangroves, and they form extensive forest communities criss-crossed by tide-draining channels along tropical and subtropical coasts. Physical destruction of this ecosystem for building materials, fuel, and land reclamation has increased greatly during this century, largely because of population pressures.

Heavy metals. Impacts of heavy metal exposures on mangrove trees are apparently minor or nonexistent. Mangrove sediments, composed of fine particles with a high organic content and low pH, are particularly effective in sequestering potentially toxic heavy metals, which are immobilized as sulfides in the usually anaerobic sediments [23–26]. However, metals can be reintroduced to nearshore waters when they are taken up by mangrove trees and concentrated in exported leaf detritus. Storms and human activities such as dredging or clearing of mangrove forests also remobilize metals and facilitate transport from mangrove forests to coastal waters.

Whether metals are available for uptake by mangrove trees and to what extent metals are transported out of mangrove forests to coastal waters are debatable. Silva et al. [27] found that sediments were the main reservoir of the total metal content in a mangrove forest dominated by *Rhizophora mangle* and concluded that the major portion of metals is probably unavailable for mangrove uptake. However, *Rhizophora* species appear to take up metals in a different manner and to a lesser extent than other mangroves species [28–31]. In other studies, metal concentrations were higher in leaves than in water or sediment [28,29,31]. Tam et al. [32] did not detect lead, chromium, or cadmium in leaf samples from the mangroves *Kandelia candel* and *Aegiceras corniculatum* in the Futian National Nature Reserve, China; these metals occurred in sediments at high concentrations at some sites but were not bioavailable. Metals were also concentrated in perennial tissues such as the trunk and limbs [27].

Lacerda et al. [33] argued that plant litter is very poor in trace metals, leading to very low export. Even if the material is poor in trace metals, substantial quantities of mangrove detritus are exported from the forests and used as a food source [34,35]. This serves as an avenue by which metals could be transported from mangrove forests to surrounding communities. Mercury, a bioaccumulative metal, was detected in red mangroves, mangrove oysters, and a variety of fishes, including great barracuda, from Puerto Rico and the U.S. Virgin Islands [36]. Mangrove leaf detritus is enriched in metals as it ages [30], possibly as a result of the loss of organic material, and metals are enriched in suspended matter [24,37]. DeLaune et al. [38] calculated that 3% of the annual accumulation of copper and 5% of the annual accumulation of zinc in a mangrove ecosystem were exported through detritus and that cadmium, lead, and manganese were also exported via detritus. Nye [39] estimated that 3,499 kg of iron, 9,130 kg of manganese, 1,063 kg of copper, and 412 kg of zinc are exported annually from mangrove forests in southeast Florida. Tidal deposition might also influence the distribution of metals in mangrove sediments [40]. Table 1 summarizes ranges of metal contaminants measured in mangrove forests.

Oil and dispersed oil. Mangroves are very sensitive to oil. The low wave action and small tidal amplitude characteristic of mangrove swamps make them excellent traps for oil slicks [41,42]. At very low concentrations, oil stimulates growth. This phenomenon, hormesis, might be due to the hormone-

Table 1. Metal concentrations in mangrove ecosystems[a]

Component	Fe	Zn	Cu	Mn	Cd	Pb
Sediments (μg/g dry weight)	100–33,492	0.28–379	0.3–75	1.23–640	0.1–2.39	1–650
Suspended material (μg/L)	195–2,808	18–595	62–76	466–788	2.85–3.2	21–139
Rhizophora spp. (μg/g dry weight)	0.16–166	0.03–32	0.12–11	0.42–497	0.04–0.24	0.43–27
Inverterates (μg/g dry weight)	124–886	30–1,800	2.6–20	4.8–31	0.4–10	0.9–5.06

[a] Compiled from [24,27,29–33,39,40,349–361].

like action of some aromatic hydrocarbons or to selective action against a parasite [43–47]. However, because mangroves span the air/water boundary, they come into direct contact with spilled oil, and the physiological adaptations that mangroves have evolved to survive anaerobic soils make them particularly vulnerable to smothering by oil slicks. The specialized gas-exchange pneumatophores are susceptible to clogging, and when they are blocked the roots die from lack of oxygen [48,49].

Oxygen concentrations are lower in warmer waters [50], whereas respiratory rates are higher, and aerobic tropical organisms must live at oxygen levels closer to their lethal limits than do the biota of cooler waters [1,3]. Spilled oil has a large biological oxygen demand (BOD), which can further exacerbate the oxygen shortage. In addition, oil contains toxic compounds that can disrupt the metabolic activity of sensitive organisms [23]. In experiments with oiled pneumatophores, both light and heavy oils effectively blocked gas exchange through the lenticels, even when most of the oil was washed away [51]. Damage to pneumatophores can take a year or more to become noticeable [43,52,53]. When aerial roots regrow, they are often deformed or abnormal [49,53,54–57].

The anaerobic conditions and low redox potential values found in mangrove sediments are known to be unfavorable for the biodegradation of hydrocarbons [58], and substantial concentrations of hydrocarbons remain in mangrove sediments 10 to 30 years after spills [59,60]. Three years after the *Zoe Colocotronis* spill in Puerto Rico, soil samples from the mangrove community had up to 80,000 ppm extractable hydrocarbons [61]. Mangroves can translocate aromatic hydrocarbons into their tissues [46], and the toxic components of retained oil continue to cause sublethal effects, including reduced productivity, lower rates of litter production, and lower seedling survival [62]. In this state of chronic stress, mangroves are highly susceptible to any additional perturbation or stress [62]. Dodge et al. [60] noted in an experimental field exposure that Prudhoe Bay crude oil killed 17% of mangrove trees after 2 years and 46% after 10 years, with degraded oil still present. Gundlach and Hayes [63] concluded that mangroves are the coastal marine environment most vulnerable to oil spill impacts [see also 64]. Oil spills from tankers have caused massive die-offs of mangroves and their associated organisms [42,48,62,65–71, reviewed in 55].

Once a mangrove forest has been damaged by oil, recovery is often slow. Some areas of Bahia Sucia, Puerto Rico, that were destroyed by a large spill in 1973 were still devoid of mangrove growth 20 years later [72]. Subsequent erosion prevented recovery, and it takes at least 3 years for debris from the former mangrove community to decay [73]. Lugo et al. [52] reported mangrove seedlings to be more resistant to oil; however, seedlings that colonize an area before debris has decomposed can be scoured away by the tidal movement of debris. When recovery occurs, it takes a minimum of about 20 years [74–78], and trees might not reach their full height for 80 years. The rate of recolonization often depends on the size of the patch affected, with large patches relying on recruitment of planktonic propagules [79]. Infaunal populations might recover rapidly, but some epibiont invertebrate populations, including shrimp, polychaetes, cerethid snails, and sipunculids [69], can be affected for several years [80–82]. Klekowski et al. [72] reported that the biota of oil-polluted mangrove habitats might experience increased mutation rates.

Because mangroves are very sensitive to oil spills and slow to recover, it has been suggested that every effort should be made to keep spilled oil from reaching mangroves [48]. In some cases, treating oil with dispersants mitigated damage [43,46,47,48,60,83,84], but in other cases, addition of a dispersant increased toxicity [43,46–48,83]. The toxicity of an added dispersant can be affected by the type of dispersant used and the concentration in water, the type of oil spilled, and the age and species of the mangroves treated. The toxicity of the dispersant to adjacent marine communities should also be taken into consideration. Seagrasses and corals are very sensitive to Corexit 9527, but Elastosol, Cold Clean, and Finasol are much less toxic to these organisms and have been recommended for tropical estuarine spills when mechanical procedures are inadequate to contain the spill [85].

Pesticides and herbicides. Mangroves are also susceptible to herbicides [83–88]. During the Second Indochina War, 18.9 million gallons of herbicides was sprayed on forested and agricultural areas of Vietnam. Of all the habitat types in South Vietnam, the coastal mangrove forests were the most vulnerable to wartime herbicide operations [89–91]. In the Rung Sat area, 73% of mangrove cover was destroyed [89]. It is estimated that 41% (124,000 ha) of the total mangrove forest area of Vietnam experienced significant mortality during the Second Indochina War [90–92].

The reason for this sensitivity is poorly understood but could be related to the physiological stress of living in a saline environment [77]. Species of *Avicennia* have been shown to be more resistant to 2,4-dichlorophenoxyacetic acid (2,4-D) [86,93], but *Rhizophora,* the most economically and ecologically important genus of trees in South Vietnam, was especially sensitive to hormone-mimicking pesticides [90,91]. In an experimental exposure of mangrove species in Florida, USA, to 2,4-D, the buttonwood *Laguncularia racemosa* was the most sensitive species tested, *Avicennia germinans* was the most resistant, and *R. mangle* was intermediate [93]. Seedlings were also much more sensitive than adults. *Rhizophora mangle* has been shown to take up and translocate 2,4-D and picloram to various plant parts [93–95], and 2,4-D induced the breakdown of cell walls in both roots and leaves [94,95]. Death of mangroves exposed to 2,4-D is believed to be due to the loss of meristematic tissues [96].

Some herbicide residues remained in the environment 5 to 10 years after intensive spraying, but they were far below levels that would inhibit seedling establishment [89]. Laboratory analyses of residual herbicides and breakdown products in aquatic or marine sediments are limited but indicate that most do not persist at high concentrations [97]. However, Snedaker [90] warned that the absence of data should not be interpreted to mean that no problems exist.

In the Rung Sat area of South Vietnam, numerous differences have been noted in biota inhabiting defoliated versus undisturbed areas, but it is not known to what degree these differences were caused by herbicide exposure or habitat destruction. The abundance and species richness of planktonic organisms and large fish were lower in sprayed areas, while fish eggs and larvae were more numerous in denuded regions [89], possibly as a result of an absence of predators. Orians and Pfeiffer [98] reported an enormous reduction in numbers of birds. Marine fishery stocks declined, and certain species disappeared [99,100].

Although the devastation of mangrove forests by herbicides is uncontested, the rate of recovery of sprayed forests, like mangroves destroyed by oil spills, is uncertain, with estimates varying from 20 years [76,90] to more than 100 years [89]. Reestablishment might occur only along the edges of the river channels and backwaters [78,98]. Natural regeneration of mangroves has been minimal in coastal South Vietnam [91,101]. Important factors impeding recovery are the almost complete elimination of mature seed- or propagule-bearing trees in sprayed areas [89,90,102], lack of vegetative cover [89], debris [102–104], and increased erosion [91,102]. However, erosion might actually accelerate revegetation by exposing reduced sediments, lowering land elevation relative to surface water elevation, promoting beneficial surface water recirculation and flushing, and removing persistent herbicides from the system. In certain deforested areas, weed species now dominate the habitat to the exclusion of various mangrove species and will probably remain dominant for some time [90].

Very little information is available on the possible effects of other pesticides on mangrove forests. Mosquito larvicides such as temephos are regularly sprayed on mangrove communities in the Florida Keys and elsewhere. Concentrations of temephos decline rapidly after application [105–107]. Temephos persists on mangrove leaves for up to 72 h and in oysters for up to 48 h [107]. At normal application rates, concentrations of temephos in water reach levels that are toxic to mysids [107] and can cause sublethal effects in fish [105,108]. Temephos has also been shown to have a significant effect on fiddler crab populations at normal application rates [109–111].

Seagrass meadows

Seven of the 12 genera of seagrasses occupy tropical climates (*Halodule, Cymodocea, Syringodium, Thallassodendrum, Enhalus, Thalassia, Halophila*). A few species of the remaining five genera are occasionally found in subtropical and tropical waters [112]. The majority of the work on the impact of anthropogenic inputs has been performed on temperate grasses such as *Zostera* and *Posidonia*. A decline in the health of seagrass beds has been reported recently in many areas of the world, including the United States [113–115] and Australia [116], and shifts in dominant species have been found to correlate with pollutant exposure [117,118]. Seagrass die-offs resulting from chemical contaminants, as well as physical damage and disease, reduce habitat availability and sediment-trapping ability, leading to increased transport of contaminated sediment particles to coral reefs. In addition, decreased decomposition of detritus alters carbon cycling and transport of contaminants.

Heavy metals. In temperate species, metals can be incorporated into seagrass leaf tissues directly from water and sediment [119], and in uncontaminated areas, seagrass leaves constitute the major transport pathway for the cycling of copper, iron, manganese, and zinc between these media [120]. Trace metals in *Halodule wrightii* are positively correlated with concentrations in sediment [121]. Metal concentrations tend to be higher in leaves than in shoots for Indonesian tropical seagrasses [122] and *H. wrightii* [121]. These results reflect the bioavailability and translocation of contaminants to biomass above the sediment. Biotic transport of trace metals from seagrass detritus may occur through the food chain [22].

Field studies of seagrasses have shown that several species are capable of accumulating a range of trace metals [e.g., 121–124]. Baseline data for nickel, copper, lead, and zinc have been collected for Indonesian seagrasses [122]. *Halodule ovalis* had higher levels of zinc than other congeners [122,124] and therefore might be a zinc accumulator. In *Thalassia* microcosm studies, seagrass communities were shown to rapidly take up and process tributyltin [125], increasing the potential exposure of associated fauna.

Several studies using microcosms demonstrated a similar potential for ecosystem-wide effects in tropical seagrass beds from toxic chemicals such as tributyltin [130], as have been found in metal exposures of temperate seagrass species [22,126]. At 22 μg/L of tributyltin, carbon turnover decreased, with reduced detrital decomposition and above-sediment plant biomass, in *T. testudinum* [127]. Benthic invertebrates decreased in abundance [128]. *Cymadusa* and *Crepidula* were identified as indicators of impending invertebrate mortality. Exposure to drilling muds containing ferrochrome lignosulfonate (FCLS) and other metals such as barium affected the abundance and species richness of the invertebrate community in a *Thalassia* microcosm [129,130]. Associated epiphytes had decreased photosynthetic potential and biomass when treated with drilling mud [129,131]. The concentration of chlorophyll *a* per gram dry weight of leaf tissue and the rate of leaf decomposition were reduced [129,130]. Productivity, measured as growth rate and carbon uptake, was reduced, with an observed seasonal variability [131].

Oil and dispersed oil. Damage to seagrass communities from oil exposure includes acute mortality resulting from physical impacts (i.e., smothering, fouling, asphyxiation) and chemical toxicity; indirect mortality as the result of light loss, death of food sources, or the destruction or removal of habitat; destruction of sensitive juvenile fishes and invertebrates; and accumulation of potentially carcinogenic or mutagenic substances in the food chain [132]. The 1986 spill at Bahia Las Minas, Panama, demonstrated several of these effects. Oil and possibly oil plus dispersant caused mortality of intertidal *Thalassia* beds [133], as well as damage to shoreward margins of the beds, which receded [118]. *Thalassia* subsurface biomass increased at oiled sites, whereas *Syringodium* biomass continued to decline [118]. These findings agree with results of Thorhaug et al. [134], who exposed both species to oil. In an experimental spill, hormesis, evident as increased biomass of seagrass, was observed, with minor or no overall effects on seagrasses resulting from oil or chemically-dispersed oil up to 10 years afterward [47,60].

Fauna and flora associated with seagrasses are also affected

Table 2. Metal concentrations in coral reef ecosystems[a,b]

Component	Fe	Zn	Cu	Pb	Ni
Water column (mg/L)	1.0–5.93	0.02–1.5	0.01–1.8	0.18	
Sediment (μg/g)	237–11,445	7.6–40	2.2–17	18–45	74–122.6
Coral skeleton (μg/g)	ND–560	0.08–25	0.24–18	0.04–39	ND–126
Coral tissue (μg/g dry weight)	ND	ND–126	7.5–18		

[a] Compiled from [196,198,362].
[b] ND = not detected.

by oil. Fleshy and calcareous algae did not recolonize oil-damaged areas at Bahia Las Minas. Infauna, nearly completely killed by oil exposure, gradually returned to abundances above prespill levels [118]. However, only species with high reproductive potential or planktonic stages recovered quickly. The effects of oil on tropical fish have not been extensively studied. Sublethal doses of No. 2 fuel oil, at concentrations greater than 1.0 ppm water-soluble fraction, caused decreased growth and slow avoidance responses in the estuarine spotted seatrout, *Cynoscion nebulosus* [135]. Metabolic rates were affected by petrochemical effluents in the warm-water pinfish (*Lagodon rhomboides*); sublethal effects were aggravated by salinity changes and differed with size class [136].

Oil in direct contact with seagrasses causes decreasing growth rates, smothering of leaves, leaf yellowing and browning, loosening of leaves, and decreasing percent cover [137]. Laboratory experiments have demonstrated reduced photosynthetic ability depending on oil type, exposure time concentration, and species tested [134,138,139]. However, the WSF of Kuwait crude oil had no effect on the photosynthetic parameters or respiration in seagrasses indigenous to the Persian Gulf [140]. Field observations of species composition, distribution, abundance, productivity, and morphology for three species of seagrass in the Arabian Gulf indicated that the Gulf War oil spill had little or no impact on the seagrasses [141].

Of the Caribbean seagrasses exposed experimentally to dispersed oil, *T. testudinum* had a higher LC50 than *H. wrightii* and *S. filliforme* [134]. Dispersant type and length of exposure influence response. The latter two species were more sensitive to dispersant alone than *T. testudinum*, which showed no effect after 100 h of exposure [134]. *Thalassia* was most affected by exposure to dispersant over longer time periods and exposure to greater concentrations of dispersant mixed with oil [142].

Pesticides and other organics. Seagrass beds located near agricultural regions likely receive runoff containing pesticides. Chemicals such as atrazine and pentachlorophenol (at concentrations of 1 ppm) depressed the rate of oxygen evolution (photosynthesis) and oxygen uptake (respiration) of leaves of *T. testudinum* [143]. Atrazine (30 ppm) caused a significant reduction in survival, production of new ramets, above-sediment biomass (greater than 50%), and growth of *H. wrightii* [144]. Cropping and variation in light and salinity did not influence the biological response to atrazine. *Halodule wrightii* appears to be more tolerant to pollution than other species [117,144]. At 50 ppb, organophosphorus, carbamate, and organochlorine pesticides, especially dichlorodiphenyltrichloroethane (DDT) and lindane, caused decreased net photosynthesis and increased dark respiration in *Halophila* and *Halodule* spp., but to a lesser degree than observed in the macroalgae tested [145].

Coral reefs

Although coral reefs are composed of diverse organisms, the major structural components of reefs are the scleractinian corals, with more than 600 species in tropical and subtropical climates. The reef-building, hermatypic coral species can form branching, foliose, and mound-, or boulder-shaped colonies occupying tens of cubic meters. These corals contain symbiotic algae (also known as zooxanthellae) within their gastrodermal tissue. This animal–algal symbiosis is maintained through chemical communication between the partners via the translocation of metabolites [11,146,147]. Branching species might be more susceptible to some chemical contaminants than are massive corals [148]. Scott [149] found that small-polyped species were more adversely affected by increased concentrations of heavy metals than are large-polyped species. Larger-polyped species appear to be better adapted to carnivory [150] and might be less dependent on high light levels required for successful algal photosynthesis. Coral species and individuals also differ in their ability to remove sediment particles falling on their surfaces, depending on their orientation, growth form, and mucus production [151]. Coral colonies are particularly susceptible to contaminants dissolved in seawater or adsorbed to particles because the layer of tissue covering the coral skeleton is thin (only approx. 100 μm) and rich in lipids, facilitating the direct uptake of lipophilic chemicals.

In addition, chemical communication plays an important role in mediating processes central to the existence and persistence of reefs. Reproduction and recruitment of many reef organisms are known to be chemically mediated. Changes in water quality, most notably caused by coastal pollution, can interfere with critical interactions among reef organisms and reef productivity and hence result in the eventual death of the coral reef [13,152]. Brooded planula larvae of some coral species might be afforded some protection from toxics; however, larvae of many broadcast spawners pass through sensitive early stages of development at the sea surface, where they can be exposed to contaminants in surface slicks. Reef communities can harbor an estimated standing crop of fishes 20 to 30 times greater than those found in temperate waters [153], but our understanding of the effects of chemicals on these fish communities, and on associated reef invertebrates and algae, is very limited.

Heavy metals. Elevated concentrations of metals have been found in the tissues of some reef invertebrates from contaminated sites (Table 2). Giant clams of the genus *Tridacna* collected from a populated atoll had significantly higher concentrations of iron, manganese, copper, zinc, and lead than clams from an unpopulated atoll [154]. Similar relationships have been seen in corals [155,156], and some corals might be better indicators of metal exposure than others [155,157,158].

Metals might occur in coral skeletons as the result of structural incorporation of metals into aragonite [159–162], inclusion of particulate materials in skeletal cavities [163], surface adsorption onto exposed skeleton [162,164,165], and chelation with the organic matrix of the skeleton [162,166,167]. Howard and Brown [155,158] found that metal concentrations were higher in the tissues than in the skeletons of corals near a tin smelter and suggested that coral skeletons were not a good indicator of environmental metal concentrations. However, several other studies reported greater concordance of data on metals concentrations in coral skeletons than in tissues at metal-contaminated sites [149,168–170].

Metals can enter coral tissues or skeleton by several pathways, and there is some evidence that corals might be able to regulate the concentrations of metals in their tissues [171–176]. Retraction of coral tissue in response to environmental stress exposes skeletal spines, which can directly take up metals from the surrounding seawater [169]. Another common response to physical and chemical stressors, particularly metals, is the production of copious amounts of mucus [177–184], which can effectively bind heavy metals [185–189] and may be involved in metals regulation [177,190]. Harland and Nganro [190] found that the rate of copper uptake in an anemone increased after mucus production ceased.

Organisms in low-nutrient tropical waters might be particularly sensitive to pollutants that can be metabolically substituted for essential elements such as manganese [191,192]. The symbiotic algae or zooxanthellae of corals can influence the skeletal concentrations of metals through enhancement of calcification rates [160,193], and they might be involved in the uptake of metals in cases where potentially toxic metals are metabolically substituted. Harland and Nganro [190] found that *Actinia equina*, an anemone lacking algal symbionts, exhibited lower uptake rates of zinc and cadmium than did *Anemonia viridis*, an anemone with algal symbionts. Benson and Summons [194] proposed that oceanic arsenate was assimilated by the symbiotic algae of giant clams. Symbiotic algae have been shown to accumulate higher concentrations of metals than do host tissues in corals [156,190,193] and in clams [194]. Sequestering metals in symbiotic algae might diminish possible toxic effects to the host [190]. The expulsion of symbiotic algae has been reported as a stress response to heavy metals [156,184,195] and has been proposed as a mechanism for the excretion of metals [176,190].

Tin was highly concentrated in reef sediments in the vicinity of a tin smelter in Thailand, and copper and zinc were slightly elevated [196]. Although there were no significant differences in coral cover, diversity, or growth rate in the study, the colony size of massive species such as *Porites* tended to be smaller in areas exposed to copper, zinc, and tin [196]. Later evidence indicated that the smelter effluent reduced the growth rate of branching corals [155], and when branching corals were transplanted to a contaminated site from a relatively pristine site 1 km distant, linear extension and calcium carbonate accretion were significantly reduced. Corals in a polluted estuary in Hong Kong also showed clear signs of stress resulting from increasing exposure to heavy metals, pesticides, nutrients, sewage, and turbidity [149]. Growth rates, species abundance, diversity, and cover declined between 1980 and 1986 during a period of decreasing water quality. Corals were replaced by ascidians, mussels, and holothurians, and corals from the more polluted site were slower growing and more heavily bioeroded [149]. Although the effects of heavy metals were not isolated from the possible effects of other stressors, the concentrations of heavy metals reported were as high as or higher than those reported by other investigators and increased with time [149]. Harland and Brown [197] noted that laboratory exposure of *Porites lutea* to elevated iron resulted in loss of symbiotic algae. The response was most noticeable in corals obtained from pristine areas, but response was less in corals that had been exposed to daily runoff from an enriched iron effluent, suggesting that the corals could develop a tolerance to the metal [156].

Metals have also been detected in reef fishes, and some effects on fish populations have been documented. Tissue metal concentrations of zinc, copper, cadmium, and mercury from 50 species of fish from the Great Barrier Reef were greater in liver than in muscle; however, all levels were very low. Of these four metals, mercury was positively correlated with size and trophic level [198]. Comparison of mercury levels in Hawaiian (USA) fish at different trophic levels showed that mercury tissue concentration increased from herbivore (0.022–0.036 ppm) to omnivore (0.058–0.070 ppm) to carnivore (0.75–0.80 ppm) and that levels are higher in the trophic food chains for fish feeding above the sediment/water interface than for those in direct contact with the sediment [199, see also 36]. Zinc concentrations in the liver of blue-striped grunt *Haemulon sciurus* from Bermuda were two to three times higher in grunts from clean sites (20–30 μg/g liver) than in grunts from sites with five times the levels of heavy metals (60–70 μg/g liver) [200]. Behavioral modifications, including erratic swimming, increased gill ventilation, and disrupted schooling ability, were noted in tropical fish exposed to heavy metals [201]. Increased mucus production, fin erosion, and change in color were also observed.

The drilling of oil wells near coral reefs has been a concern. Howard and Brown [157] noted that concentrations of up to 6 mg/L FCLS might occur within 100 m of a discharge. Six months of exposure to FCLS (concentrations not reported) decreased growth rates in the coral *Montastraea annularis* [202]. Linear growth and extension of calices (skeleton supporting the polyps) decreased in response to laboratory exposure to 100 mg/L drilling mud, and the lower calical relief might impair sediment-shedding capability [183]. Field assessment of a reef several years after drilling indicated a 70 to 90% reduction in abundance of foliose, branching, and plate-like corals within 85 to 115 m of a drilling site, although massive corals appeared relatively unaffected [203]. These findings agree well with predictions based on laboratory and field experiments, indicating that detrimental effects on corals seem probable within a minimum distance of 100 m from a drilling site [179,181–183,204–206].

Oil and dispersed oil. The potential threat of oil to reefs is high given their proximity to shipping lanes and offloading sites in shallow coastal environments. Studies indicate that small, frequent spills might be a greater threat to reefs than single-spill events [207–210]. Acute exposure of mangrove, seagrass, and reef ecosystems to spilled oil can occur from land runoff, as in the Gulf War or Panama spills, or from accidental discharges from ships at sea, as has occurred off Puerto Rico and in the Red Sea. Eighty-three of the more than 150 major oil spills in the tropics caused by shipping from 1974 to 1990 occurred near coral reefs, seagrass beds, reefs flats, sand beaches, and mangrove forests [208].

Petroleum reaching the reef might float above the reef and never directly contact it, although buoyant eggs and developing coral larvae could be affected. Reef flats are more vulnerable to direct coating by oil. Petroleum hydrocarbons, in the form of naturally and chemically dispersed oil or water-soluble components, are available for uptake by corals, and oil globules can adhere to the coral surface [133,211,212]. Petroleum hydrocarbons were found in oiled corals (25–50 mg hydrocarbon/g lipid) in Panama 5 months after the original spill [213]. Gas chromatographic patterns indicated uptake primarily from the water column and not from the sediment (19–715 μg oil/g reef sediment). Within 2 years, hydrocarbon residues were substantially reduced in sediments and coral tissues at high-energy reef sites [214]. Uptake and incorporation of petroleum hydrocarbons into coral tissues has also been demonstrated in experiments [212,215]. The relatively high lipid content of corals contributes to rapid uptake of aromatic hydrocarbons, while detoxification and depuration can be slow [212,216,217].

A combination of dispersants and oil has been shown to be more toxic than oil or dispersant alone [47,60,66,209,218–221]. Comparisons among studies are difficult because of interlaboratory variation in experimental methods, concentrations, oil fractions, durations of exposure, and species tested. No effects were observed for short exposures followed by short recovery times [222] and for short-term, low-level dosing with long recovery times [223,224]. However, even when no gross effects are seen, histological changes can occur from oil exposure [215].

Effects of oil on individual coral colonies range from tissue death to impaired reproduction to loss of the symbiotic algae (bleaching) [221]. Stress, defined by a decline in tentacular expansion, occurred at greater than 20 ppm of oil and dispersed oil, but recovery was observed within 1 week of exposure [220]. Photosynthetic rates decreased in some, but not all, laboratory-exposed corals [218,225]. Burns and Knap [213] noted a trend toward increasing protein/lipid ratios in corals exposed in the field for 5 months, suggesting that lipid reserves are reduced in oil-stressed animals. These reserves might be used to support proliferation of mucus secretory cells [215] and increased mucus production caused by exposure [133]. In the field, oiling can lead to the increased incidence of mortality and continued partial mortality of coral colonies [133]. During 5 years of monitoring on the reefs oiled by the refinery spill at Bahia Las Minas, the percentage of injured corals was correlated with sediment concentrations of oil [226,227].

In contrast to some laboratory studies that showed no change in coral growth rates with oil exposure [224], growth rates were reduced during longer-term field exposures [47,226]. In a field experiment in Panama, the dispersed-oil site had decreased percent cover of corals, which did not return to preexposure levels after 1 year. Growth rates of the corals *Montastraea annularis* and *Acropora cervicornis* were not affected by dosing with oil or dispersed oil, but the growth rates of *P. porites* and *Agaricia tenuifolia* decreased at the dispersed-oil site. These data suggest that dispersion of oil has a greater impact on reef organisms than oil alone. However, 10 years later coral coverage at the dispersed-oil site was the same as or greater than at the untreated site [60]. Newly developed dispersants might be less toxic to corals [228].

Oiling affects not only coral growth and tissue maintenance but also reproduction. Histological evidence confirmed impaired gonadal development [215,229] in both brooding and broadcasting species. In field exposures, increased injury and associated reduction of colony size led to decreased egg size, a sensitive indicator of coral reproductive viability, and decreased fecundity in colonies of *Siderastrea siderea* 5 years after the Bahia Las Minas oil spill in Panama [229]. Damaged reefs depend on recruits from unoiled reefs. Premature expulsion of planulae from brooding cnidarians following oil exposure was demonstrated by Loya and Rinkevich [230] and Ormond and Caldwell [231]. 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 area) could effectively eliminate an entire year of reproductive effort while continuing to reduce fecundity through partial mortality and impairment of gonadal development. In addition, successful recruitment and recruit survival can be compromised by oil exposure [209,226,232].

Short-term laboratory studies do not accurately predict long-term community effects, which vary greatly by region, species, and type of oil. Long-term studies are more likely to reflect impacts at sites with chronic petroleum pollution or at sites subjected to large oil spills. While Bak [207] stated that oil spills are unique events and chronic pollution is a greater threat to reefs, the results of his study of the effects of chronic refinery petroleum pollution on a reef do not differ greatly from those found at the refinery spill in Panama [226]. Both studies found decreased coral cover and diversity in Caribbean reefs due to petroleum exposure and decreased local recruitment [207,226]. *Acropora palmata* was identified as a sensitive indicator species, whereas *Diploria strigosa*, the subject of many of the Bermuda studies [233], might be hardier than other Caribbean species [207,226]. In the Gulf War spills, no long-term impacts on coral reefs of the region were identified [234]. Unlike the other field studies discussed, no dispersants were used on these spills, and significant weathering of oil may have occurred before it reached the reefs. Partial mortality of colonies, identified by Guzmán et al. [226] as a sensitive measure of impact, was observed on Kuwaiti reefs surveyed in the Gulf but was not thought to be caused by the spill [234].

Pesticides. The literature on concentrations of pesticides occurring in reef environments and reef organisms is patchy but does suggest that pesticides occasionally reach reef ecosystems, sometimes in high concentrations. In a baseline study of organochlorine pesticides on the Great Barrier Reef, lindane was the only compound that was consistently detected, with concentrations of 0.05 to 0.39 ng/g wet weight in coral reef organisms [235]. McCloskey and Chesher [236] found DDT at 3 to 12 ng/g wet weight and dieldrin at 0.260 to 0.320 ng/g weight wet in coral tissues off Florida. Pesticides were also found in 96.6% of the scleractinian corals and 100% of the gorgonian corals sampled off the northern Florida Keys, with concentrations up to 7.6 μg/g wet weight [237]. Chlordane was the most frequently encountered and highly concentrated pesticide. Samples of corals, lobsters, sponges, and fishes from sites further south on the Florida reef tract had lower concentrations of pesticides. A more recent study of sediment and biota samples in Pennekamp Coral Reef State Park and Key Largo National Marine Sanctuary, Florida, USA, also detected low concentrations of organochlorine pesticides in sediments and tissues of sponges, corals, crustaceans, and fishes [238]. No obvious effects on organisms or reef community damage were observed in these studies.

ECOTOXICOLOGY AND ECOLOGICAL RISK ASSESSMENT OF CHEMICAL STRESSORS IN TROPICAL MARINE ECOSYSTEMS

Research during the last 20 years has confirmed that chemical contaminants are present in the water, sediment, and biota of tropical marine ecosystems and that exposure concentrations, frequency, and duration are not unlike those found in temperate marine ecosystems. Although the potential for impacts from toxicants appears greatest for the nearshore mangrove and seagrass ecosystems, offshore shallow and even deeper coral reefs can be directly and indirectly affected by chemical exposures. What is less certain is whether fate and transport processes in this environment occur in the same manner, or at the same rates, as in temperate regions and to what extent tropical species react to equivalent exposures in the same way as temperate species. Gaps in our knowledge of these topics are greater than for temperate ecosystems. In the following sections, we identify several broad but key issues and information needs to focus further research in tropical marine ecosystems to close critical information gaps and to improve the use of assessment tools for examining and predicting potential impacts from chemical and other stressors.

What managers need to know

Protection of coastal marine ecosystems is becoming recognized as critical in many tropical and subtropical areas. Regulatory agencies are instituting new policies and regulations reflecting public concerns to reduce or restrict environmental impacts to these habitats, ranging from conservation to improving effluent quality. For chemical contaminants, the identification of point and nonpoint watershed or oceanic sources of pollutants might not be easy, and such contaminants do not recognize political boundaries. Restricting the movement of chemicals to the mangroves, seagrass meadows, and coral reefs might be impossible. Furthermore, persistent and bioaccumulative contaminants can remain in sediments and biota for years, causing long-term impacts and limiting ecosystem recovery.

Management issues. An assessment of risks to an ecosystem can be prospective, meaning it will attempt to predict problems that might occur in the future given a particular scenario (e.g., clearing a mangrove forest that has trapped contaminated sediments), or retrospective, meaning it will attempt to assess risk posed by stresses that have occurred in the past (e.g., effluent from a desalination plant released near a reef). Managers need to understand that chemical stressors might pose significant risks to tropical marine ecosystems. Existing statutes, laws, and regulations can be used to reduce impacts. For example, many human communities are diverting sewage discharges that contain potentially toxic chemicals to deep water, away from reefs. Pesticide runoff from golf courses or other areas might be controlled by requiring vegetation buffers or holding ponds near shore. Toxicants in ground water could reach reefs through the porous limestone substratum, which could require pumping and treatment strategies. Oil tankers could be restricted to offshore shipping lanes and well-marked harbor approaches to reduce the possibility of groundings and spills. Sediment loading from land clearing can cause problems not only by physically smothering sedentary organisms but also through chemical toxicity. Land management practices could be modified to minimize such effects. Dredged material management is a key issue in tropical marine ecosystems as well as in temperate ecosystems in regard to removal and disposal of contaminated sediments. In addition, the timing of critical events, like reproductive periods, could be incorporated into management plans to reduce interference with chemical cues used for synchronization and to reduce effects on particularly sensitive life-history stages.

At any particular site, managers need to examine the types of stressors that could affect the ecosystems and to identify management options or alternatives for consideration that could help them meet clearly defined goals to protect resources [as evaluated in 47]. They need to work with local citizens and governments, as well as with scientists, to understand the nature of the ecosystems and the best way to minimize environmental impacts while minimizing societal (e.g., economic) impacts [239]. This information is needed to develop a sound decision framework and to identify the basis for decisions and scientific data needs [3,240,241]. Ecological risk assessment is a tool that provides a framework for decision-making by using screening-level to sophisticated analyses to successively evaluate risks to the ecosystem over a long period of time. An example of the development of such a framework to support decisions on loss and recoverability of reefs of the Great Barrier Reef has been described by Done [242].

Conceptual models. As part of the problem formulation phase of an ecological risk assessment [15, see also 243], a conceptual model is developed to help focus the risk assessment process. The conceptual model illustrates, in words, pictures, or diagrams, how the ecosystem under consideration works and how the stressors are affecting or might affect the components of the natural environment. Ecological components or features of valued resources that are considered important, also known as ‘‘assessment endpoints’’ [15,16], are selected on a site-specific basis. These include scientific, cultural, and policy considerations. The interactions of the biotic and abiotic components and the energy requirements and flows that occur in the ecosystem must also be considered because any change that occurs to even a single component can alter established relationships and, hence, the ecosystem. Measurement endpoints are measurable responses to a stressor, such as concentrations of chemicals, that are related to the valued characteristics selected as the assessment endpoint, for example, fish population size and condition, areal coverage of mangroves or seagrasses, or coral reef community composition. These endpoints must be pertinent to the decisions that might be made to protect the environment. The preliminary, qualitative analysis of the ecosystem, the stressor characteristics and potential or actual ecological effects within the ecosystem, are used to identify possible exposure scenarios for the assessment.

Figure 2 presents a generic conceptual model of actual or potential effects of chemical stressors on tropical marine ecosystems, including the principal ecological components and biotic and abiotic processes. For any particular site, or particular group of chemical stressors such as metals, different aspects of this conceptual model would require modification or emphasis. Site-specific exposure scenarios need to be developed, including consideration of direct and indirect exposures for plants, invertebrates, and vertebrates, as well as factors that could modify the amount and toxicity of the chemicals, including adsorption, photolysis or photoactivation, temperature, and microbial alteration.

For example, the uptake of heavy metals by fishes, primarily through gill and intestine, depends on food choice, metabolic rate, and bioavailability of the metals [243]. In mangrove forests, metals and highly hydrophobic organics might adsorb onto sediment particles and not be bioavailable and directly toxic to fishes, but they could be ingested by worms living in the sediment, which might then be ingested by fishes or birds. In seagrass meadows, exposure of fishes to pollutants can occur

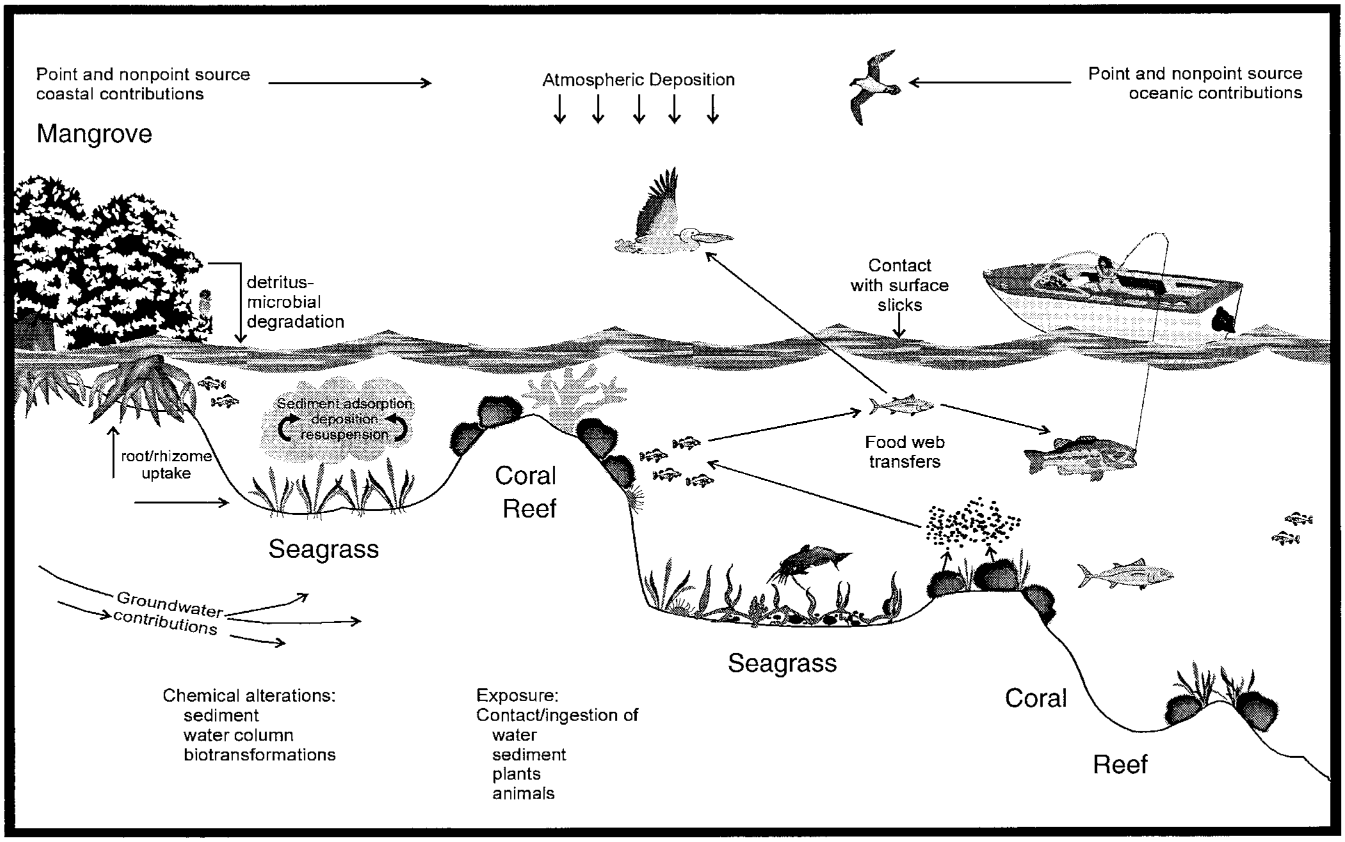

Fig. 2. Simple conceptual model of sources of chemical contaminants and potential exposure pathways in tropical marine ecosystems. Fate and transport varies with different chemicals and should be examined on a site-specific basis.

through passage across the gills, by uptake during seawater ingestion, through the food chain, or by exposure through contact with contaminated sediments. Uptake of polycyclic aromatic hydrocarbons from the water column by the warm-water benthic toadfish *Opsanus beta* is concentration-dependent and temperature-sensitive, with greater uptake occurring at higher temperatures or during acute increases in ambient temperature due to changes in respiratory rate [244]. In seagrass meadows and reefs, where sand has low organic carbon content and highly hydrophobic chemicals are readily absorbed by coral tissue and not greatly metabolized [212,215,216], these chemicals can be transferred to grazing herbivorous fishes and sea urchins or other organisms. Suspected direct and indirect contacts with chemical stressors could then be targeted to evaluate ecological effects of the chemical stressors, using literature values, laboratory and field toxicity tests, histopathological examinations, and species abundance surveys (measurement endpoints).

Linkages between assessment and measurement endpoints and policy goals need to be clearly identified for each exposure scenario. The value of organizing this information in one or more conceptual models lies in developing and refining a series of testable hypotheses about how a particular stressor might affect ecological components of concern so that analyses conducted during the assessment can establish, to the extent possible, a cause-and-effect relationship [15]. One example of such a statement is ''Pesticide *X* causes 30% or greater reduction in coral cover.'' The approach used and the types of data and analytical tools that are needed to analyze impacts and risks are based on information contained in the conceptual model. The problem formulation phase of the assessment is critical to its success and must be conducted carefully; the abundance of species, complexities of these ecosystems, and limited funding resources could quickly lead the assessment astray, leaving managers in confusion instead of helping them evaluate management options.

Selection of appropriate assessment and measurement endpoints. The valued ecological resources or assessment endpoints should be susceptible to the stressor in question and should also reflect policy goals and societal values. In our review we identified the major structural components of each of these ecosystems (mangrove trees, seagrasses, scleractinian corals) and fishes as potential assessment endpoints [15,16]. However, other biotic components need to be considered in assessments of these ecosystems, such as those encrusting mangrove roots (algae, sponges, bivalve molluscs), soft-bottom fauna (polychaetes, bivalves, burrrowing crustaceans, holothurians), and hard-bottom species, such as coralline algae, sponges, or sea urchins, that could be very sensitive to chemical stressors and might be easier to study than the structural biota. In addition, sea turtles, marine mammals and seabirds, and endangered or threatened species (e.g., osprey, manatee) are important assessment endpoints. Keystone species are those known to control the abundance and distribution of many other species in the community; the basic ecology of the tropical ecosystem should be considered at the conceptual model stage. Thus, although scleractinian corals have economic and aesthetic value, species less valued by humans but crucial for the survival and health of reefs [6], such as herbivorous fishes or long-spined sea urchins, might be useful indicator species. Effects can occur at every level of biological organization, so a variety of indicator species and appropriate biomarkers might give advance warning of the bioavailability of the contaminants and potential effects prior to the loss of significant habitat [245]. For example, amphipods, tanaids, and echinoderms (holothurians, ophiuroids, echinoids) were most affected by the Bahia Las Minas oil spill and might be appropriate sentinel species to assess pollution [118].

For each of these ecosystems, a suite of assessment and measurement endpoints that addresses potential population and community impacts might be needed to estimate the risks to the ecosystem. If possible, ecological endpoints that measure the characteristics of ecosystem sustainability and energy flow, such as physical structure of coral reefs, seagrass or fish productivity, and alterations in trophic structure, should also be included [246]. Different species vary in their susceptibilities to stressors, and different life stages of organisms also exhibit different susceptibilities. Because no single measurement can describe the effects of a stressor on an ecosystem, multiple endpoints need to be evaluated and must be selected on a case-by-case basis to develop successful, integrative assessments of chemical contamination [247]. Both Ballou et al. [47] and Keller and Jackson [248] assessed several types of measurement endpoints for each assessment endpoint to examine lethal and sublethal effects of oil on mangroves, seagrass meadows, and coral reefs, which provided important information to link cause and effect and estimate recovery rates for the affected ecosystems.

Evaluation of chemical stressors

Extensive efforts have been directed toward monitoring of coral reefs and adjacent marine ecosystems, but most monitoring programs quantify ecological responses and have only limited quantification of stressors. Rapid ecological assessments, such as that conducted at Palau [249], or comprehensive monitoring programs, such as that conducted for Jamaican coral reefs [6], provide a basis for quantifying trends in coral cover and diversity, as well as identifying possible stressors and evaluating the role of rare events such as hurricanes. However, few programs collect data on chemical contaminants in addition to assessing such biological parameters as cover, species diversity and evenness, and recruitment. A thorough examination of the uncertainties and limitations associated with these methodologies is also required to help improve assessments of chemical exposure and effects. Numerous issues still remain for developing appropriate guidance to evaluate ecosystem responses to stressors in the well-studied temperate regions of the world [reviewed in 16,23,246,250,251], and the literature should be consulted for the latest methodological details. This section highlights only a few of the many specific needs to collect data for assessing the condition of and risks to tropical marine ecosystems.

Exposure analyses. These analyses focus on determining the sources, pathways of exposure, fates, and concentrations of potentially toxic chemicals in an ecosystem, as well as identifying the populations of organisms most likely to be adversely affected by exposure to a particular concentration. Measured or estimated concentrations of chemical stressors in water and sediment [16] and the duration and frequency of exposure observed in the ecosystem are the primary data on which the assessment is based. Levels measured in biota provide important information on the form of the chemical (soluble or adsorbed onto particles), whether, and to what extent, the contaminant can be taken up by living organisms (its bioavailability), and the ability of the organism to bioconcentrate or biomagnify the contaminant. In addition, metals and hydrophobic organics can bioaccumulate in tissues and can be transferred to young (via yolk or seedling reserves) and to organisms at higher trophic levels in the ecosystem food web, so metals and lipophilic contaminant concentrations should also be measured or modeled in organisms from two or more trophic levels. This information is used to identify potential contaminants of concern in the ecosystem.

Our knowledge of the concentrations at which particular adverse effects are observed in the field is limited. Most studies failed to measure contaminant levels in water, sediment, and organisms at the same time and from the same locations [but see 252]. For example, pesticides and metals were measured in corals but not in water or sediments [237]; pesticides were measured in sediment and organisms [238]; *n*-alkanes were measured in sediments, fishes, crustaceans, corals, and molluscs [253]; hydrocarbons in dissolved and particulate phases were measured in a mangrove lagoon [254]; metals were measured in sediments [255]; and metals and pesticides were measured in oysters [256]. Johannes [219] noted that determination of pollutants in the water column coupled with observations on the plankton at Kanehoe Bay missed the impacts occurring on the reefs, where pollutants settled to the bottom and sorbed to the sediment and the impact of pollution was more profound in bottom communities. Media on reefs [207,209,257] and seagrass beds [141] were not analyzed for oil contamination, even though there was evidence that oil might be a significant, although not the only, stressor at these sites and that effects observed might be concentration-dependent. Notable exceptions to this were the studies conducted by Ballou et al. [47], Getter et al. [55], and Keller and Jackson [248], in which ecological effects of measured concentrations of petroleum hydrocarbons and dispersants were studied following application in experimental enclosures and accidental spills, respectively. Water samples, mangrove leaves, seagrasses, and oysters were analyzed for up to 20 months after the spill in one study [47]; water, sediment, and oiled producers and consumers were analyzed in another [55]; and petroleum hydrocarbons in surface sediments, coral tissues, and bivalve molluscs were measured in the third study [248]. The feasibility of measuring metal loads in samples of sediment and coral tissue [258,259] and echinoderms [260] and of measuring organochlorines in coral eggs [261] has been investigated.

The types and number of contaminants examined have also been limited compared to studies conducted in temperate regions. Polychlorinated biphenyls were detected in mangrove organisms in the Caribbean in the early 1970s [36,262] but have not been measured more recently, except in oysters in Hawaii [256], and dioxins have not been examined at all, even though atmospheric contributions might be significant in some areas [263]. Standardization of chemical analytical methodologies lags behind that for studies conducted in temperate freshwater and marine ecosystems. The high costs of chemical analyses and lack of analytical facilities are often cited as the reasons that they are not performed. Computer models have been developed to provide estimates of environmental concentrations of chemicals in temperate ecosystems. However, unless appropriate chemical analyses are performed for water, sediment, and tissues of organisms, it will be impossible to develop concentration–response relationships for tropical marine organisms or to examine differences in fate and transport, biodegradation rates, biotransformations, or bioaccumulation [23]. Invertebrates and fishes feeding on contaminated coral tissue can accumulate these compounds, but we have no idea how rapidly they might be metabolized, how easily they might be passed on to larger fishes and marine birds, and what toxicological impacts might occur.

In the 1980s extensive research was focused on developing reliable biomarkers of exposure [264] that could be used to identify populations and communities exposed to various contaminants, and several studies examined the utility of these biomarkers for tropical marine organisms (Table 3). Hogstrand and Haux [200] demonstrated that the concentration of hepatic metallothionien (MT), a protein that binds strongly to heavy metals, shows a dose–response curve following intraperitoneal injection of cadmium chloride. The concentration of MT has been shown to increase with environmental exposure in two reef species, substantiating its usefulness as an indicator of heavy metal exposure [200]. However, there can be variability in MT concentrations between species and among individuals within species; handling stress, salinity changes, and reproductive status significantly increased the amount of zinc associated with MT in estuarine teleosts [265]. Therefore, care must be taken in the interpretation of elevated MT levels.

The cytochrome P450 mixed function oxidases located in the endoplasmic reticulum of the liver and other tissues are responsible for the metabolism of lipophilic xenobiotics and endogenous compounds such as steroids and prostaglandins. Exposure to pollutants induces increased activity of these enzymes and might allow some species to detoxify pollutants. The P450 response to organics has been described for only a few warm-water species [217,266–271]. Enzymatic induction of cytochrome P450 in fish increased with greater proximity to the areas most affected by the 1991 oil spill in the Arabian Gulf during the Gulf War [272], demonstrating that mixed function oxidase activity can be used in tropical systems as an indicator of hydrocarbon exposure. Conversely, in some tropical fish, elevated enzyme levels might be indicative of allelochemical, not contaminant, exposure [271]. Temperature or seasonal changes in temperature are also important factors to consider when using xenobiotic-metabolizing enzymes as indicators of pollution exposure, even in the tropics. The Caribbean coral *Favia fragum* had very little cytochrome P450 [216], supporting observations that the rate of xenobiotic detoxification or elimination in corals might be slow [213,217,237,264,273,274]. In some corals, the activity of glutathione-*S*-transferase (GST), an enzyme that conjugates endogenous molecules to substrates to increase hydrophilicity, exceeded that found in most marine invertebrates [216,275–277] and some fish species [278,279] but was not as high as in some species of crabs and mussels [280]. Additional work will be needed to determine the most appropriate techniques for use and interpretation of these biomarkers and others, such as stress protein synthesis, in tropical marine organisms [281].

For reef corals a suite of behavioral effects has been used as an indicator of exposure to contaminants (e.g., polyp expansion, mesenterial filament extrusion, mucus production), as well as tissue color change related to loss of symbiotic algae, but there is often great variability between and within species. Furthermore, these parameters, as well as mortality in adult corals, cannot be as easily quantified for corals in laboratory toxicity tests as for other organisms [148]. Once the symbiont association breaks down and ''bleaching'' occurs (the white exoskeleton of the coral shows through the translucent tissue when the symbiotic algae are expelled or the algal pigments are destroyed), some corals die, while others show signs of stress until algal populations are recovered (reviewed in special coral bleaching issue of *Coral Reefs*, B.E. Brown, ed., vol. 9, no. 3, 1990). Large-scale bleaching events have been attributed to temperature-induced stress and increased exposure to ultraviolet radiation related to El Niño warming events and other sea surface temperature anomalies [282,283]. Pollutants appear to be responsible for more localized bleaching events; elevated levels of iron resulted in the loss of zooxanthellae in *P. lutea* [156]. The pesticide atrazine is stable enough in seawater to permit exposure of susceptible marine life, and the chemical's role as a photosynthetic inhibitor could affect the coral–algal symbiosis [284]. Few studies have been performed to determine what levels of environmental contamination are sufficient to cause localized bleaching events. Such studies are needed to address the effects of herbicides, in particular when used in agriculture and on golf courses adjacent to coral reefs.

Toxicological endpoints that indicate impaired function of the coral–algal symbiosis, such as photosynthesis and loss of symbiotic algae or algal pigments, are needed. Lang et al. [285] have quantified digitized photographic images of the reef corals *P. astreoides* and *Montastraea* spp. to create color indices, although they noted that there is considerable individual variability as well as variability in color brightness values at different seasons of the year. In situ measurements of photosynthesis and coral growth rates have also been attempted, but techniques used thus far involve handling coral colonies, which can introduce additional stress [224,286]. Other research is under way to develop more accurate noninvasive measurements of photosynthesis that are based on natural fluorescence of chlorophyll, spectral fluorescence, and spectral reflectance, which could also be linked to coral growth [287,288]. The data need to be quantitative to be converted to chronic criteria or to be used in simulation models for predicting ecosystem risks.

Temporal scales in the detection of exposure and resulting impacts can vary widely: changes in seagrass productivity could be measured in a few hours, whereas induction of mutations by polycyclic aromatic hydrocarbons (PAH) resulting in visibly manifested chlorophyll deficiencies in mangroves [72] might require more than 10 years to detect. Fate-and-transport models are increasingly being applied in coastal marine studies to determine potential inputs from chemical and physical stressors and, with modifications, might prove useful for tropical coastal ecosystems. A few attempts have been made to develop population, food web, and other simulation models in tropical marine ecosystems. Several studies have examined the ECOPATH model for coral reefs, a top-down (top carnivores to primary producers) food chain model that can provide estimates of mean annual biomass, annual biomass production, and annual biomass consumption for each species group (species having common habitat and similar diet and life-history characteristics) and can be tested by providing an independent measure of primary productivity [289,290]. Appropriate models could be applied to describe and test site-specific patterns of exposure and bioaccumulation by sensitive species or life stages to chemical or other stressors occurring in these ecosystems. The models could examine differences in pharmacokinetics between tropical and temperate organisms and the potential for toxic effects at higher trophic levels, as well as changes in biomass and trophic flows resulting from impacts at lower trophic levels.

Developing residue–effect relationships can link information from fate-and-transport models to food chain accumulation and

Table 3. Biomarkers in tropical marine organisms[a]

Biomarker	Species	Results	Considerations	Reference
Metallothionein	Squirrelfish *Holocentrus rufus*, blue-striped grunt *Haemulon sciurus*	Correlated dose–response to intraperitoneal injections of CdCl; MT increased with environmental exposure to metals	Squirrelfish had MT values 2 orders of magnitude above those of grunts; Concns. of Cd, Zn, and Cu in squirrelfish (1.0, 2,630, and 231 μg/g liver, respectively) were 10 times greater than those in grunts at unpolluted sites; Zn concns. in grunt were 20–30 μg/g liver at clean sites and 60–70 μg/g liver at sites with 5 times the level of heavy metals	[200,268]
Cytochrome P450	Blue-striped grunt *Haemulon sciurus*	Peak induction of mixed function oxidase system seen 3 d after injection with >1 mg/kg PAHs; returned to control level in 10 d	Dose–response relationship to amount of PAHs injected	[269]
	Squirrelfish *Holocentrus rufus*	Little induction observed following injection with PAHs	Possibly more susceptible to pollution	[269]
	Gulf toadfish *Opsanus beta*	Enzyme induction greater during summer months in field-collected fish	Enzyme activity increases with temperature	(N.J. Gassman et al., unpublished data)
	Warm-water fishes	Activity of enzymes and formation of DNA adducts increased when exposed to xenobiotics at higher temperatures	Enzyme activity increases with temperature	[270,363]
	Coral *Favia fragum*	Little P450 detected	Corals may not be capable of metabolizing PAHs	[216]
	Star coral *Montastrea annularis*	Cytochrome P450 and EROD activity not detected after intermittent exposure to 15 μg/L chlordane for 90 d		[364]
Glutathione-*S*-transferase	Star coral *Montastrea annularis*	GST activity significantly increased by a factor of 2.6 after intermittent exposure to 15 μg/L chlordane for 90 d	Greater GST activity induction than reported for freshwater mussels [365], not as great as that seen in crabs and shrimp [366]	[364]
Bleaching in corals	Star coral *Montastrea annularis*	3-month exposure to 1 to 15 μg/L chlordane caused rapid and significant depressions of respiration, photosynthesis, density of the algal symbionts, and chlorophyll; corals exposed to 1 μg/L chlordane attained normal rates of photosynthesis after 2 weeks in clean seawater; respiration rates returned to normal levels, but chlorophyll concns. were still significantly depressed after 5 weeks of recovery	Laboratory exposures to pesticides can have devastating effects on corals at very low concns.	[315]
Mucus production	Coral *Pocillopora damicornis*	At lowest concns. of 2,4-D and 2,4,5-T administered (100 μg/L), specimens produced copious amounts of mucus and died within 24 h	Lamberts [367] found no evidence of injury in corals exposed to concns. of 10 μg/L to 2 mg/L of 2,4-D for 24 h	[368]

[a] EROD = ethoxyresorufin-*O*-deethylase; GST = glutathione-*S*-transferase; MT = metallothionein; PAH = polycyclic aromatic hydrocarbon; 2,4-D = 2,4-dichlorophenoxyacetic acid; 2,4,5-T = trichlorophenoxyacetic acid.

acute and chronic effects determined from toxicity tests and bioassays, potentially allowing predictive modeling of population impacts [23, 250,291–294]. Models have limitations in the number of contaminants and routes of exposure that can be examined, but they could be useful in identifying important routes of exposure and potential impacts when coupled with appropriate chemical analyses of water, sediment, and biota. Rogers et al. [295] provide methods to collect data on basic water quality and on current speed and direction, which are also needed to analyze exposure in tropical marine ecosystems.

Ecological response analyses. Monitoring of ecological effects in tropical marine ecosystems and what methods are most appropriate for particular habitats or biota (e.g., forereef vs. backreef, mangrove mud or seagrass meadow vs. rubble field) have been intensely debated at numerous meetings and in the literature. Ecological effects should be examined at several levels, from individual to population to community, within any ecosystem [23,246]. Ecological effects must also be examined on appropriate spatial and temporal scales and with respect to the ability of the ecosystem to recover from observed damage. Seasonal variability occurs in the tropics as well as in temperate regions; changes in temperature, rainfall, storms, tidal ranges, current patterns, and insolation are observed, and these natural disturbances and their impacts on the ecosystems need to be analyzed. Again, biological and physical data need to be collected at the same time as data on chemical or other stressors to determine the correlation between exposure and effects.

At the individual level, effects of chemicals on physiology, biochemistry, reproduction, and pathology have been documented [see reviews in 23,296,297, this paper] and will be reflected in responses at the population and community levels. In Biscayne Bay, Florida, USA, a subtropical estuarine ecosystem, abnormalities and increased incidence of diseases have been documented in several fish species over the last 20 years and have been associated with exposure to sewage, petroleum products, heavy metals, and industrial chemicals [298–300]. Although pathological responses can be difficult to interpret as indicators of ecological effects, there is increasing evidence that cytochrome P450 (CYP1A1) induction is associated with liver neoplasms and exposure to PAHs; the distribution of abnormalities in Biscayne Bay has been correlated with sediment hydrocarbon concentrations [300]. Few studies of tropical ecosystems have examined the toxicological pathology of fishes and other organisms in mangrove forests, seagrass beds, or coral reefs, but this area of research provides important tools for linking biomarkers of exposure to ultimate effects within an organism, population, or community, as well as for comparing responses in the field with laboratory exposures to chemicals using appropriate condition indices.

The collection of information on populations and communities using quantitative techniques is important for evaluating stressor impacts [23,301]. Metrics include species composition, presence or absence of sensitive and rare or endangered species, species richness and evenness or diversity, relative abundances, dominance, resemblance indices, population biomass and age, community trophic structure, and productivity. Ballou et al. [47] examined several parameters of survival, growth, and reproduction in mangroves, seagrass beds, and corals that were experimentally oiled and exposed to dispersant. A variety of methods have been used. For sedentary soft- and hard-bottom organisms, chain or line transect and random or belt quadrat methods all have proponents and opponents, and ease of data collection must be weighed against information obtained and amount of time spent under water. In addition, the patchy distribution of fauna, comparability of sites, and hypotheses to be examined must be considered. More recently, photography and videography have been adapted for use in coastal areas, including digitizing aerial photographs and incorporating these into geographic information system (GIS) analyses and conducting high-resolution photography and videotaping along transects of coral reefs [285,302–305]. Biological monitoring methods for reefs in the tropical western Atlantic and Caribbean are described by Rogers et al. [295].

Despite numerous attempts, however, no specific criteria have been established as to what constitutes optimum conditions in tropical marine ecosystems, and baseline data are often not available. As in other ecosystems, risk assessors need to look for relative changes at a particular site over time, or differences between similar habitats in which a gradient of exposure to chemical stressors is apparent. Furthermore, several different metrics should be combined to more appropriately describe communities (species presence or absence, diversity, abundance, competition, trophic structure), but little work has been done on combining metrics for tropical marine ecosystems.

It is also important to describe and quantify the fish communities that might be affected by chemical contaminants, and more work needs to be done to determine appropriate methods for conducting fish censuses. For example, even in the extensive long-term study of the Bahia las Minas spill, the slow recovery of fish communities from oiled seagrass beds was described only by density of fish caught by push net, not species composition of the total community [118]. Shifts in community structure of herbivores favoring damselfish were documented on oiled versus unoiled reefs [226], potentially identifying a reef species tolerant of oil exposure or one that can rapidly colonize and exploit resources abandoned by other species during the early stages of oiling. Following the Gulf War, reef fish populations on oiled reefs appeared to be healthy. Comparison of long-term data revealed both increases and decreases in species composition and densities following oiling [306]. Could some of these fish species have developed PAH-induced neoplasms that are now affecting the population and community structure? A broader view is needed to understand and quantify impacts on and recovery of fish communities exposed to oil and other chemicals.

Another important analytical tool for evaluating the potential for chemical impacts is the bioassay, particularly because the toxicity or hazard potential of chemicals can vary with environmental conditions. The phrase ‘‘below detectable limits’’ is commonly used to report levels of environmental pollutants, yet this finding can often result from technological limitations or use of inappropriate protocols to identify and quantify compounds rather than the presence of levels below those that affect tropical marine organisms. Bioassays that take advantage of organisms’ responses to detect toxics are clearly needed. Testing procedures for acute and chronic toxicity of water and sediment samples using temperate marine indicator species have been developed and standardized by the U.S. Environmental Protection Agency, the American Society for Testing and Materials, and others [e.g., 23,307–309]. However, we have little knowledge of how well the sensitivities of these species compare to those of tropical marine species or how well they might test under tropical conditions. Most chemicals, including pesticides and inorganics, elicit an increase or decrease in LC50 in exposed temperate species by a factor of 2 to 4 per 10°C change, conforming to the Q_{10} concept [310]. Tropical and subtropical fauna appear to be at least as sensitive to the effects of toxic chemicals as are temperate-and cold-water species [133], but these species could be living at their upper limit of temperature, which might affect their responses to toxics [1].

Work is needed to determine suitable tropical test species (amphipods, bivalve larvae, echinoderm embryos, coral em-

Table 4. Summary of toxicity tests using coral embryos and larvae

Species, life stage	Type of test	Results	Considerations	Reference
Goniastrea aspera, Favites chinensis, Platygyra ryukuensis, eggs and sperm	Static, fertilization	Zn at 1.0 mg/L reduced fertilization to 6.3%; threshold for fertilization inhibition, 0.01–0.1 mg/L		[369]
Diploria strigosa, newly fertilized embryos	96-h static acute toxicity	Larval survival and inhibition of development affected by exposure to sodium dodecyl sulfate	Sensitivity comparable to the sea urchin fertilization and embryological development acute toxicity tests; coral larvae may be very sensitive to chemical contaminants	[370]
Pocillopora damicornis, planula larvae	96-h static acute toxicity	LC50 for Cu, 0.067 mg/L	At 0.01–0.1 mg/L Cu, symbiotic algae were expelled, and mucus was produced	[184]
Pocillopora damicornis, planula larvae	Static acute toxicity	12-h LC50 for nickel, 9 mg/L	Larval settlement reduced after exposure to 1 mg/L nickel for 12 h	[371]
Pocillopora damicornis, planula larvae	Static acute toxicity	Effects of pesticides (carbaryl, 1-naphthol, chlorpyrifos), metals, and petroleum products (benzene, gasoline/oil mixture) were examined; neither LC50s nor EC50s were calculated	Preliminary tests, used different procedures, exposure periods, and endpoints	[372,373]

bryos or larvae, symbiotic algae) that are or relate to potential impacts on keystone species as well as appropriate test conditions [311–313]. For example, the scleractinian coral species *M. annularis* and *D. strigosa* have been used most frequently in laboratory tests of stressors; they are among the major reef framework builders in the Caribbean, but they may not be the most sensitive species, depending on the chemical contaminant to which they are exposed. *Acropora* spp. are very susceptible to adverse changes in water quality and should be more sensitive to potentially toxic chemicals, but they require strong currents and rapid water exchange and are difficult to maintain in aquaria, as are many other coral species [148,313]. Specially prepared, uniformly sized ''nubbins'' and explants or cores of scleractinian corals provide more suitable material for field and laboratory studies than large, irregularly shaped whole colonies or portions [313–315].

The use of surrogate species for those actually found in the ecosystem assumes that chemical exposure in the laboratory can be related to species found in nature and that surrogate organisms are representative of the ecosystem. A factor can be added to allow a margin of safety when these results are extrapolated to the organisms in the ecosystem under investigation [148]. Nganro [316] observed that the zooxanthellate anemone *A. viridis* was as sensitive to copper exposure as corals, coral gametes, and larvae and that it might serve as a useful surrogate test species. However, this approach will not work well in complex ecosystems or with chemicals that cause persistent ecological effects at low levels [17].

Bioassays should also be designed to study the effects of pollutants on processes that are chemically mediated at the community level and not just the organismal level. Studies of the effects of chemical contaminants on embryos and larvae of fish (again, mostly freshwater and/or temperate) indicate that most are more sensitive than adults, and further decreases in populations over natural mortality, not only as a result of chemically mediated mortalities but also teratogenic effects and adverse effects on hatching and growth, can be highly detrimental [317]. Sublethal effects can occur in corals and other reef organisms exposed to minute chemical concentrations that affect chemical signals between obligate algal symbionts and coral hosts, can interfere with chemical cues used to synchronize reproductive activity among individuals within a population, and can disrupt settlement cues and metamorphic inducers, affecting recruitment processes and ultimately affecting key species and communities [9,318,319].

Coral gametes and larvae provide suitable tools for studying coral reef ecotoxicology since our present knowledge of reproductive timing and subsequent recruitment processes in corals is substantial. In addition, the numbers of gametes and larvae that can be produced and used in bioassays is large enough to allow good statistical analyses, a problem that often arises when trying to observe both lethal and sublethal effects in colonial organisms. Many of the coral species studied in the Indo-West Pacific and tropical Western Atlantic spawn once a year [320,321], often during the rainy season, when coastal contaminant concentrations, particularly in the 50-μm-thick sea surface microlayer, would be expected to peak. Considering that most coral eggs are buoyant, floating in the surface water layer for up to several hours before fertilization occurs, terrigenous runoff and coastal pollution have the potential to produce reproductive failure of spawning reef species. Gametes of *S. siderea* were released at the Bahia Las Minas oil spill site during the rainy season, when oil slicks were common [248]. Examples of toxicity tests conducted with coral gametes, embryos, and larvae are presented in Table 4.

With respect to recruitment processes, many species of benthic marine invertebrates have larvae that respond to specific metamorphic inducers [322,323]. Morse and Morse [324] found several species of corals were highly selective in choosing settling substrata, reacting only to certain species of crustose coralline algae. Metamorphic inducers can be small molecules and might be effective in concentrations below 10^{-10} M [323]. Pollutants in the water column at levels below those lethal to adult organisms, below detectable limits by high-performance liquid chromatography, or bound up in substrata where they are not identified by routine monitoring protocols,

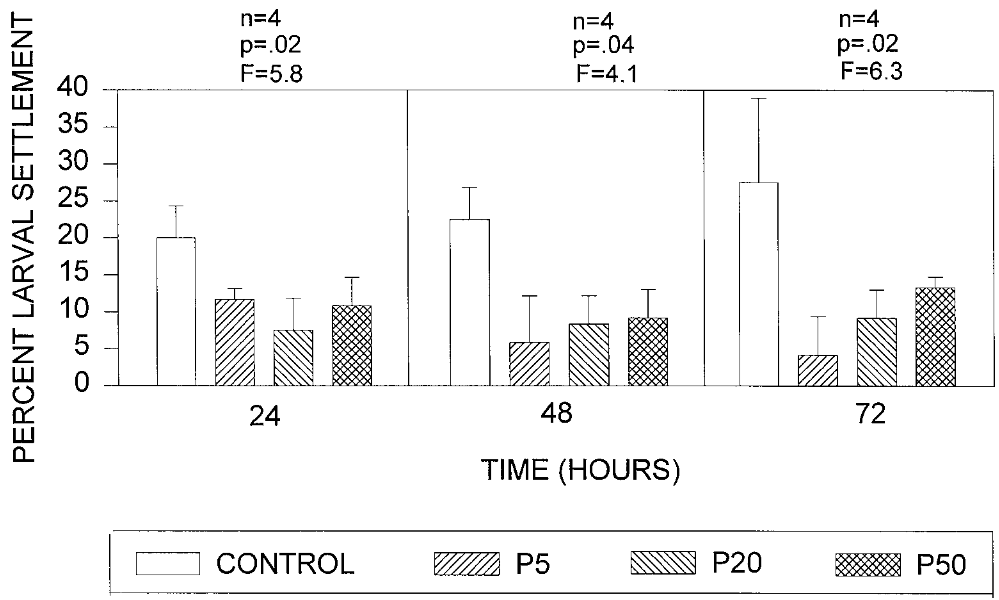

Fig. 3. Effects of chlorpyrifos on coral larval settlement. Percent of coral planula larvae that settled on coralline algae substratum incubated for 12 h in filtered seawater containing 0, 5, 20, and 50 ppb chlorpyrifos, after 24, 48, and 72 h (cumulative settlement percentages). Algal substrata were rinsed with filtered seawater, then placed in 200 ml of untreated filtered seawater for the settlement trials. Two replicates (40 larvae per replicate) were conducted for each of the four treatments.

can have a negative effect on recruitment processes. However, this type of sublethal effect can be more difficult to discern since the time frame is longer (i.e., months to detect settled, growing colonies). Coral larvae exposed to chemically treated substrata rather than directly to chlorpyrifos [325] exhibited significantly lower levels of settlement and metamorphosis than the controls for each of the 3 days of exposure (Fig. 3) (R. Richmond, D. Crosby, S. Leota, and H. Wood, manuscript in preparation). Chlorpyrifos was recovered from coralline algal substrata extracted after initial treatment in the chlorpyrifos solutions. These results were consistent with a previous study that demonstrated that the pesticide *p*-nitroanisole was taken up by the crustose coralline alga *Porolithon* sp. [326]. Many pesticides are hydrophobic, and seawater analyses might not detect these substances even though they are present in the reef environment in other media. The fate of chemical contaminants, including half-lives, breakdown rates, and breakdown products, is important to our understanding of the effects of pollutants on coral reefs.

In addition to concerns about bioassay test conditions, such as water quality and feeding, exposure to ultraviolet light can increase the toxicity of PAHs to benthic invertebrates placed in contaminated sediments [312,327] and planktonic fishes, crustaceans, and algal species exposed to PAHs in the water column [328]. Because organisms in tropical marine ecosystems could be exposed to ultraviolet irradiation [329], PAHs accumulated and not metabolized by these organisms could become more toxic through photoactivation. Bioassays need to be designed to account for a variety of factors that could affect the toxicity of chemical contaminants under field conditions.

Multiple-species tests and microcosm and mesocosm studies composed of several species of marine plants and animals should prove useful to examine effects of toxicants on ecosystem processes such as food chain and food web bioaccumulation and impacts, primary and secondary productivity and decomposition rates, nutrient cycling, and pollutant degradation, among other factors [127–129,330,331]. Controlled experimental manipulations of whole ecosystems, such as those performed by Ballou et al. [47] to study the effects of petroleum hydrocarbons on tropical marine ecosystems under field conditions, can provide a more realistic view and provide direction for extrapolation from laboratory tests to ecosystems, detecting, for example, which fishes might avoid contaminants, how metal methylation by microbes in situ increases toxicity, or which coral species are most susceptible. However, another important tool used in temperate toxicity evaluations is in situ (cage) testing using several benthic and epibenthic species and/or water column species that represent different trophic levels at the study site [297]. Cage studies could be used in mangrove channels or over seagrass beds; for reefs, small colonies of adult corals, such as those developed for physiology studies, could be cemented to inert frames and placed on the reef for observation [148,219,286].

In any case, it will be important to translate bioassay water or sediment concentrations to reflect toxicokinetics and toxicodynamics (uptake and toxic response) in the effects measured in the bioassay. Measurements of whole-body residues to determine concentrations of chemicals taken up by biota in site-specific bioassays (critical body residue method) can provide additional information on bioaccumulation and potential food web effects [294].

Risk characterization for tropical marine ecosystems

To characterize risks to an ecosystem, correlations must be developed between observed adverse ecological effects and the stressor. Comparisons might be made between adverse effects observed in the exposed ecosystem and the condition of a reference site, between toxicity test results and observed adverse ecological effects, or between expected environmental concentrations and criteria and standards. Thus, we need to develop stressor–response profiles based on measured or modeled chemical concentrations plotted against observed adverse effects in potentially exposed sites and reference sites to es-

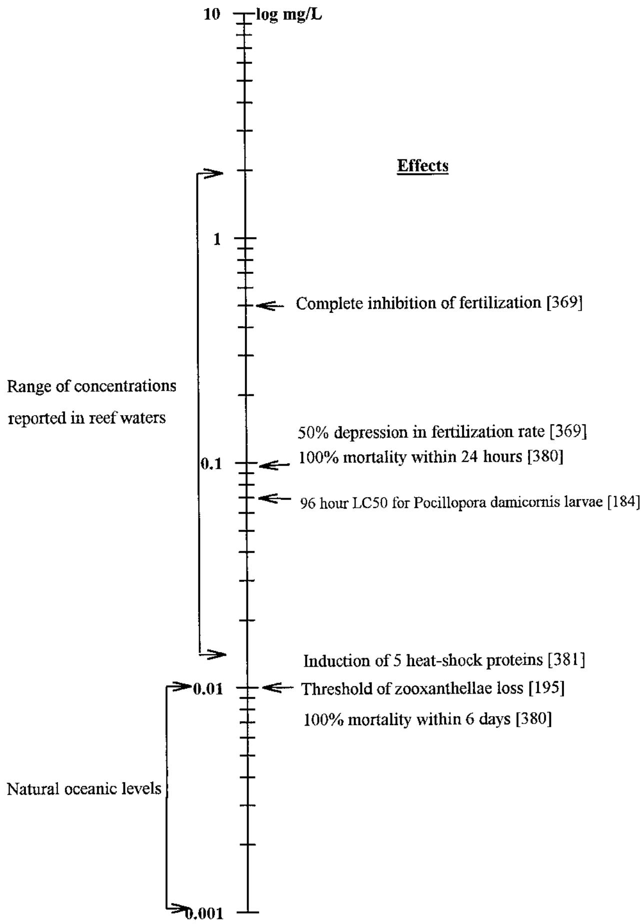

Fig. 4. Effects of copper on corals. References from which this information was compiled are cited in brackets.

timate risk to an ecosystem and for comparisons with stressor–response profiles in which toxicity has previously been documented. Examples of exposure–response profiles are presented in Figures 4 and 5, but few studies have provided these types of correlations [e.g., 72,259,316]. Keller and Jackson [248] noted concentration–response correlations of amount of oiling with coral tissue injury and growth rate, mangrove leaf longevity and biomass, and proportions of dead mangrove roots.

The risk assessment process initially relies on comparisons of measured or modeled concentrations of contaminants in water, sediment, and biota to pertinent qualitative and quantitative threshold values or benchmarks; contaminant concentrations below the benchmark should, with some degree of confidence, not result in adverse effects [16]. Benchmark comparisons are used as a screening tool to identify contaminants of potential concern and to assist in characterization of risks following the analysis of exposure and effects data. Such benchmarks are usually developed from the results of carefully controlled laboratory chemical analyses and toxicity tests to provide a standard for estimating the magnitude and probability of potential dangers. However, benchmarks cannot account for synergistic or interactive effects of multiple chemical stressors and are most useful for prioritizing contaminants for further evaluation. In the final stages of the risk assessment process, benchmarks that evaluate risks to particular assessment toxicological endpoints can be derived from statistical evaluations of site-specific bioassays and mathematical simulation models, linking observed exposures with stressor–response profiles [291].

The U.S. Environmental Protection Agency has developed saltwater ambient water quality criteria (acute, maximum criterion concentration; chronic, criterion continuous concentration) for the protection of aquatic life for approx. 30 chemicals and is finalizing sediment quality criteria for five chemicals based on the equilibrium partitioning approach [332,333]. These benchmarks were derived from toxicity test results using temperate organisms, as were the ''Apparent Effects Thresholds'' developed for Puget Sound [334] and the ''Effects Range–Low'' and ''Effects Range–Median'' values reported by Long et al. [335] for sediment contaminants. We are unsure about how protective of tropical marine species these criteria might be. For example, the toxicity of organics such as carbophenothion, chlorpyrifos, and fenvalerate varies widely between different estuarine fish species. The subtropical benthic gulf toadfish, *Opsanus beta*, has a 96-h LC50 one to three orders of magnitude higher than that of *Menidia* spp. [336], suggesting that some warm-water estuarine species might be relatively tolerant of chemicals in their environment. Ninety-six-hour LC50s for two tropical Australian fish exposed to different metals were influenced by exposure, salinity, and life stage, but not temperature [201]. For the mangrove-sediment-dwelling fish *Rivulus marmoratus*, the 96-h LC50 for cadmium at 14 ppt salinity was similar to LC50s found for zinc and copper using *Fundulus heteroclitus* [337].

Clearly, much work is needed in this area. Development of toxicity tests for tropical species is under way at a number of locations around the world. Scientists at the University of Miami have been testing tropical species of sea urchins and oysters and mangrove propagules (D. Rumbold, S. Snedaker, personal communication). Under the action plan for conservation of nature in the Association of Southeast Asian Nations (ASEAN), the 7-year (1991–1998) ASEAN–Canada Cooperative Programme on Marine Science [338], Phase II (CPMS-II), focuses on establishing environmental criteria for development and management of living marine resources. To support this objective, one of the main activities is toxicity testing of approx. 10 chemicals using tropical marine algae, invertebrates, and fishes. Acute, sublethal, and chronic toxicity test protocols are being developed for organisms indigenous to the ASEAN area. More than 11 toxicity testing laboratories from the ASEAN region are involved in CPMS-II, and the data generated will be used to support formulation of tropical environmental quality criteria. In the ASEAN region, toxicity testing is becoming more common in environmental assessment. For example, the oil and gas industry conducts biological testing to evaluate the impacts of its activities, such as the use of oil dispersants, drilling muds, and other products.

Worldwide environmental quality criteria, guidelines, and standards, with a focus on tropical criteria, where possible, were compiled in a database [339]. This project reviewed the methods used to formulate environmental quality criteria and to summarize the values obtained for various jurisdictions. The review, which was reasonably extensive but not exhaustive, indicated that few tropical marine jurisdictions have published environmental quality criteria. In Table 5, environmental quality values (i.e., criteria, guidelines, standards) for tropical regions are presented for a few representative parameters. However, the ability of these criteria to adequately protect mangrove forests, seagrass meadows, and coral reefs has not been thoroughly investigated.

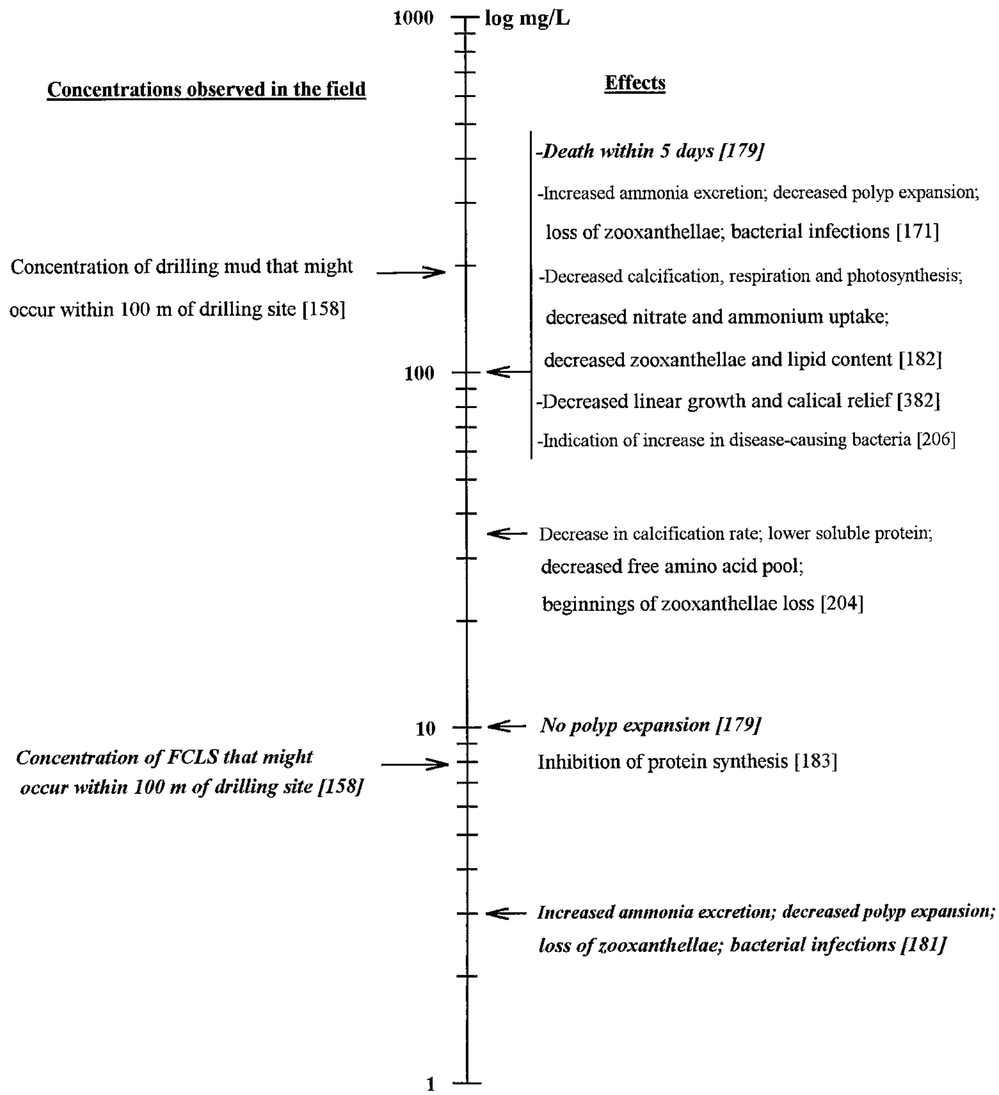

Fig. 5. Effects of ferrochrome lignosulfate (FCLS) and drilling muds on corals. References from which this information was compiled are cited in brackets.

Because of the variability in the species found in different tropical marine ecosystems of the world and perhaps their relative sensitivities to chemical stressors, toxicity tests following strict protocols should probably be developed for a suite of species in each ecoregion, such as the tropical western Atlantic, Great Barrier Reef, Hawaii, Red Sea, and southeast Asia. Simultaneous multiple-species toxicity tests [331] could provide a more rapid approach to obtaining minimum data to develop toxicity criteria [340]. Comparative toxicity tests, for example, simultaneous testing of several species of tropical amphipods from each region, could assist in determining the ability of the tests to predict toxicity for criteria development on a global basis.

Procedures for both acute and chronic toxicity tests are needed, and appropriate sublethal endpoints (growth, reproduction, immune responses, development, carcinogenesis, neurotoxicity) must be evaluated for organisms from tropical coastal ecosystems. Although it is difficult to bring massive corals, mangrove trees, and seagrasses into the laboratory, procedures for testing temperate sea urchin embryos, bivalve larvae, and seed germination can be adapted for tropical sea urchin species, coral larvae, and mangrove propagules. Perhaps simultaneous multiple-species acute and chronic tests could be conducted using mangrove propagules, fish and coral larvae, seagrass, and sea urchin embryos with appropriate endpoints (e.g., germination, mortality, productivity) to determine criteria. Gorrie et al. [341] reported that symbiotic algae from corals (*Symbiodinium kawagutii*) could be used in toxicity testing. If simultaneous tests were conducted with the intact coral–algal association to determine the responses, then the algae alone might be used as a standard test species on which to base criteria and conduct bioassays at potentially contaminated sites. The species included and endpoints examined should be linked to different assessment endpoints identified for each tropical marine ecosystem. Appropriate concentration–response distributions should be developed. Care must be taken in extrapolating results to less well-studied species [5]. Based on our review, we conclude that the effects of chemicals on reproduction and recruitment of corals and other key tropical marine organisms merit more emphasis in research programs. Efforts to coordinate species and test selection could prove fruitful and eliminate redundancy in the development of chemical criteria and benchmarks.

Risk characterization also involves the use of appropriate

Table 5. Summary of criteria values for tropical jurisdictions for several common environmental parameters[a]

Parameter	Tropical jurisdiction	Criteria	Comments	Reference
Cd	Australia	2.0 μg/L	At any time	[374]
	Hawaii	9.3 μg/L	As a 24-h average; applies to the dissolved fraction	[375]
	Hawaii	43.0 μg/L	At any time; applies to the dissolved fraction	[375]
	Thailand	5.0 μg/L	Conditions unspecified	[376]
Cu	Australia	5.0 μg/L	At any time	[374]
	Philippines	Narrative	Dissolved Cu; class SB[b]; 20.0 μg/L; class SC[c], 50.0 μg/L	[377]
	Thailand	50.0 μg/L	Conditions unspecified	[376]
Pb	Australia	5.0 μg/L	At any time	[374]
	Hawaii	140.0 μg/L	At any time; applies to the dissolved fraction	[375]
	Thailand	50.0 μg/L	Conditions unspecified	[376]
Oil and grease	California/Ocean	Narrative	Floating particulates and grease and oil shall not be visible	[378]
	Florida	Narrative	Dissolved or emulsified oils and greases shall not exceed 5.0 mg/L; no undissolved oil, or visible oil defined as iridescence, shall be present so as to cause taste or odor or otherwise interfere with the beneficial use of waters	[379]
	Hawaii	Narrative	All waters shall be free of substances attributable to domestic, industrial, or other controllable sources of pollutants, including floating debris, oil, grease, scum, or other floating materials	[375]
	Philippines	Narrative	Class SB[2], 2.0 mg/L; class SC[3], 3.0 mg/L	[377]
	Thailand	Not visible	Conservation of natural areas; aquaculture and shellfish, not visible	[376]
Zn	Australia	50.0 μg/L	At any time	[374]
	Florida	86.0 μg/L	Conditions unspecified	[379]
	Hawaii	95.0 μg/L	At any time; applies to the dissolved fraction	[374]
	Hawaii	86.0 μg/L	As a 24-h average; applies to the dissolved fraction	[374]
	Thailand	100.0 μg/L	Conditions unspecified	[376]

[a] Refer to the specific criteria documents for more details.
[b] SB classification refers to Recreational Water Class I (Areas regularly used by the public for swimming, skin diving, etc.) and Fishery Water Class I (Spawning areas for *Chanos chanos* or ''bangus'' and similar species) usage.
[c] SC classification refers to Recreational Water Class II (e.g., boating, etc.) and Fishery Water Class II (Commercial and sustenance fishing) usage.

statistical procedures to determine the significance of observed or predicted effects, and there have been many recent developments in this field that could be applied to tropical marine ecosystems. For example, Logan and Wilson [342] outlined a methodology for estimating ecological risks in situations in which populations are exposed to mixtures of chemical contaminants. Multivariate techniques, such as nonmetric clustering and association analysis [343], can combine different types of scales or metrics in multivariate analyses and could be used to combine large data sets into distinct measures to be used as a measure of risk and a test of the prediction of risk. These procedures could also improve the selection and interpretation of assessment and measurement endpoints because multivariate data can be used to identify natural and toxicant-induced patterns. New technologies such as GIS can provide overlays of spatial and temporal scaling of stresses with spatial and temporal aspects of species and life stages at risk [246,305] to improve the interpretation of data collected in support of chemical contaminant assessments. Risk assessment methodologies should be applied in tropical marine ecosystems as appropriate benchmarks and toxicity tests are developed and refined to investigate site-specific risks to specified assessment endpoints and to further validate and refine the methodologies for general use with populations and communities of the tropics, as well as to identify the uncertainties and limitations that will be associated with these methods.

Risk characterizations for tropical marine ecosystems need to be as rigorous as possible and should include the spectrum of factors known to affect organisms and their habitats and evaluations at several levels of organization to allow interpretation of the ecological significance of exposure to stressors (i.e., potential for ecosystem recovery from observed or predicted impacts). Multiple stressors (e.g., sedimentation, overfishing) should be factored into site-specific ecological risk assessments, and ranking of stressors (exposure and effects) must be conducted [242].

Even for the well-studied temperate ecosystems and biota, however, uncertainties abound in ecological risk assessment because few ecological risks can be measured with precision, resulting in wide confidence intervals for ecological predictions [17]. Careful discussion of uncertainties and use of ''best professional judgment'' in assessing ecological risk will always be important. Ecological risk assessments should provide information that can help managers evaluate their options and direct further data collection to refine the assessment if necessary. The weight-of-evidence approach, involving interpretation of qualitative and quantitative data and consideration of risks from all stressors identified to different ecological components at different levels of organization, is recommended [15]. Throughout the process, there must be sufficient knowledge and documentation of uncertainties, assumptions, and limitations that might affect the evaluation of risks.

SUMMARY AND CONCLUSIONS

Mangrove forests are sinks for heavy metals because the physical and chemical properties of mangrove sediments allow them to sequester large quantities of metals. For this reason mangroves tolerate high metal fluxes and serve as a buffer protecting adjacent marine communities. However, mangroves are extremely sensitive to oiling and herbicide exposure. When mangroves are destroyed, metals and organic compounds that might be bound in the sediments can be mobilized by erosion and oxidation of the sediments. Destruction of mangroves also exposes seagrass and coral reef communities to a pulse of nutrients and increased sedimentation. Seagrass meadows and coral reefs exposed to chemical contaminants from point and nonpoint sources from land and ocean also experience adverse effects on productivity, reproduction, and recruitment of component organisms, leading to eventual replacement of the communities with fewer, more tolerant species, as in temperate ecosystems [301].

These retrospective insights suggest that tropical marine ecosystems can be evaluated and that risks from chemical contaminants can be assessed and used to assess the need for remediation or prevention of adverse ecological effects. Further work with tropical species will be required, however. The primary producers discussed in this review are sensitive to changes in water quality, and their abundance can be visually assessed with relative ease, but they can be difficult to maintain and test in the laboratory. Other species might be used more successfully in toxicity tests to develop benchmarks for estimating risks.

One final question on ecotoxicology of coral reefs and adjacent ecosystems: Is this a moot point? Tropical marine ecosystems are vulnerable to other environmental stressors, particularly habitat destruction, sedimentation, and nutrient loading [2,3,4,12]. At some sites these stressors can be overwhelmingly more important than chemical contaminants. For example, eutrophication might pose a greater threat to seagrass communities than do organic pollutants and heavy metal contamination [116,344–346]. Yet, despite much research, our knowledge of the tolerance ranges and critical levels of nutrients for seagrass meadows and coral reefs is also minimal. Elevated nutrient levels can be considered ''toxic'' to corals in that they alter the animal–algal symbiotic association [12]. The photosynthetic efficiency of corals is reduced in the presence of added nutrients because of the accelerated growth of zooxanthellae (to the point of becoming self-shading) [347]. Shifting competitive interactions among coral reef species to favor faster-growing forms and fleshy algae increases the risk of bacterial infections in corals and reduces the amount of light available to the zooxanthellae by supporting phytoplankton growth. No studies on the role that chemical contaminants in sewage discharges might play in the ''toxic'' response have been conducted.

Chemical contaminants have sublethal chronic, as well as acute, direct and indirect impacts on mangrove, seagrass, and coral reef organisms. However, little is known about the interactions of these contaminants with calcareous sediments and bioavailability, the influence of high light and temperature, microbial degradation in the tropics, and food web bioaccumulation. The ecological effects of contaminants are often not distinguishable from anthropogenic or natural sedimentation, nutrient loading, and habitat destruction. Metals, petroleum hydrocarbons, and some pesticides often persist in the environment in sediments and organisms long after sources are removed. Changes in water quality or movement of sediments can increase their bioavailability and toxicity.

Therefore, considerations of chemical contamination can be important in evaluating the status of a tropical marine ecosystem and in predicting the likelihood of its demise or recovery under different management scenarios [242]. Research must be relevant to managers and must include quantification of diverse effects and interactions, including physical and biological stressors and overfishing [5,242,348]. Further research on the quantities and roles of anthropogenic chemical contaminants and the development of appropriate risk criteria are needed by managers now, and managers need to interact with scientists and assessors to focus research efforts [15]. As more guidance on tolerance levels becomes available, the ecological risk assessment process can assist in sorting out the relative risks at particular sites to strengthen management. Prevention through early detection and appropriate action should be the goal of scientists and managers concerned with protection of these vulnerable ecosystems. With a better understanding of how tropical marine ecosystems work, management practices that both reduce and mitigate for the negative effects of controllable human activities on these biological communities can be developed.

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