ANNUAL STOCK ASSESSMENT AND FISHERY EVALUATION REPORT: PACIFIC REMOTE ISLAND AREA FISHERY ECOSYSTEM PLAN 2016

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The ANNUAL STOCK ASSESSMENT AND FISHERY EVALUATION REPORT for the PACIFIC REMOTE ISLAD AREA FISHERY ECOSYSTEM 2016 was drafted by the Fishery Ecosystem Plan Team. This is a collaborative effort primarily between the Western Pacific Regional Fishery Management Council, NMFS-Pacific Island Fisheries Science Center, Pacific Islands Regional Office, Division of Aquatic Resources (HI) Department of Marine and Wildlife Resources (AS), Division of Aquatic and Wildlife Resources (Guam), and Division of Fish and Wildlife (CNMI).

This report attempts to summarize annual fishery performance looking at trends in catch, effort and catch rates as well as provide a source document describing various projects and activities being undertaken on a local and federal level. The report also describes several ecosystem considerations including fish biomass estimates, biological indicators, protected species, habitat, climate change and human dimensions. Information like marine spatial planning and best scientific information available for each fishery are described. This report provides a summary of annual catches relative to the Annual Catch Limits established by the Council in collaboration with the local fishery management agencies.

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The **Western Pacific Regional Fishery Management Council** acknowledges the valuable contributions of the following Plan Team members for drafting sections of this report:

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NMFS Pacific Islands Regional Office: Melanie Brown and Sarah Ellgen

Pacific Islands Regional Planning Body: Sarah Pautzke

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The Council would like to thank the following individuals for their contributions to the report:
Eileen Shea, Catherine Pham, Dawn Golden

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Executive Summary

As part of its 5 year fishery ecosystem plan (FEP) review, the Council identified the annual reports as a priority for improvement. The former annual reports have been revised to meet National Standard regulatory requirements for the Stock Assessment and Fishery Evaluation (SAFE) reports. The purpose of the report is twofold: monitor the performance of the fishery and ecosystem to assess the effectiveness of the FEP in meeting its management objectives; and maintain the structure of the FEP living document.. The reports are comprised of three chapters: fishery performance, ecosystem considerations, and data integration. The 2015 Pacific Remote Island Area (PRIA) annual SAFE report does not contain the fishery performance chapter. The Council will iteratively improve the annual SAFE report as resources allow.

Ecosystem considerations were added to the annual SAFE report following the Council's review of its fishery ecosystem plans and revised management objectives (pending Secretarial transmittal). Fishery independent ecosystem survey data, socioeconomics, protected species, climate and oceanographic, essential fish habitat, and marine planning information are included in the ecosystem considerations section. Fishery dependent sections will be included in later years as resources allow. Because human habitation is limited in the PRIA, socioeconomic information is also limited. The socioeconomics section of this report will be expanded in later years if activity increases.

Fishery independent ecosystem survey data was acquired through visual surveys conducted in PRIA, American Samoa, Guam, Commonwealth of Northern Mariana Islands, Main Hawaiian Islands, and Northwest Hawaiian Islands. This report illustrates the mean fish biomass for the reef areas within these locations. Additionally, the mean reef fish biomass and mean size of fishes (>10 cm) for PRIA are presented by sampling year and reef area. Finally, the reef fish population estimates for each PRIA study site are provided for hardbottom habitat (0-30 m).

The protected species section of this report summarizes information and monitors protected species interactions in fisheries managed under the PRIA FEP. There are currently no bottomfish, crustacean, coral reef or precious coral fisheries operating in the PRIA, and no historical observer data are available for fisheries under this FEP.

The climate change section of this report includes indicators of current and changing climate and related oceanic conditions in the geographic areas for which the Western Pacific Regional Fishery Management Council has responsibility. In developing this section, the Council relied on a number of recent reports conducted in the context of the U.S. National Climate Assessment including, most notably, the 2012 Pacific Islands Regional Climate Assessment and the Ocean and Coasts chapter of the 2014 report on a Pilot Indicator System prepared by the National Climate Assessment and Development Advisory Committee. The primary goal for selecting the indicators used in this report is to provide fisheries-related communities, resource managers and businesses with climate-related situational awareness. In this context, indicators were selected to be fisheries relevant and informative, build intuition about current conditions in light of changing climate, provide historical context and recognize patterns and trends. The atmospheric concentration of carbon dioxide (CO_2) trend is increasing exponentially with a time series maximum at 406.43 ppm. The oceanic pH at Station Aloha, in Hawaii has shown a significant linear decrease of -0.0386 pH units, or roughly a 9% increase in acidity ([H+]) since 1989. 2015 showed extreme high temperature anomalies, with values surpassing 20 degree heating weeks in the equatorial regions. The central Pacific saw seven named storms in 2016, three of which were hurricanes and two major.

The 2016 annual report includes an update of the precious corals species descriptions, effects of non-fishing and cumulative impacts on EFH. The guidelines also require a report on the condition of the habitat. In the 2016 annual report, mapping progress and benthic cover are included as indicators, pending development of habitat condition indicators for the PRIA not otherwise represented in other sections of this report. The annual report also addresses any Council directives toward its plan team. There were no directives in 2016.

The marine planning section of the annual report tracks activities with multi-year planning horizons and begins to track the cumulative impact of established facilities. Development of the report in later years will focus on identifying appropriate data streams. No ocean activities with multi-year planning horizons were identified for the Pacific Remote Islands Areas.

The Data Integration Chapter of this report is still under development. The Council hosted a Data Integration Workshop on November 30 - December 1, 2017 with participants from the NMFS Pacific Islands Regional Office and Pacific Islands Fisheries Science Center. The goal of the workshop was to identify policy-relevant fishery ecosystem relationships. The archipelagic data integration chapter will investigate 30 fishery dependent variable-ecological/environmental indicator combinations. A contractor is currently conducting the region-wide analysis and results will be included in the 2017 SAFE report if available.

The Archipelagic Plan Team made the following recommendations with respect to the PRIA FEP.

> Regarding the data integration chapter of the SAFE report, the Archipelagic Fishery Ecosystem Plan Team recommends the Council include the following variables in the exploratory data analysis being conducted by the Council's contractor:
>
> - Effect of subsidy program
> - Market forcing
> - Effects of fish import-export
>
> Regarding the species table, the Archipelagic Fishery Ecosystem Plan Team recommends the Council direct staff, in coordination with NMFS staff, to convene a working group to finalize the species table used to generate fishery statistics
>
> Regarding Essential Fish Habitat, the Plan Team recommends that the Council:
>
> - Consider amending the non-fishing impacts, cumulative impacts, and conservation and enhancement recommendations in the Western Pacific FEPs based on the options provided by the Plan Team, and

- Consider amending the EFH designations and species descriptions for precious corals based on the options provided by the Plan Team.

The Archipelagic Fishery Ecosystem Plan Team recognizes the importance of the ecosystem component amendment to address the operational issues associated with the data limited stocks managed under Annual Catch Limits.

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ACRONYMS AND ABBREVIATIONS

ABC	Acceptable Biological Catch
ACL	Annual Catch Limits
AM	Accountability Measures
BiOp	Biological Opinion
BOEM	Bureau of Ocean Energy Management
BSIA	best scientific information available
CFR	Code of Federal Regulations
CMS	coastal and marine spatial
CNMI	Commonwealth of the Northern Mariana Islands
CPUE	Catch Per Unit Effort
CREMUS	Coral Reef Eco Management Unit Species
CREP	Coral Reef Ecosystem Program (PIFSC)
EEZ	Exclusive Economic Zone
EFH	Essential Fish Habitat
EO	Executive Order
ESA	Endangered Species Act
FEP	Fishery Ecosystem Plan
FMP	Fishery Management Plan
HAPC	Habitat Area of Particular Concern
ITS	Incidental Take Statement
LOF	List of Fisheries
MFMT	Maximum Fishing Mortality Threshold
MHI	Main Hawaiian Islands
MMA	marine managed area
MPA	marine protected area
MPCC	Marine Planning and Climate Change
MPCCC	Council’s MPCC Committee
MSA	Magnuson-Stevens Fishery Conservation and Management Act
MSST	Minimum Stock Size Threshold
MSY	Maximum Sustainable Yield
MUS	management unit species
MSFCMA	Magnuson-Stevens Fishery Conservation and Management Act
NEPA	National Environmental and Policy Act
NMFS	National Marine Fisheries Service
NWHI	Northwestern Hawaiian Islands
OFL	Over-fishing Limit
OY	Optimum Yield
Pelagic FEP	Fishery Ecosystem Plan for the Pacific Pelagic Fisheries
PI	Pacific Islands
PIFSC	Pacific Islands Fisheries Science Center
PIRO	NOAA NMFS Pacific Islands Regional Office
PMUS	pelagic management unit species

RAMP	Reef Assessment and Monitoring Program (CREP)
ROA	Risk of Overfishing Analysis
RPB	Regional Planning Body
SAFE	Stock Assessment and Fishery Evaluation
SDC	Status Determination Criteria
SEEM	Social, Ecological, Economic, and Management Uncertainty Analysis
TAC	Total Annual Catch
USACE	United States Army Corps of Engineers
WPRFMC	Western Pacific Regional Fishery Management Council

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1 FISHERY PERFORMANCE

Fisheries in the Pacific Remote Island Area (PRIA) are limited. Fishery performance statistics will be made available for the PRIA in subsequent reports as resources allow.

1.1 Number of Federal Permit Holders

In the PRIA, the following Federal permits are required for fishing in the EEZ:

1.1.1 Special Coral Reef Ecosystem Permit

The coral reef ecosystem special permit is required for anyone fishing for coral reef ecosystem management unit species in a low-use MPA, fishing for species on the list of Potentially Harvested Coral Reef Taxa, or using fishing gear not specifically allowed in the regulations. The permit expires one year after the date of issuance. Permit holder must submit a logbook to NOAA Fisheries within 30 days of each landing of coral reef harvest.

A transshipment permit is required for any receiving vessel used to land or transship potentially harvested coral reef taxa, or any coral reef ecosystem management unit species caught in a low-use MPA. Exceptions to this permit requirement are made for anyone issued a permit to fish under the other western Pacific fishery management plans (pelagic, bottomfish and seamount groundfish, crustacean, or precious corals) who catch coral reef management unit species incidentally while fishing for the management unit species covered by the permit they possess. Permit holders must submit a logbook to NOAA Fisheries within seven days following the date the vessel arrived in port to land transshipped fish. Regulations governing this fishery can be found in the Code of Federal Regulations, Title 50, Part 665.

1.1.2 Western Pacific Precious Corals Permit

This permit is required for anyone harvesting or landing black, bamboo, pink, red, or gold corals in the EEZ in the western Pacific. The permit expires one year from the date of issuance. Permit holders must submit a logbook to NMFS within 72 hours of landing. Specific conditions are associated with various established, provisional, and exploratory areas throughout the region.

1.1.3 Western Pacific Crustaceans Permit (Lobster or Deepwater Shrimp)

A permit is required by the owner of a U.S. fishing vessel used to fish for lobster or deepwater shrimp in the EEZ around American Samoa, Guam, Hawaii, and the Pacific Remote Islands Areas, and in the EEZ seaward of 3 nautical miles of the shoreline of the Northern Mariana Islands. The permit expires one year after the date of issuance. Permit holders must submit a logbook to NMFS within 72 hours of landing (except when fishing in the Pacific Remote Island Areas – those reports are due within 30 days).

1.1.4 Pacific Remote Island Areas Bottomfish Permit

This permit is required by anyone using bottomfish gear to fish for bottomfish management unit species in the EEZ around the Pacific Remote Island Areas (Palmyra Atoll, Kingman Reef, Jarvis Island, Baker Island, Howland Island, Johnston Atoll, and Wake Island). The permit expires one year after the date of issuance. Vessel operators must submit a logbook to NMFS within 30 days of landing. Regulations governing this fishery can be found in the Code of Federal Regulations, Title 50, Part 665, Subpart E. Commercial fishing is prohibited within the boundaries of the Pacific Remote Islands Marine National Monument.

There is no record of special coral reef or precious coral fishery permits issued for the EEZ around the PRIAs since 2007. Table 1 provides the number of permits issued for PRIA fisheries between 2007 and 2017. Historical data are from the PIFSC accessed on February 9, 2017 and 2017 data are from the PIRO Sustainable Fisheries Division permits program as of February 3, 2017.

Table 1. 2017 Number of federal permits holders between 2007 and 2017 for the crustacean and bottomfish fisheries of the PRIA

PRIA Fisheries	**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**
Lobster	3	2	3								
Shrimp				1*							
Bottomfish	7	2	3	6	5	4	1	2		1	1

***Same permit applies to American Samoa, Guam, CNMI and PRIA.**

1.2 Administrative and Regulatory Actions

PIRO took no management actions specific to PRIA since the April 2016 Joint FEP Plan Team meeting, as reported to the 166rd to 168th Western Pacific Fishery Management Council meetings held June 2016, October 2016, and March 2017. One proposed rule was published as described below.

January 18, 2017 (82 FR 5517). **Pacific Island 2016 Annual Catch Limits and Accountability Measures**. NMFS proposed annual catch limits (ACLs) for Pacific Island bottomfish, crustacean, precious coral, and coral reef ecosystem fisheries, and accountability measures (AMs) to correct or mitigate any overages of catch limits. The proposed ACLs and AMs would be effective for fishing year 2016. The fishing year for each fishery begins on January 1 and ends on December 31, except for precious coral fisheries, which begin July 1 and end on June 30 the following year. Although the 2016 fishing year has ended for most stocks, NMFS evaluates 2016 catches against the 2016 ACLs when data become available in mid-2017. The proposed ACLs and AMs support the long-term sustainability of fishery resources of the U.S. Pacific Islands. The comment period ended February 2, 2017.

2 ECOSYSTEM CONSIDERATIONS

2.1 Coral Reef Fish Ecosystem Parameters

2.1.1 Archipelagic Reef Fish Biomass

Description: ‘Reef fish biomass’ is mean biomass of reef fishes per unit area derived from visual survey data (details of survey program below) between 2009 and 2015.

Category:
- ✓ Fishery independent
- ☐ Fishery dependent
- ☐ Biological

Timeframe: Triennial

Jurisdiction:
- ✓ American Samoa
- ✓ Guam
- ✓ Commonwealth of Northern Mariana Islands
- ✓ Main Hawaiian Islands
- ✓ Northwest Hawaiian Islands
- ✓ Pacific Remote Island Areas

Spatial Scale:
- ✓ Regional
- ☐ Archipelagic
- ☐ Island
- ☐ Site

Data Source: Data used to generate biomass estimates comes from visual surveys conducted by NOAA PIFSC Coral Reef Ecosystem and partners, as part of the Pacific Reef Assessment and Monitoring Program (http://www.pifsc.noaa.gov/cred/pacific_ramp.php). Survey methods are described in detail elsewhere (http://www.pifsc.noaa.gov/library/pubs/admin/PIFSC_Admin_Rep_15-07.pdf), but in brief involve teams of divers conducting stationary point count cylinder (SPC) surveys within a target domain of <30m hard-bottom habitat at each island, stratified by depth zone and, for larger islands, by section of coastline. For consistency among islands, only date from forereef habitats is used here. At each SPC, divers record the number, size and species of all fishes within or passing through paired 15m-diameter cylinders in the course of a standard count procedure. Fish sizes and abundance are converted to biomass using standard length-to-weight conversion parameters, taken largely from FishBase (http://www.fishbase.org), and converted to biomass per unit area, by dividing by the area sampled per survey. Site-level data were pooled into island-scale values by first calculating mean and variance within strata, and then calculating weighted island-scale mean and variance using the formulas given in (Smith et al., 2011), with strata weighted by their respective sizes.

Rationale: Reef Fish biomass, i.e. the weight of fish per unit area has been widely used as an indicator of relative status, and has repeatedly been shown to be changes in fishing pressure, habitat quality, and oceanographic regime.

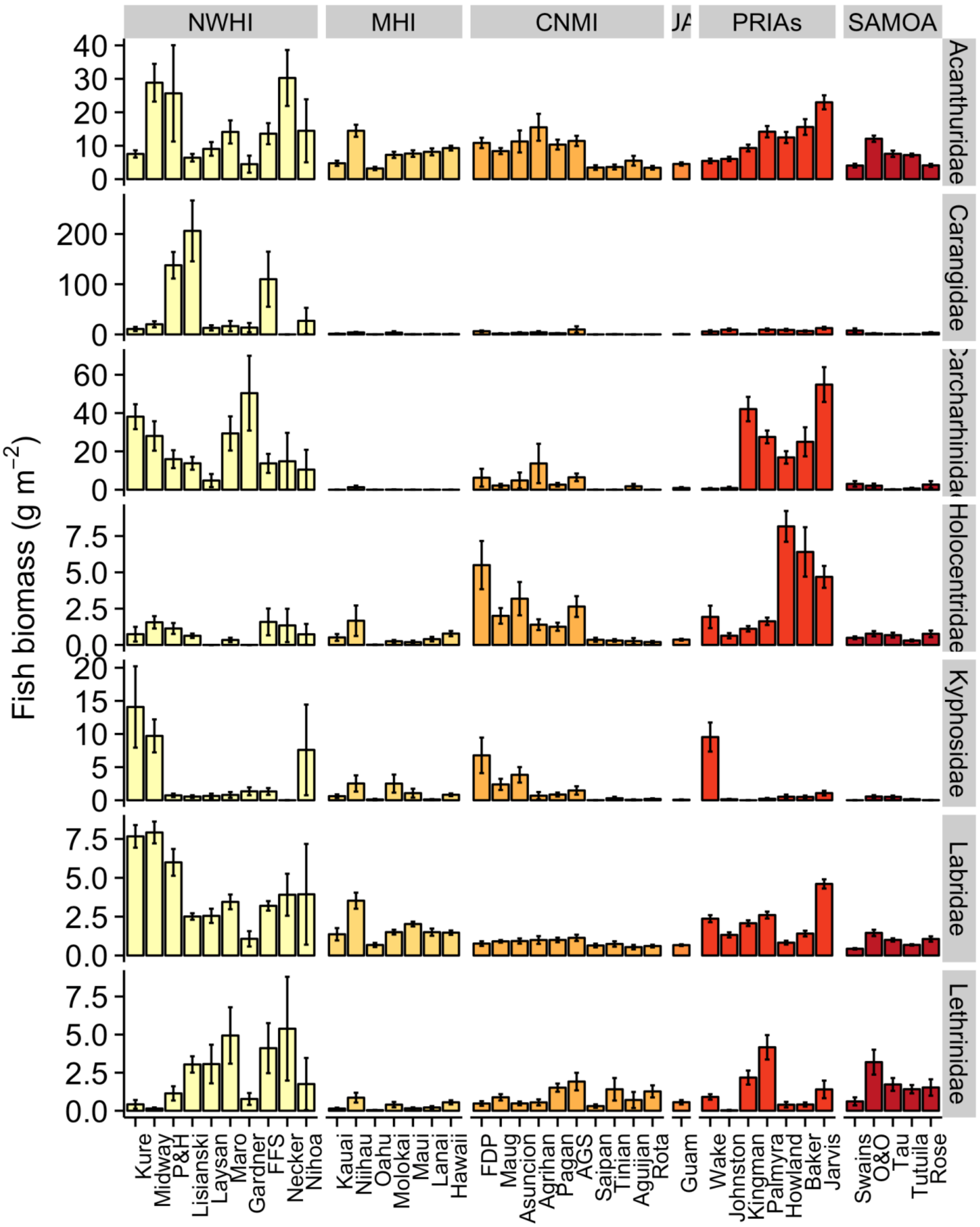
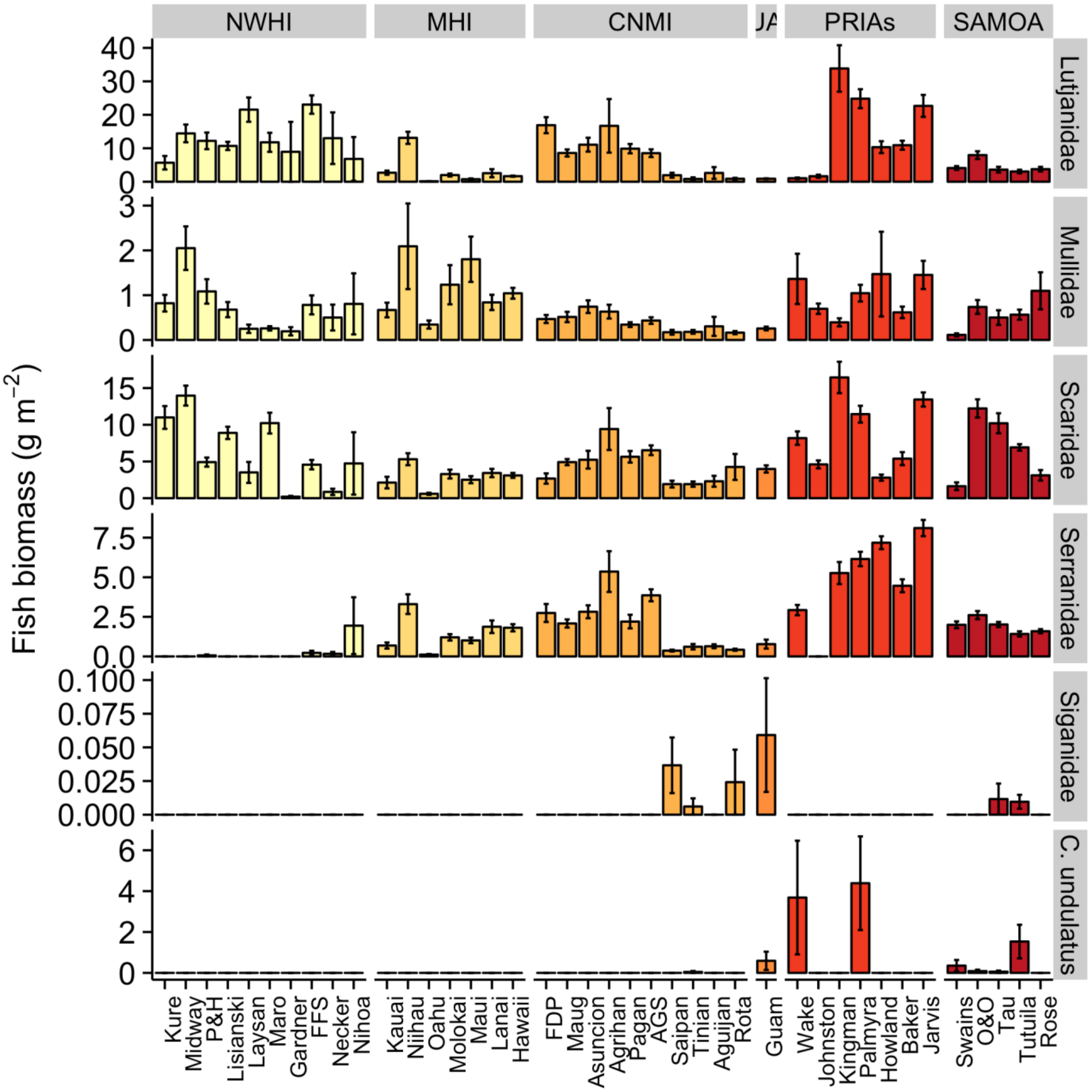


Figure 1. Mean fish biomass by Coral Reef Management Unit Species (CREMUS) grouping per US Pacific reef area. Mean fish biomass (± standard error) per CREMUS grouping per

reef area pooled across survey years (2009-2015). Islands ordered within region by latitude. Continues to next page.


NWHI
MHI
CNMI
JA
PRIAs
SAMOA
Fish biomass (g m^{-2})
Lutjanidae
Mullidae
Scaridae
Serranidae
Siganidae
C. undulatus
0
10
20
30
40
0
1
2
3
0
5
10
15
0.0
2.5
5.0
7.5
0.000
0.025
0.050
0.075
0.100
0
2
4
6
Kure
Midway
P&H
Lisianski
Laysan
Maro
Gardner
FFS
Necker
Nihoa
Kauai
Niihau
Oahu
Molokai
Maui
Lanai
Hawaii
FDP
Maug
Asuncion
Agrihan
Pagan
AGS
Saipan
Tinian
Aguijan
Rota
Guam
Wake
Johnston
Kingman
Palmyra
Howland
Baker
Jarvis
Swains
O&O
Tau
Tutuila
Rose

2.1.2 Regional Reef Fish Biomass

Description: 'Reef fish biomass' is mean biomass of reef fishes per unit area derived from visual survey data (details of survey program below) between 2009 and 2015.

Category:

- ✓ Fishery independent
- ☐ Fishery dependent
- ☐ Biological

Timeframe: Triennial

Jurisdiction:

- ☐ American Samoa
- ☐ Guam
- ☐ Commonwealth of Northern Mariana Islands
- ☐ Main Hawaiian Islands
- ☐ Northwest Hawaiian Islands
- ✓ Pacific Remote Island Areas

Scale:

- ☐ Regional
- ☐ Archipelagic
- ✓ Island
- ☐ Site

Data Source: Data used to generate biomass estimates comes from visual surveys conducted by NOAA PIFSC Coral Reef Ecosystem and partners, as part of the Pacific Reef Assessment and Monitoring Program (http://www.pifsc.noaa.gov/cred/pacific_ramp.php). Survey methods are described in detail elsewhere (http://www.pifsc.noaa.gov/library/pubs/admin/PIFSC_Admin_Rep_15-07.pdf), but in brief involve teams of divers conducting stationary point count cylinder (SPC) surveys within a target domain of <30m hard-bottom habitat at each island, stratified by depth zone and, for larger islands, by section of coastline. For consistency among islands, only date from forereef habitats is used here. At each SPC, divers record the number, size and species of all fishes within or passing through paired 15m-diameter cylinders in the course of a standard count procedure. Fish sizes and abundance are converted to biomass using standard length-to-weight conversion parameters, taken largely from FishBase (http://www.fishbase.org), and converted to biomass per unit area, by dividing by the area sampled per survey. Site-level data were pooled into island-scale values by first calculating mean and variance within strata, and then calculating weighted island-scale mean and variance using the formulas given in (Smith et al., 2011), with strata weighted by their respective sizes.

Rationale: Reef Fish biomass, i.e. the weight of fish per unit area has been widely used as an indicator of relative status, and has repeatedly been shown to be changes in fishing pressure, habitat quality, and oceanographic regime.

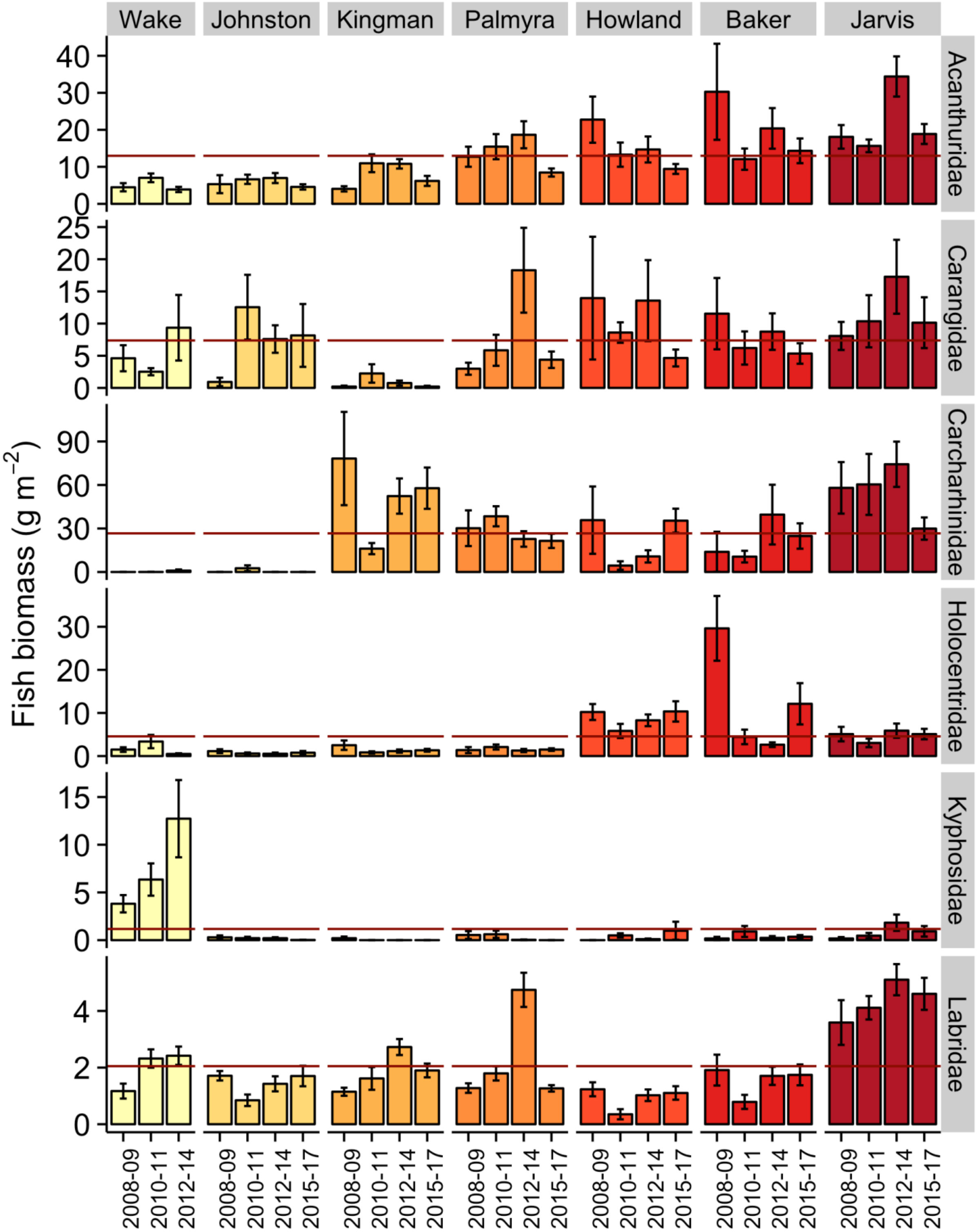
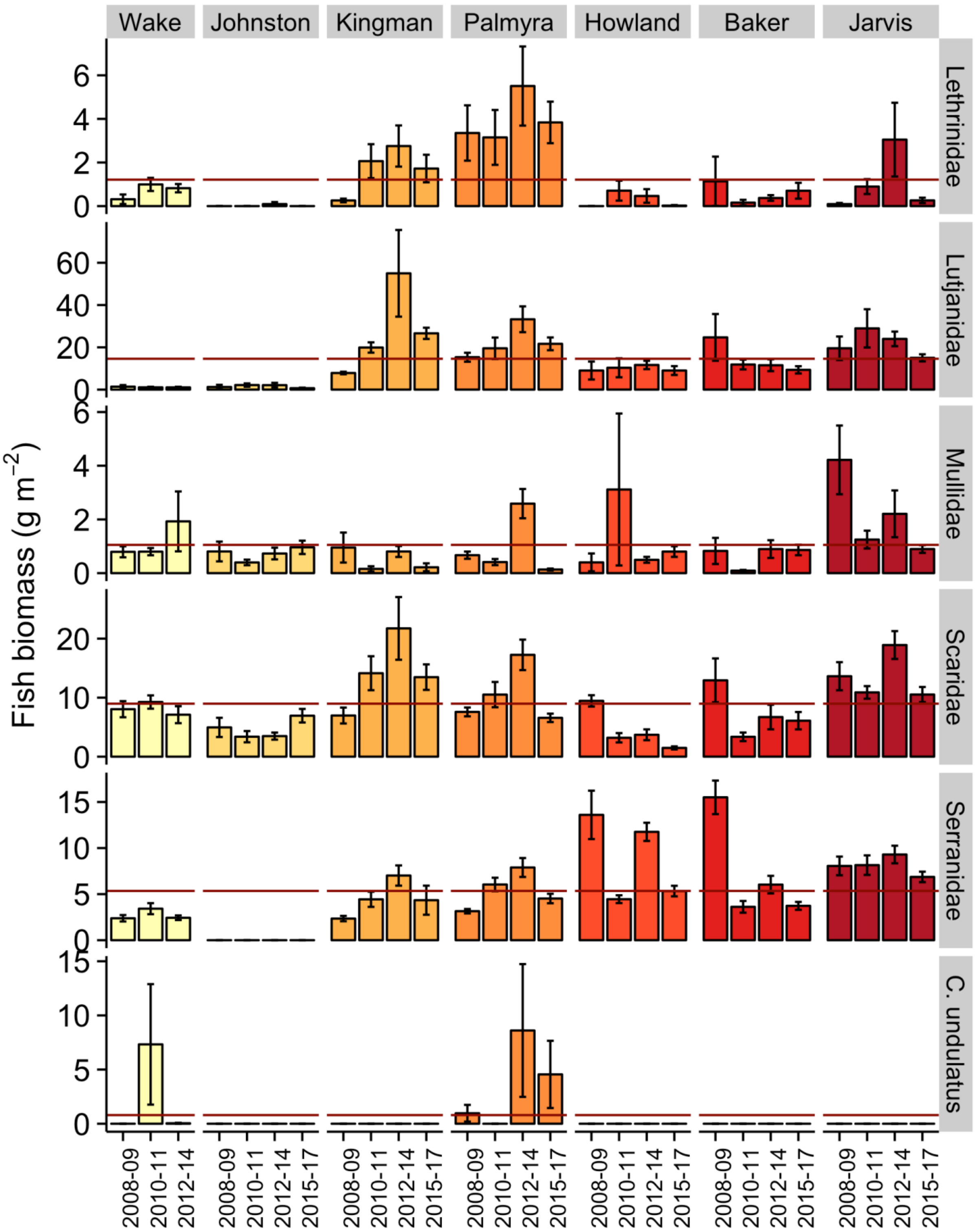


Figure 2. PRIA mean reef fish biomass. Continues to next page.


Wake
Johnston
Kingman
Palmyra
Howland
Baker
Jarvis
Lethrinidae
Lutjanidae
Mullidae
Scaridae
Serranidae
C. undulatus
Fish biomass (g m−2)
0
2
4
6
0
20
40
60
0
2
4
6
0
10
20
0
5
10
15
0
5
10
15
2008-09
2010-11
2012-14
2015-17

2.1.3 Archipelagic Mean Fish Size

Description: ’Mean fish size’ is mean size of reef fishes > 10 cm TL (i.e. excluding small fishes) derived from visual survey data (details of survey program below) between 2009 and 2015 .

Category:

- ✓ Fishery independent
- ☐ Fishery dependent
- ☐ Biological

Timeframe: Triennial

Jurisdiction:

- ☐ Regional
- ☐ American Samoa
- ☐ Guam
- ☐ Commonwealth of Northern Mariana Islands
- ☐ Main Hawaiian Islands
- ☐ Northwest Hawaiian Islands
- ✓ Pacific Remote Island Areas

Scale:

- ☐ Regional
- ☐ Archipelagic
- ✓ Island
- ☐ Site

Data Source: Data used to generate mean size estimates comes from visual surveys conducted by NOAA PIFSC Coral Reef Ecosystem and partners, as part of the Pacific Reef Assessment and Monitoring Program (http://www.pifsc.noaa.gov/cred/pacific_ramp.php). Survey methods are described in detail elsewhere (http://www.pifsc.noaa.gov/library/pubs/admin/PIFSC_Admin_Rep_15-07.pdf), but in brief involve teams of divers conducting stationary point count cylinder (SPC) surveys within a target domain of <30m hard-bottom habitat at each island, stratified by depth zone and, for larger islands, by section of coastline. For consistency among islands, only date from forereef habitats is used here. At each SPC, divers record the number, size (total length, TL) and species of all fishes within or passing through paired 15m-diameter cylinders in the course of a standard count procedure. Fishes smaller than 10 cm TL are excluded so that the fish assemblage measured more closely reflects fishes that are potentially fished, and so that mean sizes are not overly influenced by variability in space and time of recent recruitment. Site-level data were pooled into island-scale values by first calculating mean and variance within strata, and then calculating weighted island-scale mean and variance using the formulas given in (Smith et al., 2011), with strata weighted by their respective sizes.

Rationale: Mean size is important as\mean size is widely used as an indicator of fishing pressure – not only do fishers sometimes preferentially target large individuals, but also because one effect of fishing is to reduce the number of fishes reaching older (and larger) size classes. Large fishes also contribute disproportionately to community fecundity and can have important ecological roles – for example, excavating bites by large parrotfishes probably have a longer lasting impact on reef benthos than bites by smaller fishes.

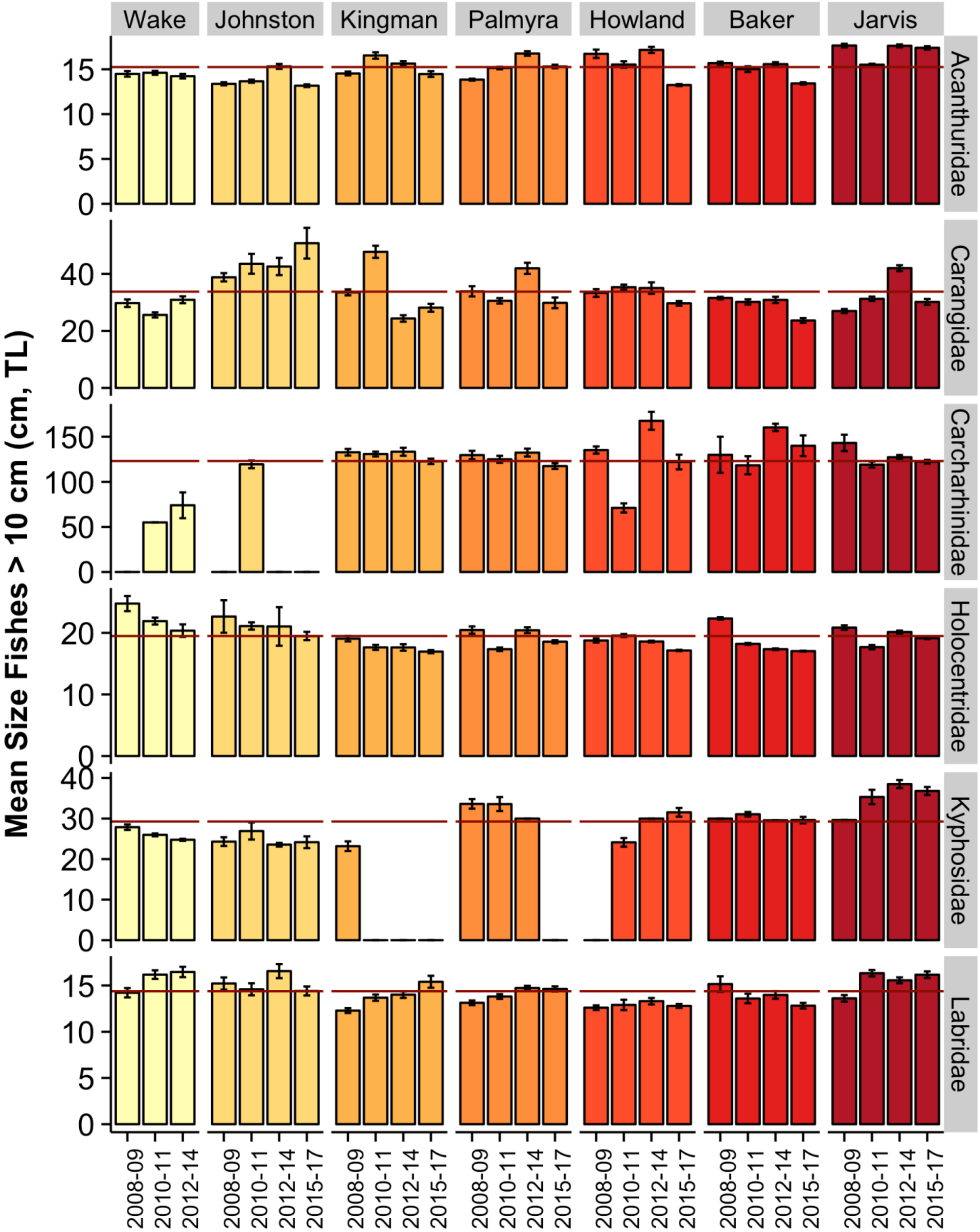
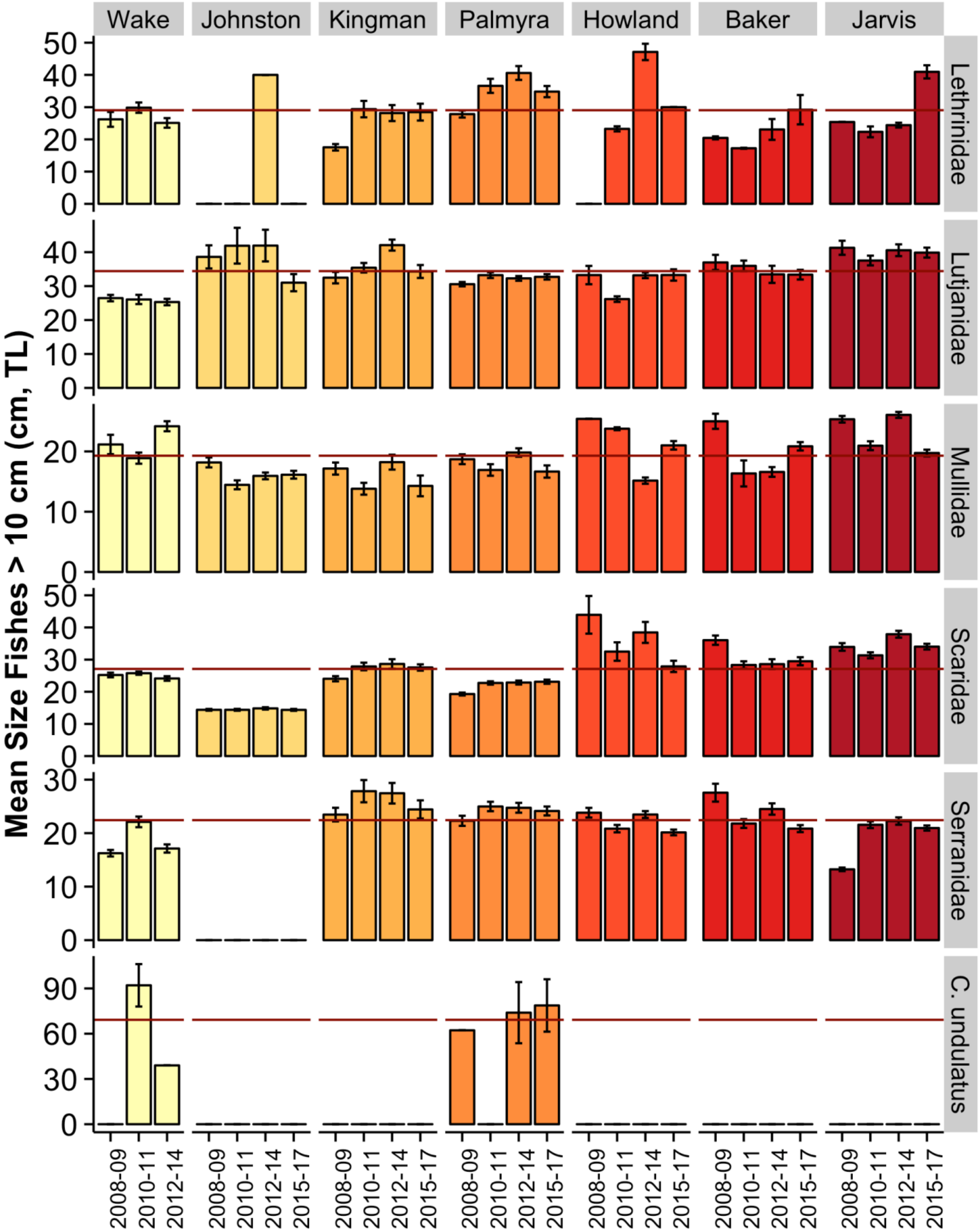


Figure 3. PRIA mean fish size. Continues to next page.


Wake
Johnston
Kingman
Palmyra
Howland
Baker
Jarvis
Lethrinidae
Lutjanidae
Mullidae
Scaridae
Serranidae
C. undulatus
Mean Size Fishes > 10 cm (cm, TL)
0
10
20
30
40
50
60
90
2008-09
2010-11
2012-14
2015-17

2.1.4 Reef Fish Population Estimates

Description: Reef fish population estimates are made by multiplying mean biomass per unit area by estimated area of hardbottom in a consistent habitat across all islands (specifically, the area of hardbottom forereef habitat in < 30m water).

Category:
- ✓ Fishery independent
- ☐ Fishery dependent
- ☐ Biological

Timeframe: Triennial

Jurisdiction:
- ☐ Regional
- ☐ American Samoa
- ☐ Guam
- ☐ Commonwealth of Northern Mariana Islands
- ☐ Main Hawaiian Islands
- ☐ Northwest Hawaiian Islands
- ✓ Pacific Remote Island Areas

Scale:
- ☐ Regional
- ☐ Archipelagic
- ✓ Island
- ☐ Site

Data Source: Data used to generate mean size estimates comes from visual surveys conducted by NOAA PIFSC Coral Reef Ecosystem and partners, as part of the Pacific Reef Assessment and Monitoring Program (http://www.pifsc.noaa.gov/cred/pacific_ramp.php). Survey methods and sampling design, and methods to generate reef fish biomass are described above (SECTION: REEF FISH BIOMASS). Those estimates are converted to population estimates by multiplying biomass (g/m2) per island by the estimated area of hardbottom habitat <30m deep at the island, which is the survey domain for the monitoring program that biomass data comes from.. Estimated habitat areas per island are derived from GIS bathymetry and habitat maps maintained by NOAA Coral Reef Ecosystems Program. It is important to recognize that many reef fishes taxa are present in other habitats and in deeper water than is surveyed by that program, and even that some taxa likely have the majority of their populations in deeper water. Additionally, fish counts have the potentiual to be biased by the nature of fish responses to divers- curious fishes, particularly in locations where divers are not perceived as a threat, will tend to be overcounted by visual survey, and skittish fishes will tend to be undercounted. Likely numbers of jacks and sharks in some locations (particularly the NWHI) are overcounted by visual survey.

Nevertheless, in spite of these issues, the data shown here are consistently gathered across space and time.

Rationale: These data have utility in understanding the size of poulations from which fishery harvests are extracted.

Table 2. Reef fish population estimates for PRIA. Fish species are pooled by CREMUS groupings. Estimated population biomass is for 0-30 m hardbottom habitat only. (n) is number of sites surveyed per island. Each site is surveyed by means of 2-4 7.5 m diameter S SPCs -– however, those are not considered to be independent samples, so data from those is pooled to site level before other analysis.

ISLAND	Total Area of reef (Ha)	N	ESTIMATED POPULATION BIOMASS (metric Tonnes) in SURVEY DOMAIN OF <30m HARDBOTTOM					
			Acanthuridae	Carangidae	Carcharhinids	Holocentridae	Kyphosidae	Labridae
Wake	1,282.0	75	69.9	76.1	6.3	24.8	122.3	30.4
Johnston	9,410.2	104	570.1	887.6	81.2	60.1	13.5	124.7
Kingman	3,721.1	130	346.8	39.8	1,566.1	41.5	-	77.4
Palmyra	4,212.7	160	597.7	400.5	1,160.4	68.6	9.2	109.7
Howland	172.9	90	21.5	15.5	29.1	14.1	0.9	1.4
Baker	390.3	81	60.9	26.4	97.5	25.0	2.0	5.5
Jarvis	365.9	134	84.1	46.1	200.8	17.1	3.9	16.9
TOTAL	**19,555.1**	**774**	**1,754.9**	**1,490.6**	**3,217.0**	**249.3**	**111.2**	**363.0**
ISLAND	Total Area of reef (Ha)	N	Lethrinidae	Lutjanidae	Mullidae	Scaridae	Serranidae	*C. undulatus*
Wake	1,282.0	75	11.6	13.5	17.5	104.9	37.5	47.2
Johnston	9,410.2	104	2.9	155.1	65.6	433.2	-	-
Kingman	3,721.1	130	81.1	1,259.5	14.7	611.9	195.9	-
Palmyra	4,212.7	160	175.5	1,045.6	44.0	482.1	259.2	184.8
Howland	172.9	90	0.7	17.9	2.5	4.8	12.4	-
Baker	390.3	81	1.6	42.6	2.4	21.0	17.4	-
Jarvis	365.9	134	5.1	82.9	5.3	49.2	29.7	-
TOTAL	**19,555.1**	**774**	**280.1**	**2,661.1**	**148.8**	**1,707.2**	**549.1**	**220.8**

Note (1): No Siganidae or *Bolbometopon muricatum* were observed in PRIAs during these surveys.

2.2 Socioeconomics

This section outlines the pertinent economic, social, and community information available for assessing the successes and impacts of management measures or the achievements of the Fishery Ecosystem Plan for the Pacific Remote Island Area (PRIA, Western Pacific Regional Fishery

Management Council, 2016). It meets the objective “Support Fishing Communities” adopted at the 165th Council meeting; specifically, it identifies the various social and economic groups within the region’s fishing communities and their interconnections. The section begins with an overview of the socioeconomic context for the region, then provides a summary of relevant studies and data for the PRIA.

In 1996, the Magnuson-Stevens Fishery Conservation and Management Act’s National Standard 8 (NS8) specified that conservation and management measures take into account the importance of fishery resources to fishing communities, to provide for their sustained participation in fisheries and to minimize adverse economic impacts, provided that these considerations do not compromise the achievement of conservation. Unlike other regions of the U.S., the settlement of the Western Pacific region was intimately tied to the sea (Figure 4), which is reflected in local culture, customs, and traditions.

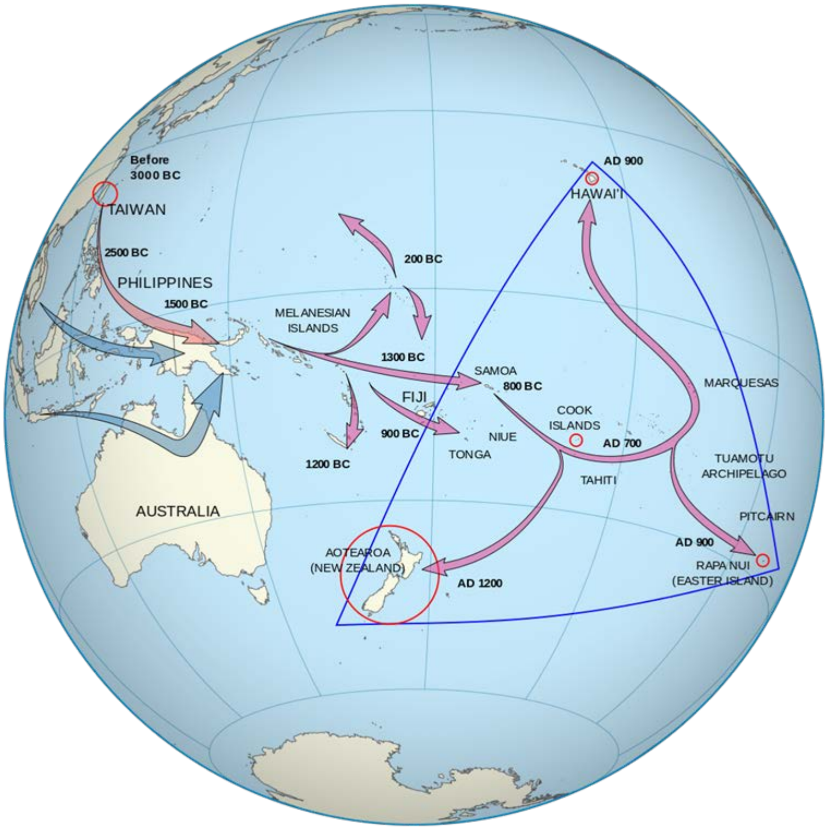


Figure 4. Settlement of the Pacific Islands, courtesy Wikimedia Commons, https://commons.wikimedia.org/wiki/File:Polynesian_Migration.svg.

Polynesian voyagers relied on the ocean and marine resources on their long voyages in search of new islands, as well as in sustaining established island communities. Today, the population of the region also represents many Asian cultures from Pacific Rim countries, which reflect similar importance of marine resources. Thus, fishing and seafood are integral local community ways of life. This is reflected in the amount of seafood eaten in the region in comparison to the rest of the United States, as well as the language, customs, ceremonies, and community events. It can also affect seasonality in prices of fish. Because fishing is such an integral part of the culture, it is difficult to cleanly separate commercial from non-commercial fishing, with most trips involving multiple motivations and multiple uses of the fish caught. While economics are an important consideration, fishermen report other motivations such as customary exchange as being equally, if not more, important. Due to changing economies and westernization, recruitment of younger fishermen is becoming a concern for the sustainability of fishing and fishing traditions in the region.

2.2.1 Response to Previous Council Recommendations

At its 166th meeting held in Tumon, Guam, the Council directed staff to develop a brief report identifying data sources, quality and coverage for each required socioeconomic parameter in the annual/SAFE reports, as resources permit. This report should also identify the quality and coverage of this data, as well as any gaps. This data synthesis was conducted and used to guide the development of this chapter with further input and guidance from the Council Social Science Planning Committee and Archipelagic Plan Team.

The Council also directed the Plan Team to consider for future Annual/SAFE reports including enhanced information on social, economic and cultural impacts of a changing climate and increased pressure on the ocean and its resources. PIFSC developed a Regional Action Plan and Climate Science Strategy as a first step in providing this information (Polovina et al., 2016).

2.2.2 Introduction

Human habitation in the Pacific Remote Island Area is limited. The Fishery Ecosystem Plan for the Pacific Remote Islands Area provides a description of the geography, history, and socio-economic considerations of the archipelago in section 1.3 (Western Pacific Regional Fishery Management Council, 2016). Grace-McCaskey (2014) provided a brief review of the importance of this area from a cultural perspective. She noted that although this region was uninhabited when first visited by Westerners, Polynesians and Micronesians likely had been visiting all of the islands periodically for centuries. Most of the islands in the PRIA were modified during WWII and many have subsequently become National Wildlife Refuges and part of the Pacific Remote Islands Marine National Monument. Only Wake, Johnston, and Palmyra have seasonal and year-round residents, primarily related to military and refuge management. Because they are located far from areas of high human population, they are considered to be some of the healthiest reef ecosystems in the world, although some are experiencing residual impacts from military use. There are no designated fishing communities residing in the PRIA. Most of the fishing effort has been concentrated around Johnston and Palmyra by members of the Hawaii fishing community.

2.2.3 Ongoing Research and Information Collection

There is currently no ongoing research specific to the Pacific Remote Island Area.

2.2.4 Relevant PIFSC Econmoics and Human Dimensions Publications: 2016

No publications specific to the Pacific Remote Islands Area were produced in 2016.

2.2.5 References

Grace-McCaskey, C. 2014. Examining the potential of using secondary data to better understand human-reef relationships across the Pacific. Pacific Islands Fish. Sci. Cent., Natl. Mar. Fish. Serv., NOAA, Honolulu, HI 96818-5007. Pacific Islands Fish. Sci. Cent. Admin.

Rep. H-14-01, 69 p. https://www.pifsc.noaa.gov/library/pubs/admin/PIFSC_Admin_Rep_14-01.pdf

Polovina J and Dreflak K (Chairs), Baker J, Bloom S, Brooke S, Chan V, Ellgen S, Golden D, Hospital J, Van Houtan K, Kolinski S, Lumsden B, Maison K, Mansker M, Oliver T, Spalding S, Woodworth-Jefcoats P. 2016. Pacific Islands Regional Action Plan: NOAA Fisheries climate science strategy. U.S. Dept. of Commerce, NOAA Technical Memorandum NOAA-TM-NMFS-PIFSC-59, 33 p. doi:10.7289/V5/TM-PIFSC-59.

Western Pacific Regional Fishery Management Council. 2016. Draft Fishery Ecosystem Plan for the Pacific Remote Island Area. Honolulu, HI. 81 p. + Appendices.

2.3 Protected Species

This section of the report summarizes information on protected species interactions in fisheries managed under the PRIA FEP. Protected species covered in this report include sea turtles, seabirds, marine mammals, sharks, and corals. Most of these species are protected under the Endangered Species Act (ESA), the Marine Mammal Protection Act (MMPA), and/or the Migratory Bird Treaty Act (MBTA). A list of protected species found in or near PRIA waters and a list of critical habitat designations in the Pacific Ocean are included in Appendix B.

2.3.1 Indicators for Monitoring Protected Species Interactions in the PRIA FEP Fisheries

This report monitors the status of protected species interactions in the PRIA FEP fisheries using proxy indicators such as fishing effort and changes in gear types as these fisheries do not have observer coverage. Logbook programs are not expected to provide reliable data about protected species interactions.

2.3.1.1 FEP Conservation Measures

Bottomfish, precious coral, coral reef and crustacean fisheries managed under this FEP have not had reported interactions with protected species, and no specific regulations are in place to mitigate protected species interactions. Destructive gear such as bottom trawls, bottom gillnets, explosives and poisons are prohibited under this FEP, and these provide benefit to protected species by preventing potential interactions with non-selective fishing gear.

2.3.1.2 ESA Consultations

ESA consultations were conducted by NMFS and the U.S. Fish and Wildlife Service (for species under their jurisdiction) to ensure ongoing fisheries operations managed under the PRIA FEP are not jeopardizing the continued existence of any listed species or adversely modifying critical habitat. The results of these consultations, conducted under section 7 of the ESA, are briefly described below and summarized in Table 3.

Table 3. Summary of ESA consultations for PRIA FEP Fisheries

Fishery	Consultation date	Consultation type[a]	Outcome[b]	Species

Fishery	Consultation date	Consultation type[a]	Outcome[b]	Species
Bottomfish	3/8/2002	BiOp	NLAA	Loggerhead sea turtle, leatherback sea turtle, olive ridley sea turtle, green sea turtle, hawksbill sea turtle, humpback whale, blue whale, fin whale, sei whale, sperm whale
Coral reef ecosystem	3/7/2002	LOC	NLAA	Loggerhead sea turtle, leatherback sea turtle, olive ridley sea turtle, green sea turtle, hawksbill sea turtle, humpback whale, blue whale, fin whale, sei whale, sperm whale
	5/22/2002	LOC (USFWS)	NLAA	Green, hawksbill, leatherback, loggerhead and olive ridley turtles, Newell's shearwater, short-tailed albatross, Laysan duck, Laysan finch, Nihoa finch, Nihoa millerbird, Micronesian megapode, 6 terrestrial plants
Crustacean	9/28/2007	LOC	NLAA	Loggerhead sea turtle, leatherback sea turtle, olive ridley sea turtle, green sea turtle, hawksbill sea turtle, humpback whale, blue whale, fin whale, sei whale, sperm whale
Precious coral	10/4/1978	BiOp	Does not constitute threat	Sperm whale, leatherback sea turtle
	12/20/2000	LOC	NLAA	Humpback whale, green sea turtle, hawksbill sea turtle
All fisheries	1/16/2015	No effect memo	No effect	Reef-building corals
	2/20/2015	LOC	NLAA	Scalloped hammerhead shark (Indo-west Pacific DPS)

[a] BiOp = Biological Opinion; LOC = Letter of Concurrence; BE = Biological Evaluation.

[b] LAA = likely to adversely affect; NLAA = not likely to adversely affect.

Bottomfish Fishery

In a biological opinion issued on March 3, 2002, NMFS concluded that the ongoing operation of the Western Pacific Region’s bottomfish and seamount fisheries is not likely to jeopardize the continued existence of five sea turtle species (loggerhead, leatherback, olive ridley, green and hawksbill turtles) and five marine mammal species (humpback, blue, fin, sei and sperm whales) (NMFS 2002a). NMFS also concluded in an informal consultation dated February 20, 2015 that fisheries managed under the PRIA FEP are not likely to adversely affect the Indo-West Pacific DPS of scalloped hammerhead shark. NMFS also concluded on January 16, 2015 that fisheries managed under the PRIA FEP have no effects on ESA-listed reef-building corals.

Crustacean Fishery

An informal consultation completed by NMFS on September 28, 2007 concluded that PRIA crustacean fisheries are not likely to adversely affect five sea turtle species (loggerhead, leatherback, olive ridley, green and hawksbill turtles) and five marine mammal species (humpback, blue, fin, sei and sperm whales). NMFS also concluded in an informal consultation dated February 20, 2015 that fisheries managed under the PRIA FEP are not likely to adversely affect the Indo-West Pacific DPS of scalloped hammerhead shark. NMFS also concluded on January 16, 2015 that fisheries managed under the PRIA FEP have no effects on ESA-listed reef-building corals.

Coral Reef Fishery

An informal consultation completed by NMFS on March 7, 2002 concluded that fishing activities conducted under the Coral Reef Ecosystems FMP are not likely to adversely affect five sea turtle species (loggerhead, leatherback, olive ridley, green and hawksbill turtles) and five marine mammal species (humpback, blue, fin, sei and sperm whales). NMFS also concluded in an informal consultation dated February 20, 2015 that fisheries managed under the PRIA FEP are not likely to adversely affect the Indo-West Pacific DPS of scalloped hammerhead shark. NMFS also concluded on January 16, 2015 that fisheries managed under the PRIA FEP have no effects on ESA-listed reef-building corals.

On May 22, 2002, the USFWS concurred with the determination of NMFS that the activities conducted under the Coral Reef Ecosystems FMP are not likely to adversely affect listed species under USFWS's exclusive jurisdiction (i.e., seabirds and terrestrial plants) and listed species shared with NMFS (i.e., sea turtles).

Precious Coral Fishery

An informal consultation completed by NMFS on December 20, 2000 concluded that PRIA precious coral fisheries are not likely to adversely affect humpback whales, green turtles or hawksbill turtles. NMFS also concluded in an informal consultation dated February 20, 2015 that fisheries managed under the PRIA FEP are not likely to adversely affect the Indo-West Pacific DPS of scalloped hammerhead shark. NMFS also concluded on January 16, 2015 that fisheries managed under the PRIA FEP have no effects on ESA-listed reef-building corals.

2.3.1.3 Non-ESA Marine Mammals

The MMPA requires NMFS to annually publish a List of Fisheries (LOF) that classifies commercial fisheries in one of three categories based on the level of mortality and serious injury of marine mammals associated with that fishery. PRIA fisheries are not classified under the LOF.

2.3.1 Status of Protected Species Interactions in the PRIA FEP Fisheries

There are currently no bottomfish, crustacean, coral reef or precious coral fisheries operating in the PRIA, and no historical observer data are available for fisheries under this FEP. No new fishing activity has been reported, and there is no other information to indicate that impacts to protected species from PRIA fisheries have changed in recent years.

2.3.2 Identification of Emerging Issues

Several species are currently candidates for listing under the ESA, and several more ESA-listed species are being evaluated for critical habitat designation (

Table **4**). If these species are listed or critical habitat are designated, they will be included in this SAFE report and impacts from FEP-managed fisheries will be evaluated under applicable mandates.

Table 4. Candidate ESA species, and ESA-listed species being evaluated for critical habitat designation.

Species		Listing process			Post-listing activity	
Common name	**Scientific name**	**90-day finding**	**12-month finding / Proposed rule**	**Final rule**	**Critical Habitat**	**Recovery Plan**
Oceanic whitetip shark	*Carcharhinus longimanus*	Positive (81 FR 1376, 1/12/2016)	Positive, threatened (81 FR 96304, 12/29/2016)	Public comment period closed 3/29/2017, final rule expected 12/29/2017	N/A	N/A
Pacific bluefin tuna	*Thunnus orientalis*	Positive (81 FR 70074, 10/11/2016)	In progress, expected 6/2017	N/A	N/A	N/A
Chambered nautilus	*Nautilus pompilius*	Positive (81 FR 58895, 8/26/2016)	In progress, expected 5/2017	N/A	N/A	N/A
Giant manta ray	*Manta birostris*	Positive (81 FR 8874, 2/23/2016)	Positive, threatened (82 FRN 3694, 1/12/2017)	Public comment period closed 3/13/2017, final rule expected 1/2018	N/A	N/A
Reef manta ray	*Manta alfredi*	Positive (81 FR 8874, 2/23/2016)	Not warranted (82 FRN 3694, 1/12/2017)	N/A	N/A	N/A
Corals	N/A	Positive for 82 species (75 FR 6616, 2/10/2010)	Positive for 66 species (77 FR 73219, 12/7/2012)	20 species listed as threatened (79 FR 53851, 9/10/2014)	In development, proposal expected 2017	In development, expected TBA, interim recovery outline in place
Green sea turtle	*Chelonia mydas*	Positive (77 FR 45571, 8/1/2012)	Identification of 11 DPSs, endangered and threatened (80 FR 15271, 3/23/2015)	11 DPSs listed as endangered and threatened (81 FR 20057, 4/6/2016)	In development, proposal expected 2017	TBA

2.3.3 Identification of research, data and assessment needs

The following research, data and assessment needs for insular fisheries were identified by the Council's Protected Species Advisory Committee and Plan Team:

- Improve the precision of non-commercial fisheries data to improve understanding of potential protected species impacts.

- Define and evaluate innovative approaches to derive robust estimates of protected species interactions in insular fisheries.

2.4 Climate and Oceanic Indicators

2.4.1 Introduction

The 2016 Annual Report includes a chapter on indicators of current and changing climate and related oceanic conditions in the geographic areas for which the Western Pacific Regional Fishery Management Council has responsibility. There are a number of reasons for the Council's decision to provide and maintain an evolving discussion of climate conditions as an integral and continuous consideration in their deliberations, decisions and reports:

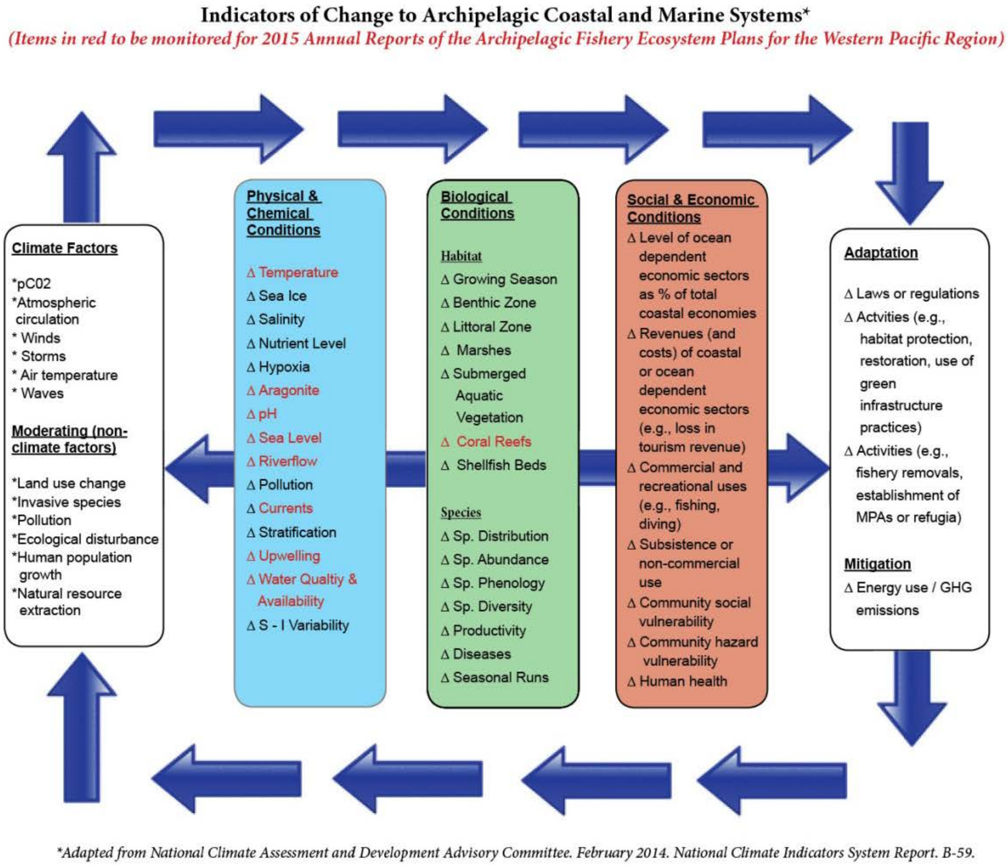
- Emerging scientific and community understanding of the impacts of changing climate conditions on fishery resources, the ecosystems that sustain those resources and the communities that depend upon them;
- Recent Federal Directives including the 2010 implementation of a National Ocean Policy that identified Resiliency and Adaptation to Climate Change and Ocean Acidification as one of nine National priorities; the development of a Climate Science Strategy by the National Marine Fisheries Service (NMFS) in 2015 and the ongoing development of Pacific Regional Climate Science program;
- The Council's own engagement with the National Oceanic and Atmospheric Administration (NOAA) as well as jurisdictional fishery management agencies in American Samoa, the Commonwealth of the Northern Mariana Islands, Guam, Hawaii as well as fishing industry representatives and local communities in those jurisdictions; and
- Deliberations of the Council's Marine Planning and Climate Change Committee.

Beginning with the 2015 Report, the Council and its partners have described changes in a series of climate and oceanic indicators that will grow and evolve over time as they become available and their relevance to Western Pacific fishery resources becomes clear.

2.4.2 Conceptual Model

In developing this chapter, the Council relied on a number of recent reports conducted in the context of the U.S. National Climate Assessment including, most notably, the 2012 Pacific Islands Regional Climate Assessment (PIRCA) and the Ocean and Coasts chapter of the 2014 report on a Pilot Indicator System prepared by the National Climate Assessment and Development Advisory Committee (NCADAC).

The Advisory Committee Report presented a possible conceptual framework designed to illustrate how climate factors can connect to and interact with other ecosystem components to ocean and coastal ecosystems and human communities. The Council adapted this model with considerations relevant to the fishery resources of the Western Pacific Region:



Indicators of Change to Archipelagic Coastal and Marine Systems*

(Items in red to be monitored for 2015 Annual Reports of the Archipelagic Fishery Ecosystem Plans for the Western Pacific Region)

**Adapted from National Climate Assessment and Development Advisory Committee. February 2014. National Climate Indicators System Report. B-59.*

Figure 5. Indicators of change to archipelagic coastal and marine systems.

As described in the 2014 NCADAC report, the conceptual model represents a “simplified representation of climate and non-climate stressors in coastal and marine ecosystems.” For the purposes of this Annual Report, the modified Conceptual Model allows the Council and its partners to identify indicators of interest to be monitored on a continuing basis in coming years. The indicators shown in red were considered for inclusion in the 2016 Annual Report; the specific indicators used in the Report are listed in Section 2.3. Other indicators will be added over time as datasets become available and understanding of the nature of the causal chain from stressors to impacts emerges.

The Council also hopes that this Conceptual Model can provide a guide for future monitoring and research that will enable the Council and its partners to move from observations and correlations to understanding the specific nature of interactions and developing capabilities to predict future changes of importance in developing, evaluating and adapting ecosystem-fishery plans in the Western Pacific Region.

2.4.3 Selected Indicators

The primary goal for selecting the Indicators used in this (and future reports) is to provide fisheries-related communities, resource managers and businesses with a climate-related situational awareness. In this context, Indicators were selected to:

- Be fisheries-relevant and informative
- Build intuition about current conditions in light of changing climate
- Provide historical context and
- Recognize patterns and trends.

For the 2016 report on Western Pacific Pelagic resources, the Council has included the following climate and oceanic indicators:

Atmospheric Carbon Dioxide (at Mauna Loa Observatory) --Increasing atmospheric CO_2 is a primary measure of anthropogenic climate change.

Ocean pH (at Station ALOHA) – Ocean pH provides a measure of ocean acidification. Increasing ocean acidification limits the ability of marine organisms to build shells and other hard structures.

Oceanic Niño Index (ONI) – Sea surface temperature anomaly from Niño 3.4 region (5°N - 5°S, 120° - 170°W). This index is used to determine the phase of the El Niño – Southern Oscillation (ENSO), which has implications across the region affecting migratory patterns of key commercial fish stocks which, in turn, affect the location, safety and costs of commercial fishing.

Pacific Decadal Oscillation (PDO) – Sea surface temperature anomaly from Niño 3.4 region (5°N - 5°S, 120° - 170°W). This index is used to determine the phase of the El Niño – Southern Oscillation (ENSO), which has implications across the region affecting migratory patterns of key commercial fish stocks which, in turn, affect the location, safety and costs of commercial fishing.

Sea Surface Temperature – Monthly sea surface temperature anomaly from 2003-2015 from the AVHRR instrument aboard the NOAA Polar Operational Environmental Satellite (POES). Sea surface temperature is one of the most directly observable measures we have for tracking increasing ocean temperature.

Sea Surface Temperature Anomaly – Sea surface temperature anomaly highlights long term trends. Filtering out seasonal cycle is one of the most directly observable measures we have for tracking increasing ocean temperature.

Degree Heating Weeks (DHW) – DHW from the CoralReefWatch team provide the best available metric to track exposure of coral reef ecosystems to anomalously high temperature events

Sea Level (Sea Surface Height) and Anomaly – Rising sea levels can result in a number of coastal impacts, including inundation of infrastructure, increased damage resulting from storm-driven waves and flooding, and saltwater intrusion into freshwater supplies. NOTE that no water level gauges are available in Pacific Remote Island Areas (PRIA) so only regional information on this indicator are included.

Heavy Weather (Tropical Cyclones) – Measures of tropical cyclone occurrence, strength, and energy. Tropical cyclones have the potential to significantly impact fishing operations.

Wave Data – To describe patterns in wave forcing, we present data from the Wave Watch 3 global wave model run by the Department of Ocean and Resources Engineering at the University of Hawaiʻi in collaboration with NOAA/NCEP and NWS Honolulu. Wave forcing can have major implications for both coastal ecosystems and pelagic fishing operations.

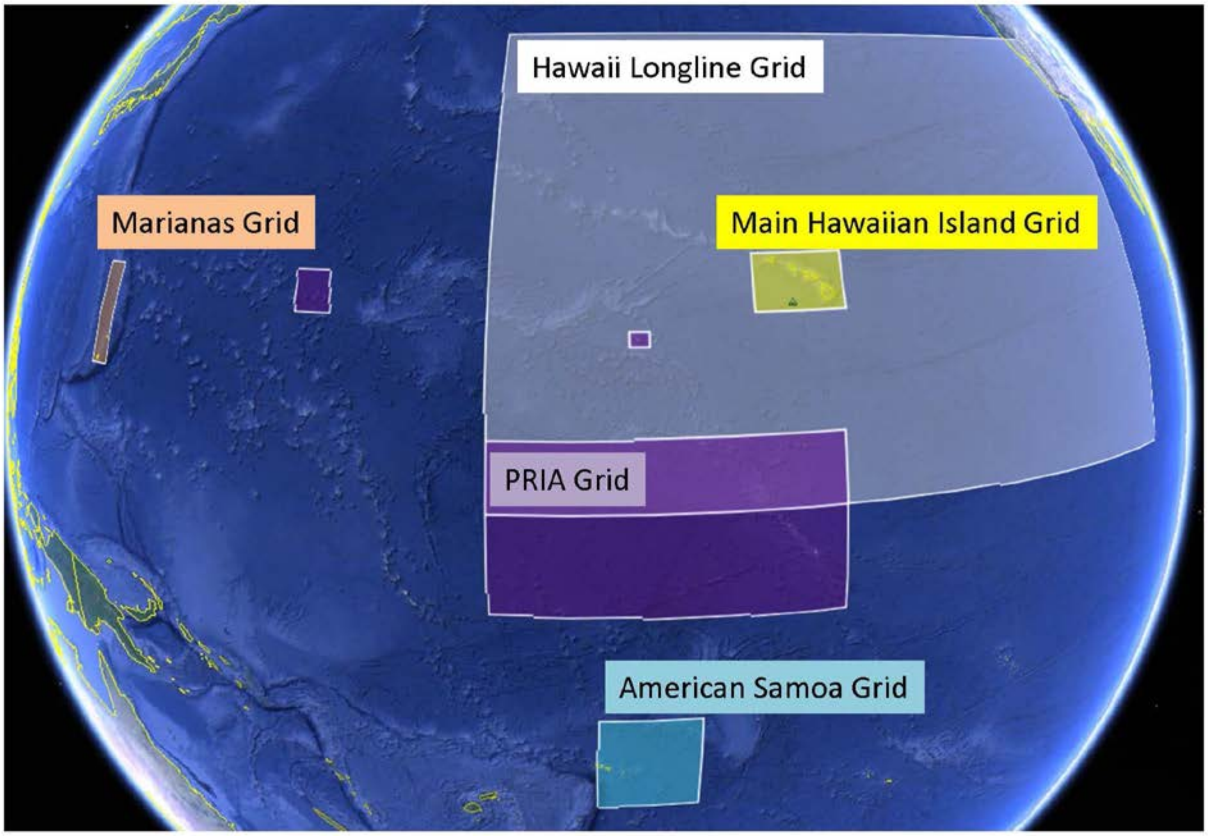


Figure 6. Regional Spatial Grids

Table 5. PRIA climate and ocean indicator summary.

Indicator	Definition and Rationale	Indicator Status
Atmospheric Concentration of Carbon Dioxide (CO_2)	Atmospheric concentration CO_2 at Mauna Loa Observatory. Increasing atmospheric CO_2 is a primary measure of anthropogenic climate change.	Trend: increasing exponentially 2017: time series maximum 406.43 ppm
Oceanic pH	Ocean surface pH at Station ALOHA. Ocean pH provides a measure of ocean acidification. Increasing ocean acidification limits the ability of marine organisms to build shells and other hard structures.	Trend: pH is decreasing at a rate of 0.039 pH units per year, equivalent to 0.4% increase in acidity per year
Oceanic Niño Index (ONI)	Sea surface temperature anomaly from Niño 3.4 region (5°N - 5°S, 120° - 170°W). This index is used to determine the phase of the El Niño – Southern Oscillation (ENSO), which has implications across the region, affecting migratory patterns of key commercial fish stocks which in turn affect the location, safety, and costs of commercial fishing.	2015: Strong El Niño 2016: weak La Niña dissipating, potential rapid return to El Niño
Pacific Decadal Oscillation (PDO)	The Pacific Decadal Oscillation (PDO) Index is defined as the leading principal component of North Pacific monthly sea surface temperature variability (poleward of 20N for the 1900-93 period).	2016: Strong Positive Phase
Sea Surface Temperature[1] (SST)	Satellite remotely-sensed sea surface temperature. SST is projected to rise, and impacts phenomena ranging from winds to fish distribution.	SST in waters surrounding Palmyra ranged between 28.4-28.6^o while Howland and Baker ranged between 28.6-28.8^o Wake Islands between 27.5-28.0^{o}C Johnston between 27.0-27.25^{o}C with West Johnston being the warmest Showing positive anomalies in all PRIA locations
Degree Heating Weeks (DHW)	Satellite remotely-sensed sea surface temperature, transformed to a metric relevant for coral bleaching. Each degree heating week indicates a one degree excess over long term summer means (Maximum Monthly Mean SST), that persists for a week. At 4 DHW, bleaching is expected, at 8 DHW bleaching is expected to be widespread and to induce mortality.	2015 showed extreme high temperature anomalies, with values surpassing 20 DHW in the equatorial regions.
Tropical Cyclones	Measures of tropical cyclone occurrence, strength, and energy. Tropical cyclones have the potential to significantly	Eastern Pacific, 2016: 21 named storms, 11 hurricanes, 5 major.

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[1] 2016 data are incomplete.

	impact fishing operations.	Central Pacific, 2016: 7 named storms, 3 hurricanes, 2 major. Western Pacific 2016: 26 named storms, 13 typhoons, 6 major
Sea Level/Sea Surface Height	Monthly mean sea level time series, including extremes. Data from satellite altimetry & in situ tide gauges. Rising sea levels can result in a number of coastal impacts, including inundation of infrastructure, increased damage resulting from storm-driven waves and flooding, and saltwater intrusion into freshwater supplies.	No tide gauge data for the Pacific Remote Island Area.
Wave Energy	WaveWatch III (WW3) Global Wave Model" run by UH Department of Ocean Resources & Engineering in collaboration with NOAA/NCEP & NOAA/NWS-Pacific Wave forcing can have major implications for both coastal ecosystems and pelagic fishing operations.	Significant wave heights ranged between 1.5-1.8m for Palmyra; 1.8-2.5m for Howland & Baker and 1.8-2.1m for Jarvis. Significant wave heights for Wake ranged from 2.1-2.4m and for Johnston 2.4-2.7m

2.4.3.1 Atmospheric Concentration of Carbon Dioxide (CO_2)) Mauna Loa.

Description: Monthly mean atmospheric carbon dioxide at Mauna Loa Observatory, Hawaii in ppm from March 1958 to present. The carbon dioxide data is measured as the mole fraction in dry air, on Mauna Loa. A dry mole fraction is defined as the number of molecules of carbon dioxide divided by the number of molecules of dry air multiplied by one million (ppm). This constitutes the longest record of direct measurements of CO_2 in the atmosphere. The measurements were started by C. David Keeling of the Scripps Institution of Oceanography in March of 1958 at a facility of the National Oceanic and Atmospheric Administration [Keeling, 1976]. NOAA started its own CO_2 measurements in May of 1974, and they have run in parallel with those made by Scripps since then [Thoning, 1989].

The observed increase in monthly average carbon dioxide data is due primarily to CO_2 emissions from fossil fuel burning. Carbon dioxide remains in the atmosphere for a very long time, and emissions from any location mix throughout the atmosphere in about one year. The annual oscillations at Mauna Loa, Hawaii are due to the seasonal imbalance between the photosynthesis and respiration of plants on land. During the summer photosynthesis exceeds respiration and CO_2 is removed from the atmosphere, whereas outside the growing season respiration exceeds photosynthesis and CO_2 is returned to the atmosphere. The seasonal cycle is strongest in the northern hemisphere because of the presence of the continents. The difference in CO_2 between Mauna Loa and the South Pole has increased over time as the global rate of fossil fuel burning, most of which takes place in the northern hemisphere, has accelerated.

Timeframe: Yearly (by month)

Region/Location: Hawaii but representative of global concentration of carbon dioxide.

Data Source: "Full Mauna Loa CO_2 record" at http://www.esrl.noaa.gov/gmd/ccgg/trends/ , NOAA ESRL Global Monitoring Division. The National Oceanic and Atmospheric Administration (NOAA) Global Monitoring Division provides high-precision measurements of the abundance and distribution of long-lived greenhouse gases that are used to calculate global average concentrations.

Measurement Platform: In-situ Station

Rationale: Atmospheric carbon dioxide is a measure of what human activity has already done to affect the climate system through greenhouse gas emissions. It provides quantitative information in a simplified, standardized format that decision makers can easily understand. This indicator demonstrates that the concentration (and, in turn, the warming influence) of greenhouse gases in the atmosphere has increased substantially over the last several decades. In January of 2017, the monthly mean concentration of $C0_2$ was 406.43 ppm. In January of 1959, the onset year, it was 315.62 ppm. It passed 350 ppm in 1988.

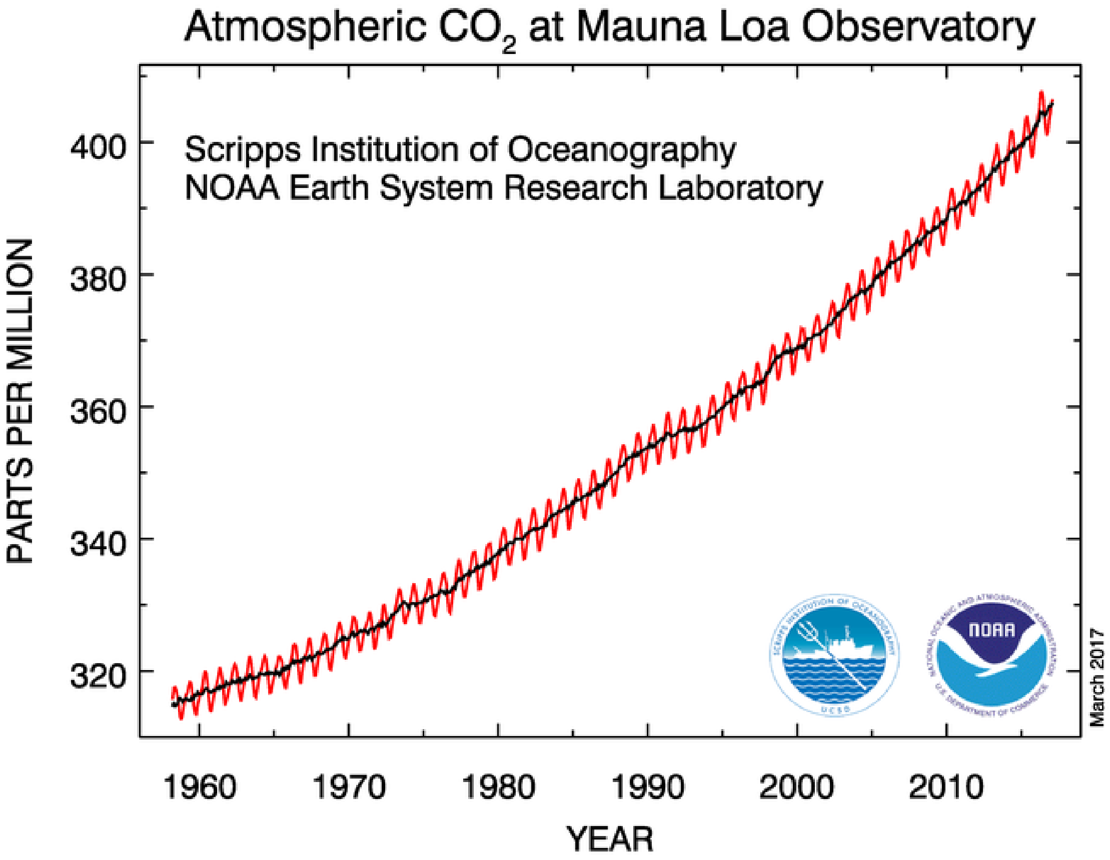


Figure 7. Monthly mean atmospheric carbon dioxide at Mauna Loa Observatory, Hawaii. The carbon dioxide data (red curve), measured as the mole fraction (ppm) in dry air, on Mauna Loa. The black curve represents the seasonally corrected data.

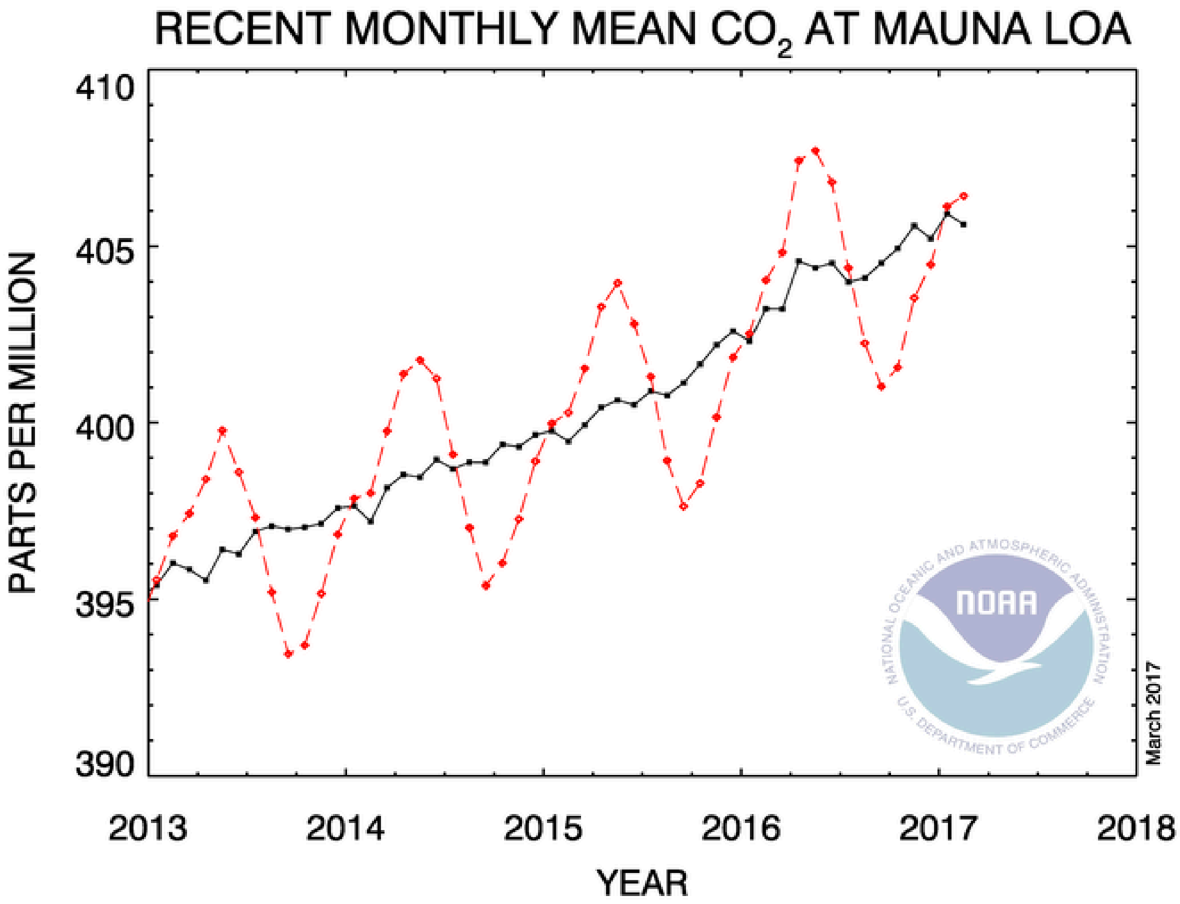


Figure 8. Monthly mean atmospheric carbon dioxide at Mauna Loa Observatory, Hawaii, 2013-2017. The carbon dioxide data (red curve), measured as the mole fraction (ppm) in dry air, on Mauna Loa. The black curve represents the seasonally corrected data.

2.4.3.2 Ocean pH:

Description: Trends in surface (0-10m) pH and pCO2 at Station ALOHA, North of Oahu (22° 45’ N, 158° W), collected by the Hawai‘i Ocean Time-series (HOT). Red dots represent directly measured pH, blue dots represent pH calculated from total alkalinity (TA) and dissolved inorganic carbon (DIC).

The 25+ year time-series at Station ALOHA represents the best available documentation of the significant downward trend of ocean pH since 1989. Actual ocean pH varies in both time and space, but over last 25 years, the HOTS Station ALOHA time series has shown a significant linear decrease of -0.0386 pH units, or roughly a 9% increase in acidity ([H+]) over that period. With the new year of data added since the last SAFE report (i.e. 2015 data), this declining trend continues.

Timeframe: Updated Monthly

Region/Location: North Oahu.

Data Source/Responsible Party: Hawai‘i Ocean Time Series. (http://hahana.soest.hawaii.edu/hot/)

Measurement Platform: Oceanographic research station, shipboard collection.

Rationale: Increasing ocean acidification affects coral reef growth and health, which in turn affects the health of coral reef ecosystems and the ecosystems and resources that they sustain. Monitoring pH on a continuous basis provides a foundational basis for documenting, understanding and, ultimately, predicting the effects of ocean acidification.

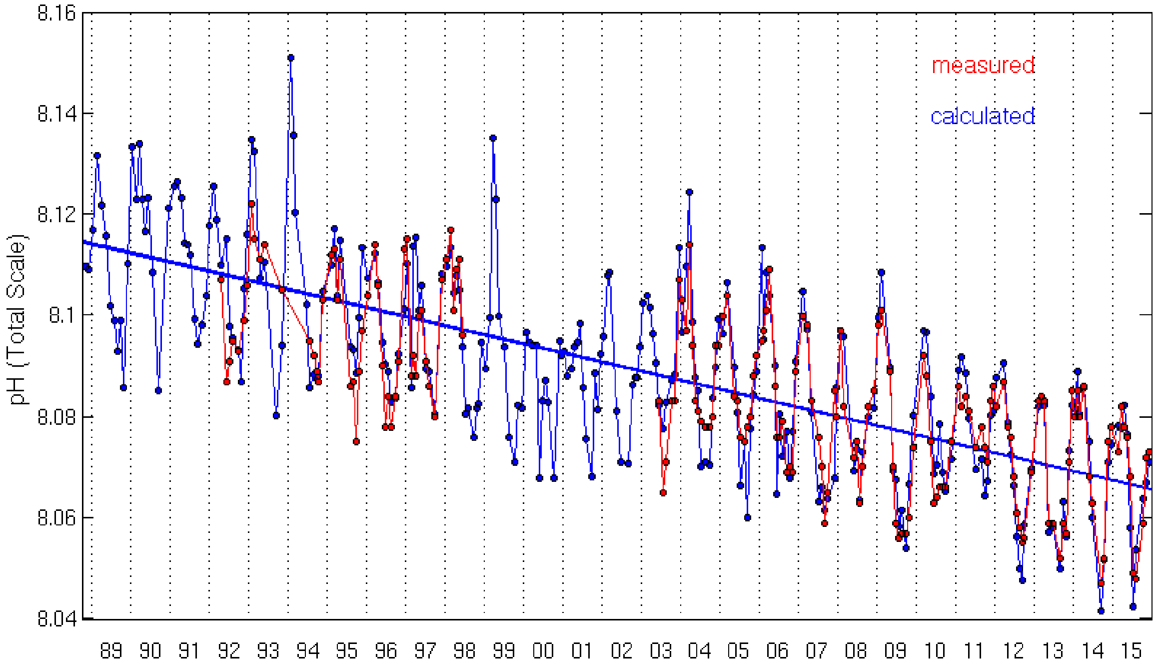


Figure 9. pH Trend at Station Aloha, 1989-2015.

2.4.3.3 Oceanic Niño Index (ONI)

Description: Warm (red) and cold (blue) periods based on a threshold of +/- 0.5^{o}C for the Oceanic Niño Index (ONI) [three-month running mean of ERSST.v4 SST anomalies in the Niño 3.4 region (5^{o}N-5^{o}S, 120^{o}-170^{o}W)], based on centered 30-year base periods updated every five years.

For historical purposes, periods of below and above normal sea surface temperatures (SSTs) are colored in blue and red when the threshold is met for a minimum of five consecutive overlapping seasons. The ONI is one measure of the El Niño-Southern Oscillation, and other indices can confirm whether features consistent with a coupled ocean-atmosphere phenomenon accompanied these periods.

Description was inserted from:
http://www.cpc.ncep.noaa.gov/products/analysis_monitoring/ensostuff/ensoyears.shtml

Timeframe: Every three months.

Region/Location: Niño 3.4 Region: 5°S - 5°N, 120°-170°W

Data Source/Responsible Party: NOAA NCEI Equatorial Pacific Sea Surface Temperatures (www.ncdc.noaa.gov/teleconnections/enso/indicators/sst.php)

Measurement Platform: In-situ Station, Satellite, Model, Other…

Rationale:

The ONI focuses on ocean temperature which has the most direct effect on those fisheries. The atmospheric half of this Pacific basin oscillation is measured using the Southern Oscillation Index.

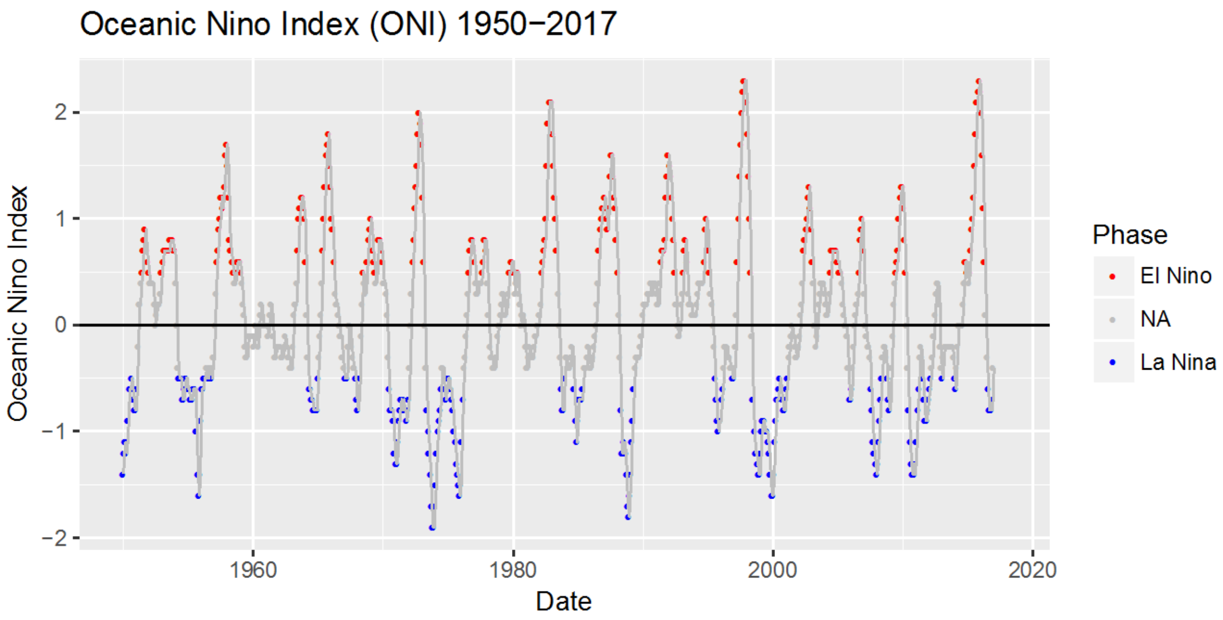


Figure 10. Oceanic Nino Index, 1950-2017.

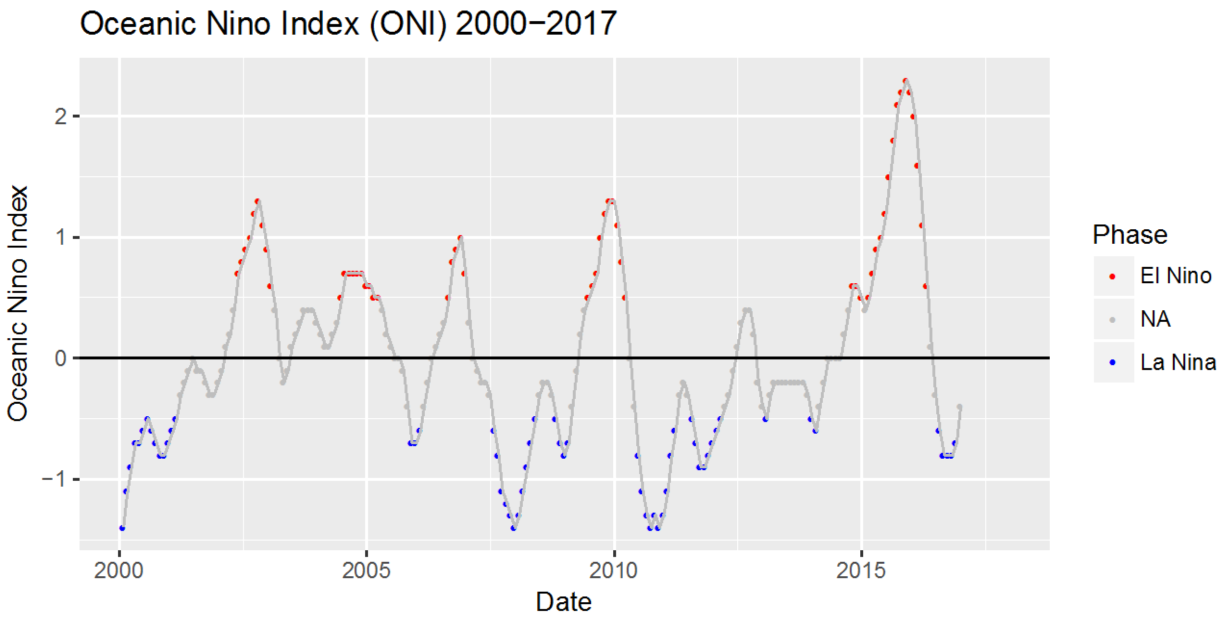


Figure 11. Oceanic Nino Index, 2000-2017.

2.4.3.4 Pacific Decadal Oscillation (PDO)

Description: The "Pacific Decadal Oscillation" (PDO) is a long-lived El Niño-like pattern of Pacific climate variability. While the two climate oscillations have similar spatial climate fingerprints, they have very different behavior in time. Fisheries scientist Steven Hare coined the term "Pacific Decadal Oscillation" (PDO) in 1996 while researching connections between Alaska salmon production cycles and Pacific climate (his dissertation topic with advisor Robert Francis). Two main characteristics distinguish PDO from El Niño/Southern Oscillation (ENSO): first, 20th century PDO "events" persisted for 20-to-30 years, while typical ENSO events persisted for 6 to 18 months; second, the climatic fingerprints of the PDO are most visible in the North Pacific/North American sector, while secondary signatures exist in the tropics - the opposite is true for ENSO. Several independent studies find evidence for just two full PDO cycles in the past century: "cool" PDO regimes prevailed from 1890-1924 and again from 1947-1976, while "warm" PDO regimes dominated from 1925-1946 and from 1977 through (at least) the mid-1990's. Shoshiro Minobe has shown that 20th century PDO fluctuations were most energetic in two general periodicities, one from 15-to-25 years, and the other from 50-to-70 years.

Major changes in northeast Pacific marine ecosystems have been correlated with phase changes in the PDO; warm eras have seen enhanced coastal ocean biological productivity in Alaska and inhibited productivity off the west coast of the contiguous United States, while cold PDO eras have seen the opposite north-south pattern of marine ecosystem productivity.

Causes for the PDO are not currently known. Likewise, the potential predictability for this climate oscillation are not known. Some climate simulation models produce PDO-like oscillations, although often for different reasons. The mechanisms giving rise to PDO will determine whether skillful decades-long PDO climate predictions are possible. For example, if PDO arises from air-sea interactions that require 10 year ocean adjustment times, then aspects of the phenomenon will (in theory) be predictable at lead times of up to 10 years. Even in the absence of a theoretical understanding, PDO climate information improves season-to-season and year-to-year climate forecasts for North America because of its strong tendency for multi-season and multi-year persistence. From a societal impacts perspective, recognition of PDO is important because it shows that "normal" climate conditions can vary over time periods comparable to the length of a human's lifetime.

[Description inserted from: http://research.jisao.washington.edu/pdo/]

Timeframe: Monthly.

Region/Location: North Pacific

Data Source/Responsible Party: Joint Institute for the Study of the Atmosphere and Ocean (JISAO, UW) (http://research.jisao.washington.edu/pdo/PDO.latest.txt)

Measurement Platform: In-situ Station, Satellite, Model, Other…

Rationale:

The Pacific Decadal Oscillation (PDO) Index is defined as the leading principal component of North Pacific monthly sea surface temperature variability (poleward of 20N for the 1900-93 period). Digital values of our PDO index are available from Nate Mantua's anonymous ftp directory (linked here). Please send email to Nate (nate.mantua@noaa.gov) or Steven Hare (hare@iphc.washington.edu) to let them know that you have obtained this data. Nate updates the PDO index every two or three months.

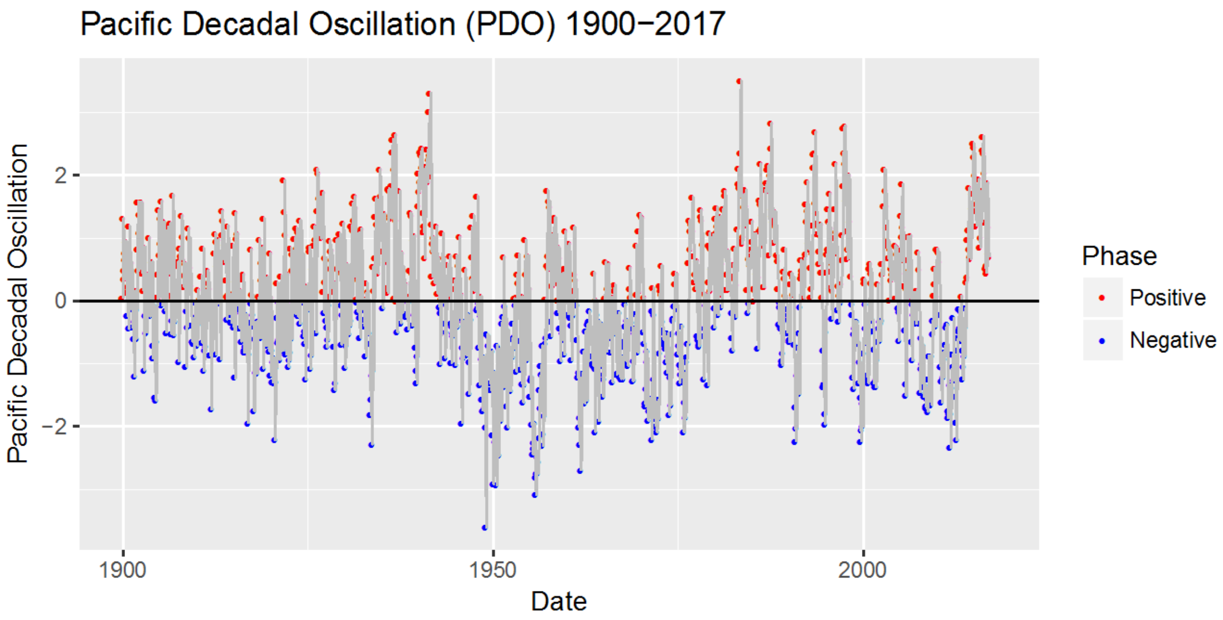


Figure 12. Pacific Decadal Oscillation, 1900-2017.

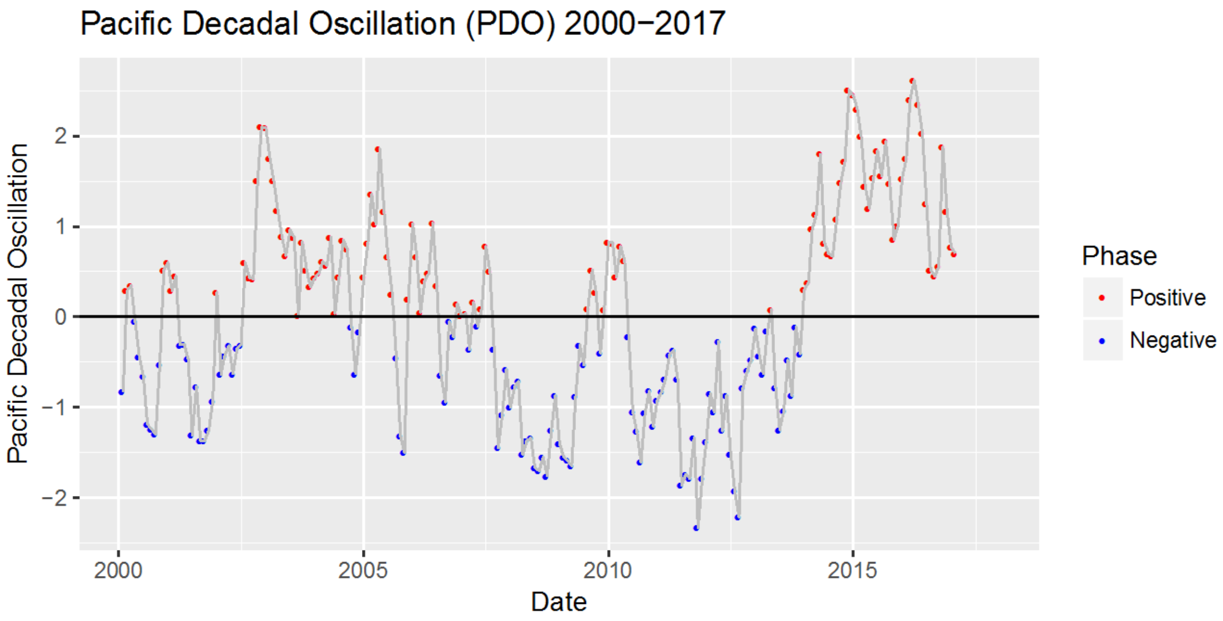
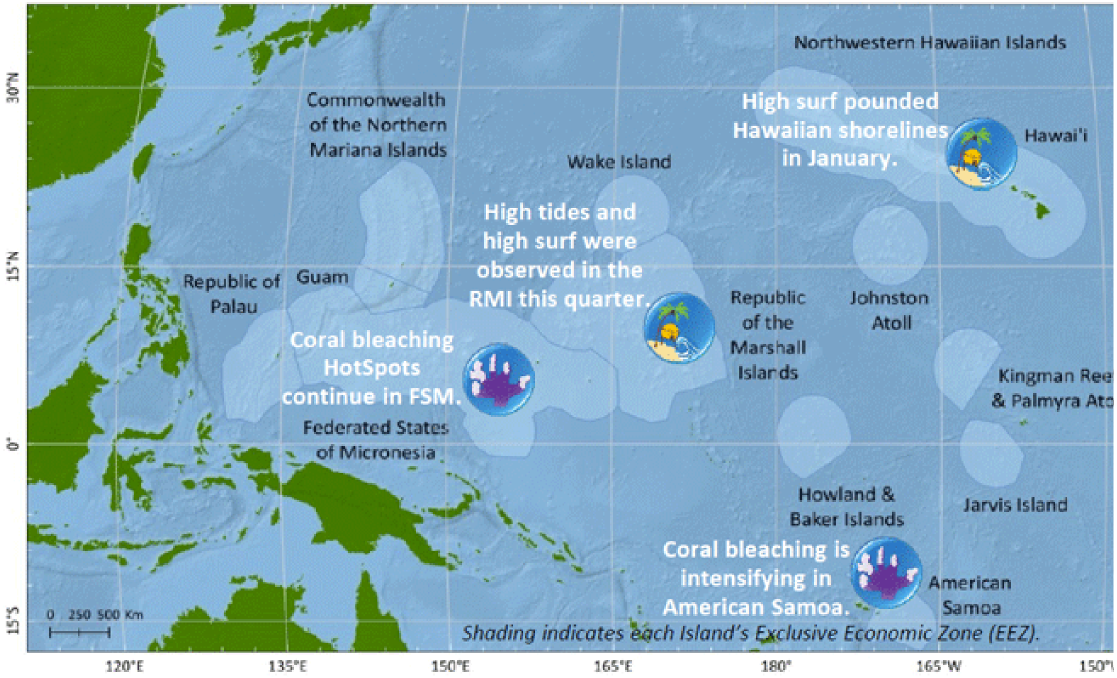


Figure 13. Pacific Decadal Oscillation, 2000-2017.

The Climate Impacts and Outlook Q4 2016

From: http://www.pacificcis.org/dashboard



High surf pounded Hawaiian shorelines in January.

High tides and high surf were observed in the RMI this quarter.

Coral bleaching HotSpots continue in FSM.

Coral bleaching is intensifying in American Samoa.

Shading indicates each Island's Exclusive Economic Zone (EEZ).

Significant Events and Archipelagic Impacts

Near-normal rainfall was recorded in parts of the Common wealth of the Northern Mariana Islands, while above-normal rainfall was reported in Guam. Much below normal rainfall was reported in Hawaii, while most of the Federated States of Micronesia, the Republic of Palau, and the Marshall Islands were above normal. Near normal rains were observed in American Samoa.

There were a total of 31 tropical cyclones in the western North Pacific during 2016.

***Facilities and Infrastructure* –** A series of large NW swell events in early-to-mid November led to sharp erosion at Sunset Beach, north shore, Oahu. The high swell and morning high tide of 2016-11-14 allowed wave run-up to cross the highway in Waianae, west shore, Oahu and select sections of the coastal highway on the north shore of Oahu. Coastal wave run-up was also high enough to cross select sections of the highway on the north shore 2017-01-13, -25, and -30. Meanwhile, gale-force trade winds 2017-01-21 and -22 had seas to 17 feet as measured by a wave buoy off Kailua, Oahu. It caused minor coastal wave run-up.

***Water Resources* –** Despite high surf, high tides, and higher than normal sea levels, long-period swell did not affect the capital of the RMI during the quarter. Water reservoir levels in the Majuro, FSM, and Koror remain adequate with regularly-occurring rains, however the northern Marshall Islands are very dry.

***Natural Resources* –** Eddy kinetic energy near Hawaii has been unusually high over the last quarter. Eddies (gyres) have important biological implications in that they can drive upwelling of cooler, nutrient rich water that influences ocean temperatures and fuels a localized increase in phytoplankton production, an essential source of energy for higher trophic groups. In American Samoa, coral bleaching patterns are evident on

Figure 14. Q4 2016 Climate Impact and Outlook Infographic.

2.4.3.5 Sea Surface Temperature

Description: Monthly sea surface temperature from 2003-2016 from the Advanced Very High Resolution Radiometer (AVHRR) instrument aboard the NOAA Polar Operational Environmental Satellite (POES). These data take us back to 2003. If we were to blend this record with Pathfinder, we could reach back to 1981.

Background Below Inserted From *CoastWatch West Coast Node.* We would like to acknowledge the NOAA CoastWatch Program and the NOAA NWS Monterey Regional Forecast Office.

Short Description: The global area coverage (GAC) data stream from NOAA | NESDIS | OSDPD provides a high-quality sea surface temperature product with very little cloud contamination. This data is used for a variety of fisheries management projects, including the El Niño Watch Report, which stress data quality over high spatial resolution.

Technical Summary: CoastWatch offers global sea surface temperature (SST) data from the Advanced Very High Resolution Radiometer (AVHRR) instrument aboard NOAA's Polar Operational Environmental Satellites (POES). Two satellites are currently in use, NOAA-17 and NOAA-18. The AVHRR sensor is a five-channel sensor comprised of two visible radiance channels and three infrared radiance channels. During daytime satellite passes, all five radiance channels are used. During nighttime passes, only the infrared radiance channels are used.

The POES satellite stores a sub-sample of the AVHRR radiance measurements onboard, generating a global data set. The satellite downloads this dataset once it is within range of a receiving station. The sub-sampling reduces the resolution of the original data from 1.47km for the HRPT SST product to 11km for the global data product.

AVHRR radiance measurements are processed to SST by NOAA's National Environmental Satellite, Data, and Information Service (NESDIS), Office of Satellite Data Processing and Distribution (OSDPD) using the non-linear sea surface temperature (NLSST) algorithm detailed in *Walton et al., 1998*. SST values are accurate to within 0.5 degrees Celsius. Ongoing calibration and validation efforts by NOAA satellites and information provide for continuity of quality assessment and algorithm integrity (e.g., *Li et al., 2001a and Li et al., 2001b*). In addition, the CoastWatch West Coast Regional Node (WCRN) runs monthly validation tests for all SST data streams using data from the NOAA National Weather Service and National Data Buoy Center (NDBC).

The data are cloud screened using the CLAVR-x method developed and maintained by NOAA Satellites and Information (e.g., *Stowe et al., 1999*). The data are mapped to an equal angle grid (0.1 degrees latitude by 0.1 degrees longitude) using a simple arithmetic mean to produce individual and composite images of various durations (e.g., 1, 3, 8, 14-day).

Timeframe: 2003-2016. Daily data available. Monthly means shown.

Region/Location: Global.

Data Source: "SST, POES AVHRR, GAC, Global, Day and Night (Monthly Composite)" http://coastwatch.pfeg.noaa.gov/erddap/griddap/erdAGsstamday.html.

Measurement Platform: *AVHRR, POES Satellite*

Rationale: Sea surface temperature is one of the most directly observable measures we have for tracking increasing ocean temperature.

References: Li, X., W. Pichel, E. Maturi, P. Clemente-Colón, and J. Sapper, 2001a. Deriving the operational nonlinear multi-channel sea surface temperature algorithm coefficients for NOAA-15 AVHRR/3, Int. J. Remote Sens., Volume 22, No. 4, 699 - 704.

Li, X, W. Pichel, P. Clemente-Colón, V. Krasnopolsky, and J. Sapper, 2001b. Validation of coastal sea and lake surface temperature measurements derived from NOAA/AVHRR Data, Int. J. Remote Sens., Vol. 22, No. 7, 1285-1303.

Stowe, L. L., P. A. Davis, and E. P. McClain, 1999. Scientific basis and initial evaluation of the CLAVR-1 global clear/cloud classification algorithm for the advanced very high resolution radiometer. J. Atmos. Oceanic Technol., 16, 656-681.

Walton C. C., W. G. Pichel, J. F. Sapper, D. A. May, 1998. The development and operational application of nonlinear algorithms for the measurement of sea surface temperatures with the NOAA polar-orbiting environmental satellites, J. Geophys. Res., 103: (C12) 27999-28012.

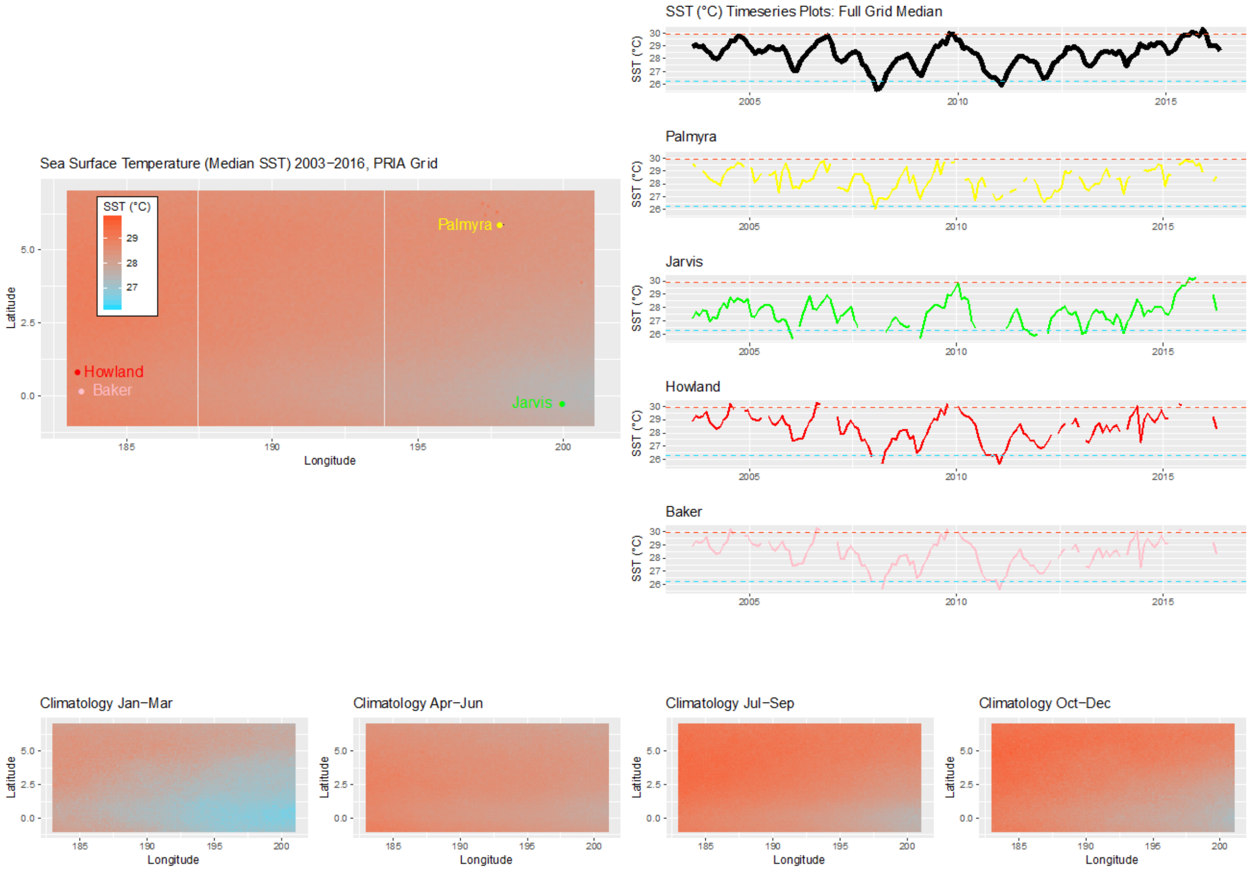


Figure 15. Sea surface temperature for Pacific Remote Island regional grid.

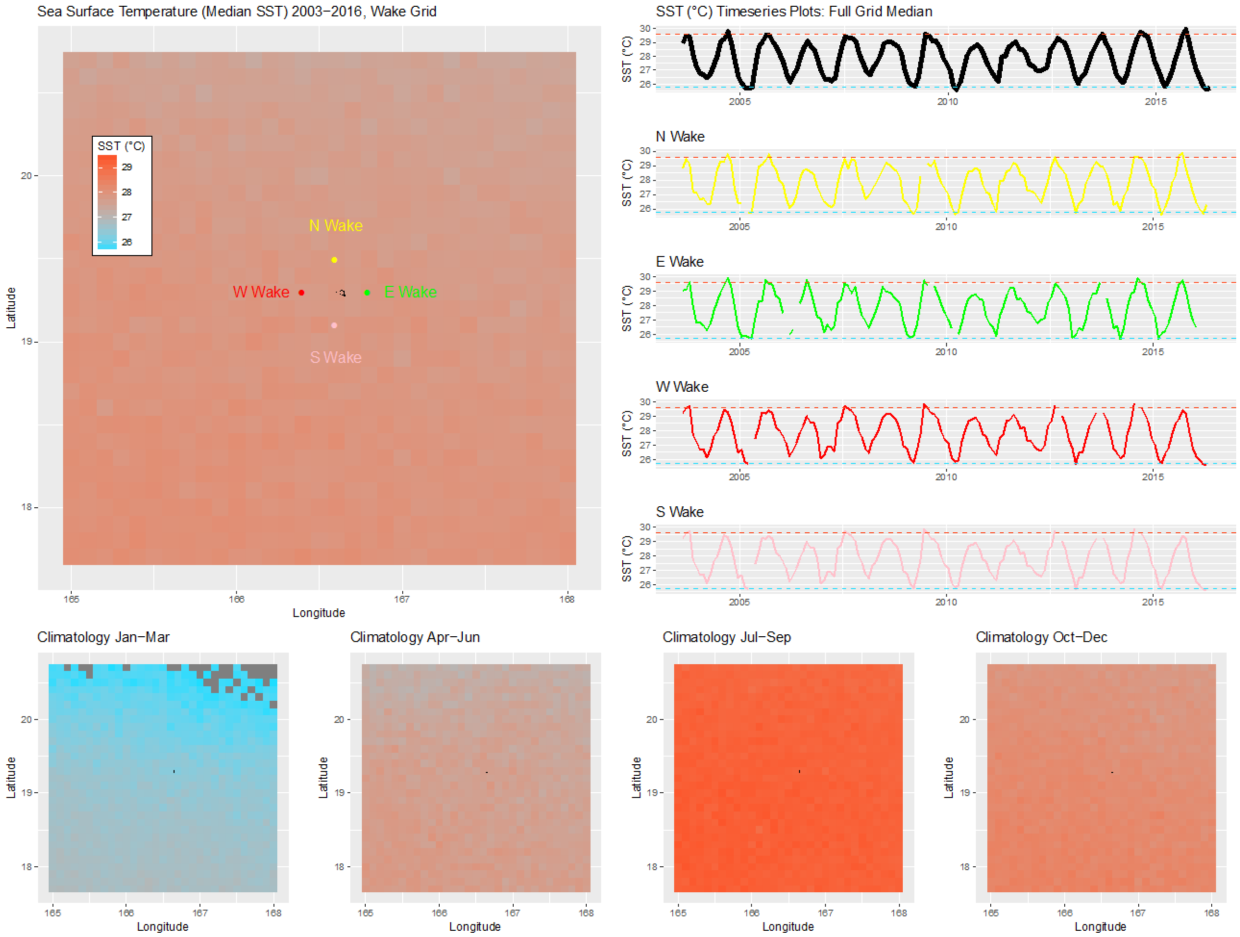

Sea Surface Temperature (Median SST) 2003−2016, Wake Grid
SST (°C)
29
28
27
26
N Wake
W Wake
E Wake
S Wake
Latitude
Longitude
20
19
18
165
166
167
168
SST (°C) Timeseries Plots: Full Grid Median
N Wake
E Wake
W Wake
S Wake
SST (°C)
30
29
28
27
26
2005
2010
2015
Climatology Jan−Mar
Climatology Apr−Jun
Climatology Jul−Sep
Climatology Oct−Dec

Figure 16. Sea surface temperature for Wake Island regional grid.

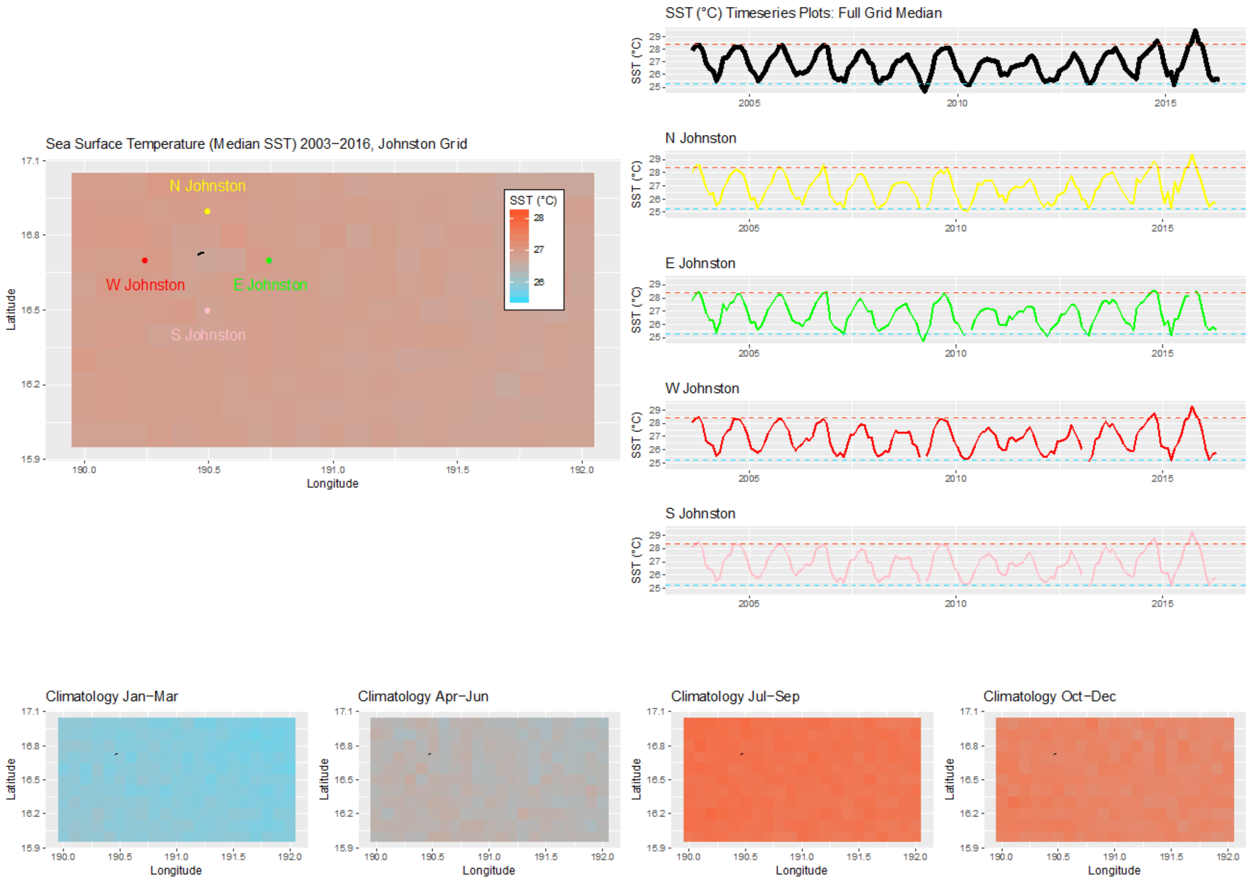


Figure 17. Sea Surface Temperature for Johnston Atoll regional grid.

2.4.3.6 Sea Surface Temperature Anomaly

Description: Monthly sea surface temperature anomaly from 2003-2016 from the AVHRR instrument aboard the NOAA Polar Operational Environmental Satellite (POES), compared against the Casey and Cornillon Climatology (Casey and Cornillion 1999). These data take us back to 2003. If we were to blend this record with Pathfinder, we could reach back to 1981.

Background Below Inserted From *Coastwatch West Coast Node****:*** [http://coastwatch.pfeg.noaa.gov/infog/AG_tanm_las.html]. We would like to acknowledge the NOAA CoastWatch Program and the NOAA NESDIS Office of Satellite Data Processing and Distribution.

Short Description:
The SST anomaly product is used to show the difference between the surface temperature at a given time and the temperature that is normal for that time of year. This effectively filters out seasonal cycles and allows one to view intra-seasonal and inter-annual signals in the data. The global SST anomaly product is produced by comparing the AVHRR GAC SST with a climatology by *Casey and Cornillon, 1999*, for the region and time period specified. The AVHRR GAC SST is a high quality data set provided by NOAA | NESDIS | OSDPD.

Technical Summary:
SST anomaly data are distributed at 11km resolution. AVHRR GAC SST values are accurate to within plus or minus 0.5 degrees Celsius. The time-averaged SST from AVHRR GAC is compared to the climatological SST from *Casey and Cornillon, 1999*, for the specific time period and region. The data are mapped to an equal angle grid of 0.1 degrees latitude by 0.1 degrees longitude using a simple arithmetic mean to produce composite images of various duration (e.g., 1, 3, 8, 14-day).

Reference: Casey, K.S. and P. Cornillon. 1999. A comparison of satellite and in situ based sea surface temperature climatologies. J. Climate. Vol. 12, no. 6, 1848-1863.

Timeframe: 2003-2015. Daily data available. Monthly means shown.

Region/Location: Global.

Data Source: "SST Anomaly, POES AVHRR, Casey and Cornillon Climatology, Global (Monthly Composite)"
http://coastwatch.pfeg.noaa.gov/erddap/griddap/erdAGtanmmday_LonPM180.html

Measurement Platform: *POES, AVHRR Satellite*

Rationale: Sea surface temperature anomaly highlights long-term trends. Filtering out seasonal cycle is one of the most directly observable measures we have for tracking increasing ocean temperature.

References: Casey, K.S. and P. Cornillon. 1999. A comparison of satellite and in situ based sea surface temperature climatologies. J. Climate. Vol. 12, no. 6, 1848-1863.

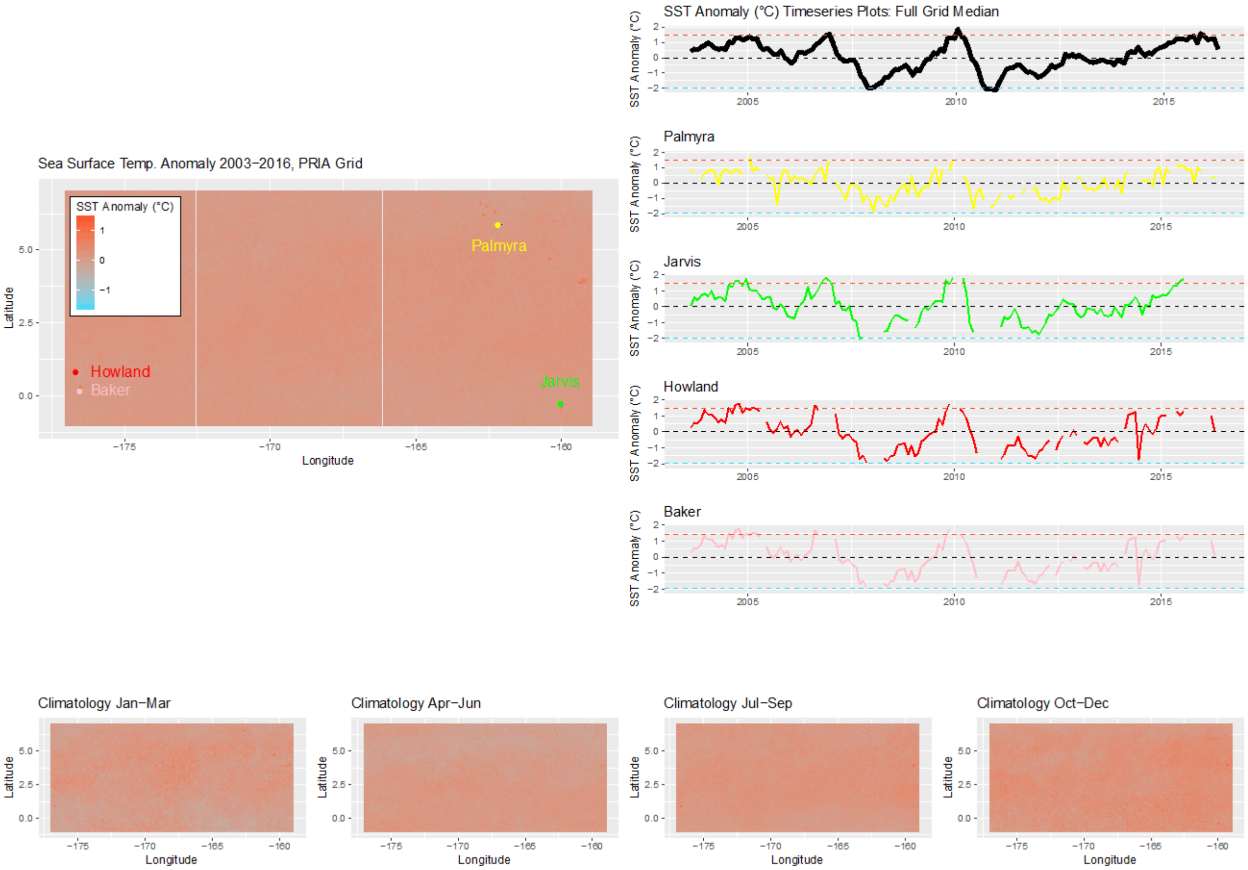


Figure 18. Sea surface temperature anomaly for Pacific remote island regional grid.

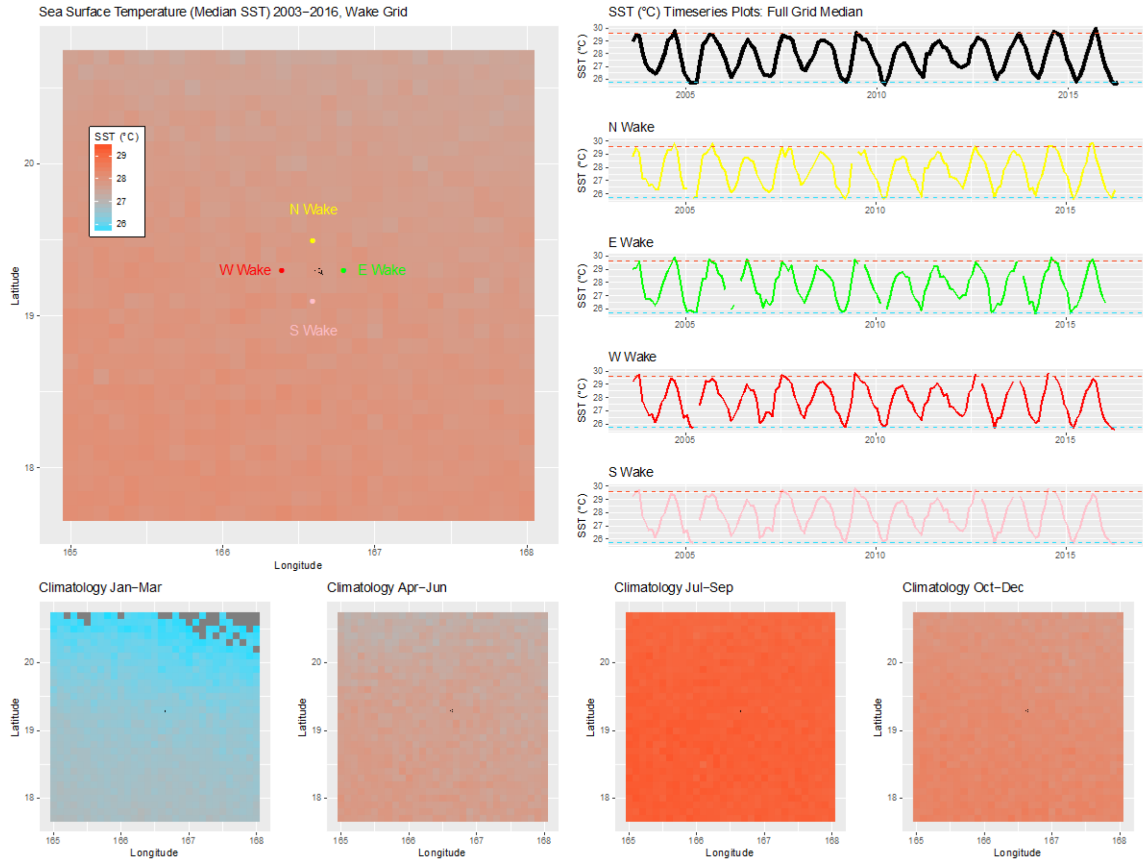


Figure 19. Sea surface temperature anomaly for Pacific remote island (Johnston Atoll) regional grid.

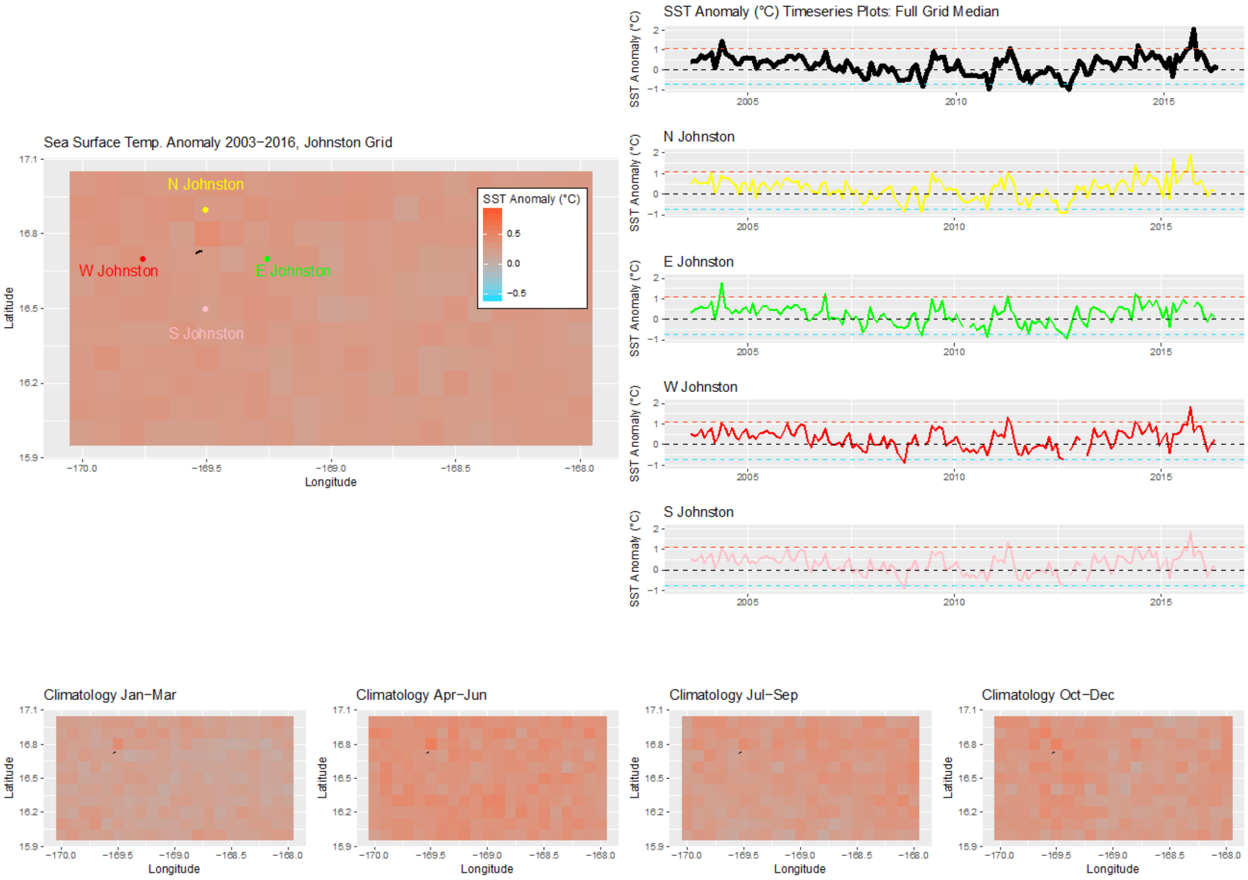


Figure 20. Sea surface temperature anomaly for Wake Island regional grid.

2.4.3.7 Degree Heating Weeks (Coral Bleaching)

Description: The NOAA Coral Reef Watch program's satellite data provide current reef environmental conditions to quickly identify areas at risk for coral bleaching, where corals lose the symbiotic algae that give them their distinctive colors. If a coral is severely bleached, disease and partial mortality become likely, and the entire colony may die.

Continuous monitoring of sea surface temperature at global scales provides researchers and stakeholders with tools to understand and better manage the complex interactions leading to coral bleaching. When bleaching conditions occur, these tools can be used to trigger bleaching response plans and support appropriate management decisions.

[Descriptions from: https://coralreefwatch.noaa.gov/satellite/index.php]

Technical Summary: The NOAA Coral Reef Watch (CRW) daily 5-km satellite coral bleaching Degree Heating Week (DHW) product presented here shows accumulated heat stress, which can lead to coral bleaching and death. The scale goes from 0 to 20 °C-weeks. The DHW product accumulates the instantaneous bleaching heat stress (measured by Coral Bleaching HotSpots) during the most-recent 12-week period. It is directly related to the timing and intensity of coral bleaching. Significant coral bleaching usually occurs when DHW values reach 4 °C-weeks. By the time DHW values reach 8 °C-weeks, widespread bleaching is likely and significant mortality can be expected.

Timeframe: 2013-2016. Weekly 5 km data.

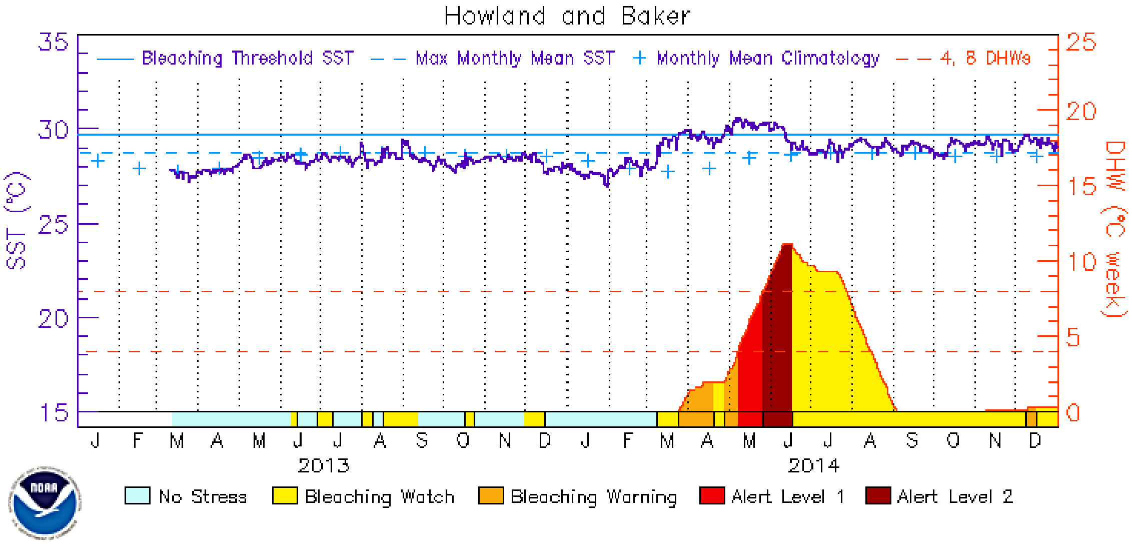
Region/Location: Global.

Data Source: NOAA Coral Reef Watch. 2013, updated daily. *NOAA Coral Reef Watch Daily Global 5-km Satellite Virtual Station Time Series Data for Southeast Florida*, Mar. 12, 2013-Mar. 11, 2014. College Park, Maryland, USA: NOAA Coral Reef Watch. Data set accessed 2017-03-21 at http://coralreefwatch.noaa.gov/vs/index.php

Measurement Platform: CRW operational near-real-time nighttime SST product: AVHRR,

Rationale: Degree Heating Weeks are the best available metric to track coral bleaching relevant high temperature exposure.

References: Liu, G., A.E. Strong, W.J. Skirving and L.F. Arzayus (2006). Overview of NOAA Coral Reef Watch Program's Near-Real-Time Satellite Global Coral Bleaching Monitoring Activities. *Proceedings of the 10th International Coral Reef Symposium, Okinawa*: 1783-1793.


Howland and Baker
Bleaching Threshold SST
Max Monthly Mean SST
Monthly Mean Climatology
4, 8 DHWs
SST (°C)
35
30
25
20
15
DHW (°C week)
25
20
15
10
5
0
J F M A M J J A S O N D J F M A M J J A S O N D
2013
2014
No Stress
Bleaching Watch
Bleaching Warning
Alert Level 1
Alert Level 2

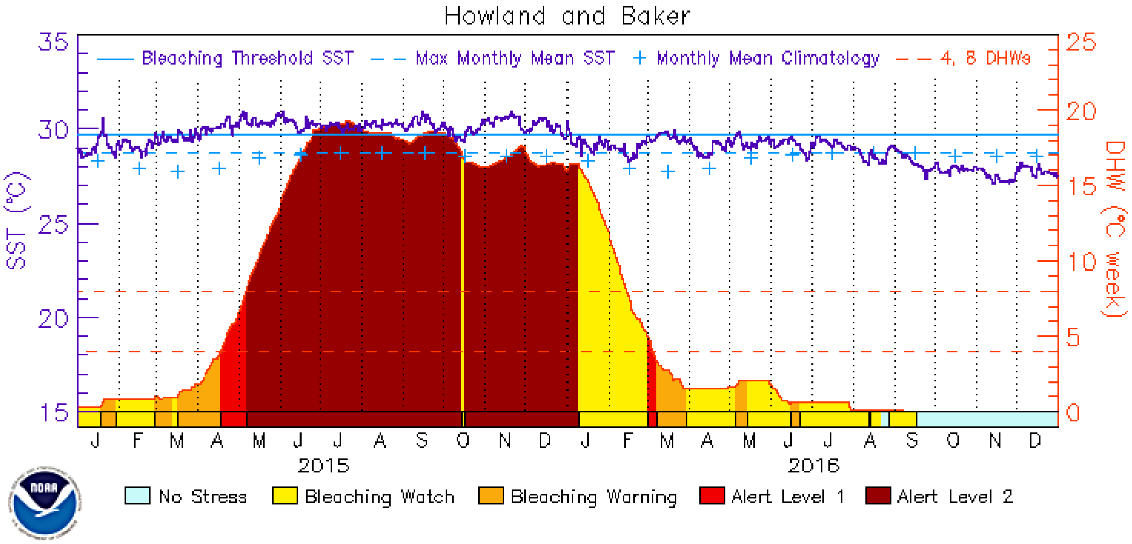

Howland and Baker
Bleaching Threshold SST
Max Monthly Mean SST
Monthly Mean Climatology
4, 8 DHWs
SST (°C)
35
30
25
20
15
DHW (°C week)
25
20
15
10
5
0
J F M A M J J A S O N D J F M A M J J A S O N D
2015
2016
No Stress
Bleaching Watch
Bleaching Warning
Alert Level 1
Alert Level 2

Figure 21. Degree Heating Weeks Time-series in the Howland/Baker 2013-2016.

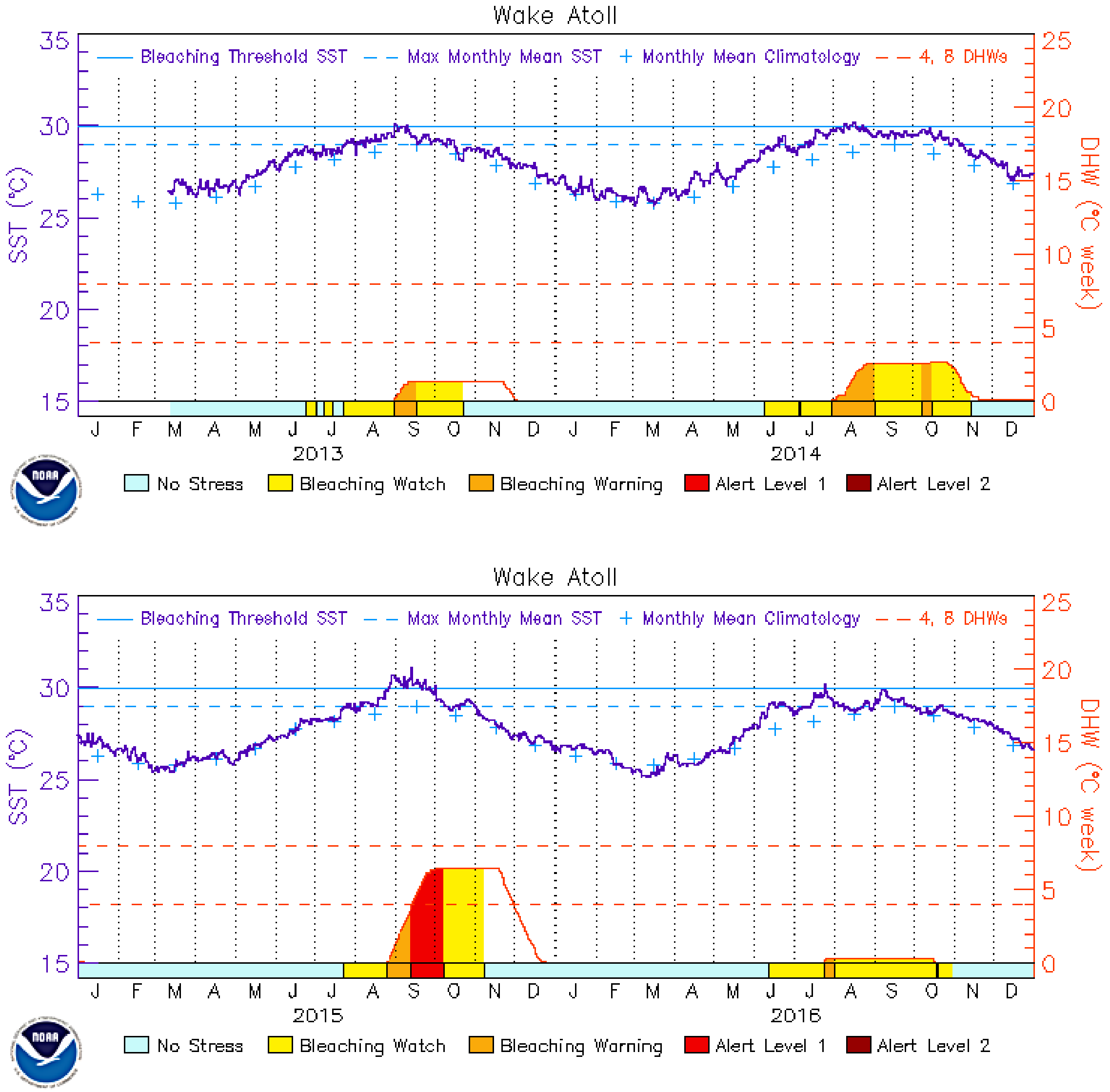
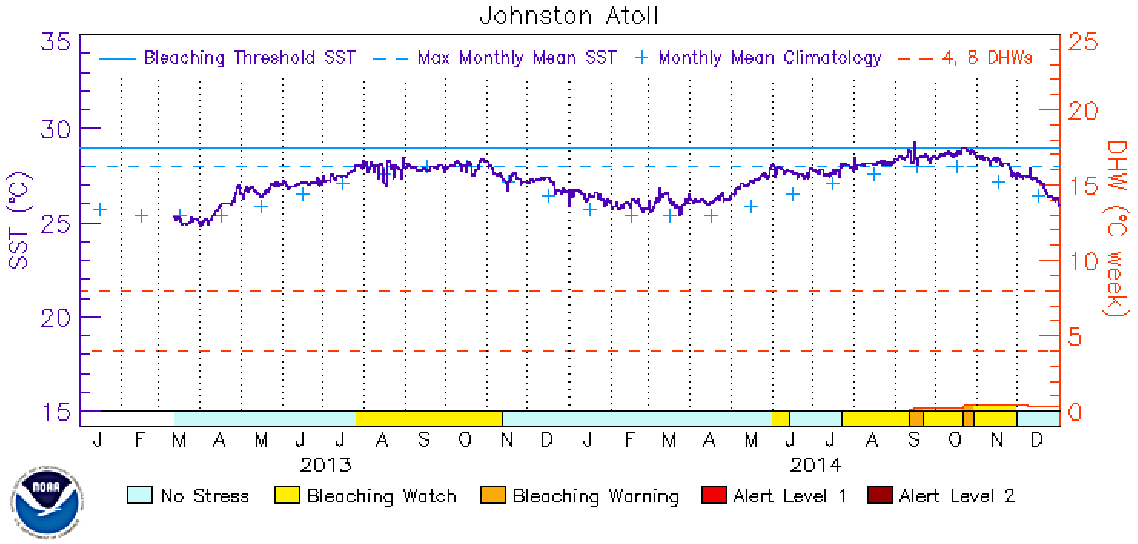


Figure 22. Degree Heating Weeks Time-series at Wake Atoll 2013-2016.



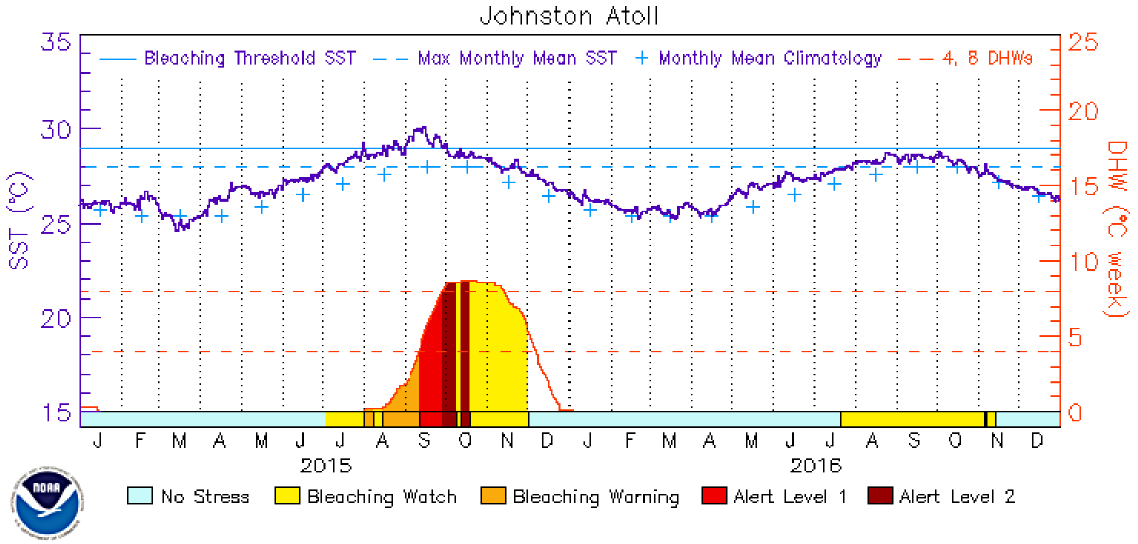
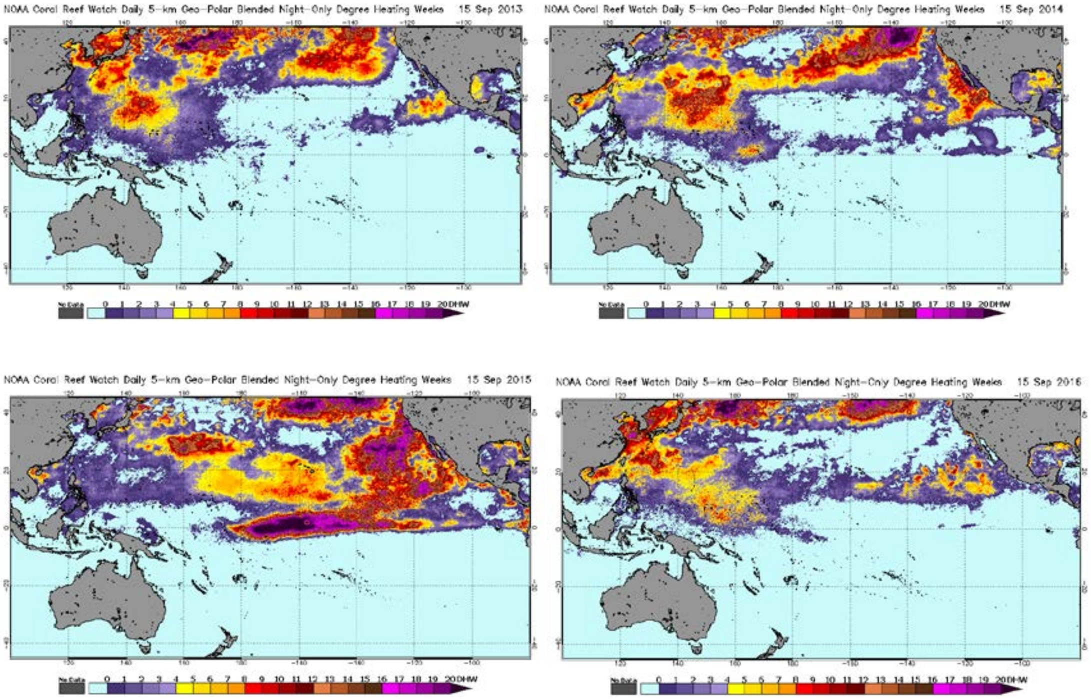


Figure 23. Degree Heating Weeks Time-series at Johnston Atoll 2013-2016.



NOAA Coral Reef Watch Daily 5-km Geo-Polar Blended Night-Only Degree Heating Weeks 15 Sep 2013

NOAA Coral Reef Watch Daily 5-km Geo-Polar Blended Night-Only Degree Heating Weeks 15 Sep 2014

NOAA Coral Reef Watch Daily 5-km Geo-Polar Blended Night-Only Degree Heating Weeks 15 Sep 2015

NOAA Coral Reef Watch Daily 5-km Geo-Polar Blended Night-Only Degree Heating Weeks 15 Sep 2016

Figure 24. Degree Heating Weeks Maps, showing Annual DHW Maximum (Sep 15, 2013-2016) across the Pacific Ocean.

2.4.3.8 Heavy Weather (Tropical Cyclones)

Description: This indicator uses historical data from the National Oceanic and Atmospheric Administration’s (NOAA) National Climatic Data Center (NCDC) International Best Track Archive for Climate Stewardship (IBTrACS) to track the number of tropical cyclones in the western, central, and south Pacific basins. This indicator also monitors the Accumulated Cyclone Energy (ACE) Index and the Power Dissipation Index (PDI) which are two ways of monitoring the frequency, strength, and duration of tropical cyclones based on wind speed measurements.

The annual frequency of storms passing through the western North Pacific basin is tracked and a stacked time series plot will show the representative breakdown of the Saffir-Simpson hurricane categories. Three solid lines across the graph will also be plotted representing a) the annual long-term average number of named storms, b) the annual average number of typhoons, and c) the annual average number of major typhoons (Cat 3 and above). Three more lines will also be shown (in light gray) representing the annual average number of named-storms for ENSO a) neutral, b) warm, and c) cool.

Every cyclone has an ACE Index value, which is a number based on the maximum wind speed measured at six-hourly intervals over the entire time that the cyclone is classified as at least a tropical storm (wind speed of at least 34 knot; 39 mph). Therefore, a storm’s ACE Index value accounts for both strength and duration. This plot will show the historical ACE values for each typhoon season and will have a solid line representing the annual average ACE value. Three more lines will also be shown (in light gray) representing the annual average ACE values for ENSO a) neutral, b) warm, and c) cool.

Timeframe: Yearly

Region/Location: Hawaii and U.S. Affiliated Pacific Islands

Data Source/Responsible Party: NCDC’s International Best Track Archive for Climate Stewardship (IBTrACS).

Measurement Platform: Satellite

Rationale: The effects of tropical cyclones are numerous and well-known. At sea, storms disrupt and endanger shipping traffic as well as fishing effort and safety. The Hawaii longline fishery, for example, had serious problems between August and November 2015 with vessels dodging storms at sea, delayed departures and inability to make it safely back to Honolulu because of bad weather. When cyclones encounter land, their intense rains and high winds can cause severe property damage, loss of life, soil erosion, and flooding. The associated storm surge, the large volume of ocean water pushed toward shore by the cyclone’s strong winds, can cause severe flooding and destruction.

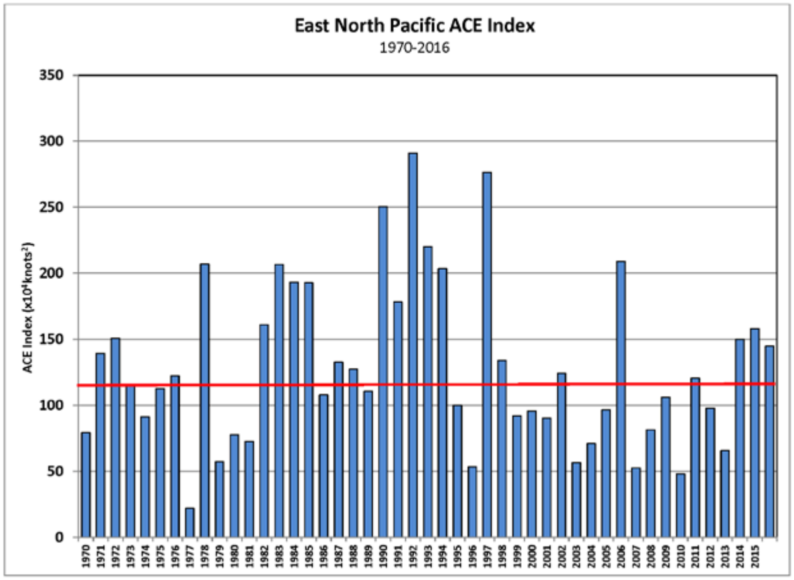


Figure 25. 2016 East Pacific Tropical Cyclone ACE 1970-2016. Source: NOAA's National Hurricane Center

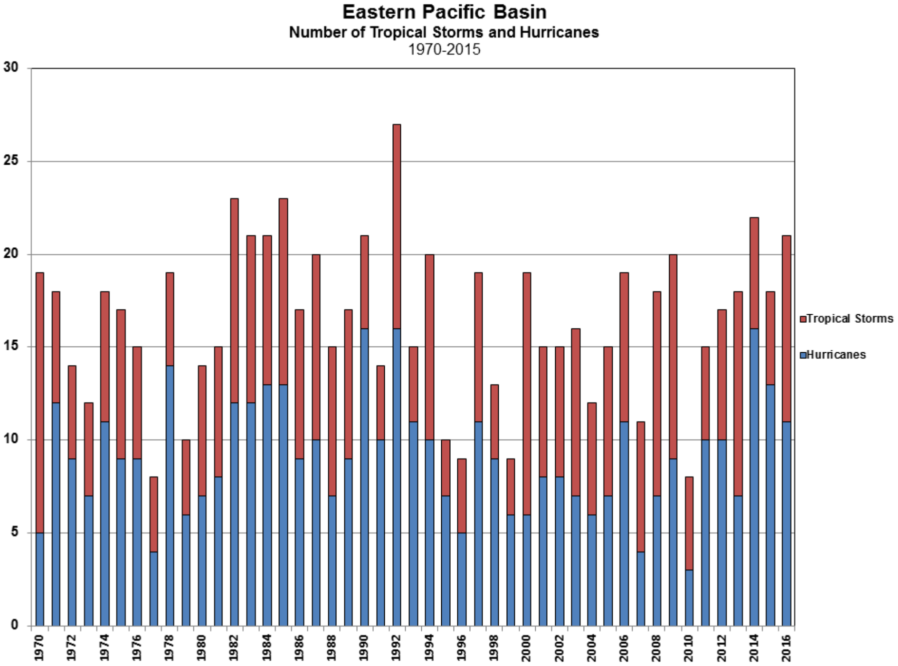


Figure 26. East Pacific tropical cyclone count 1970-2016. Source: NOAA's National Hurricane Center

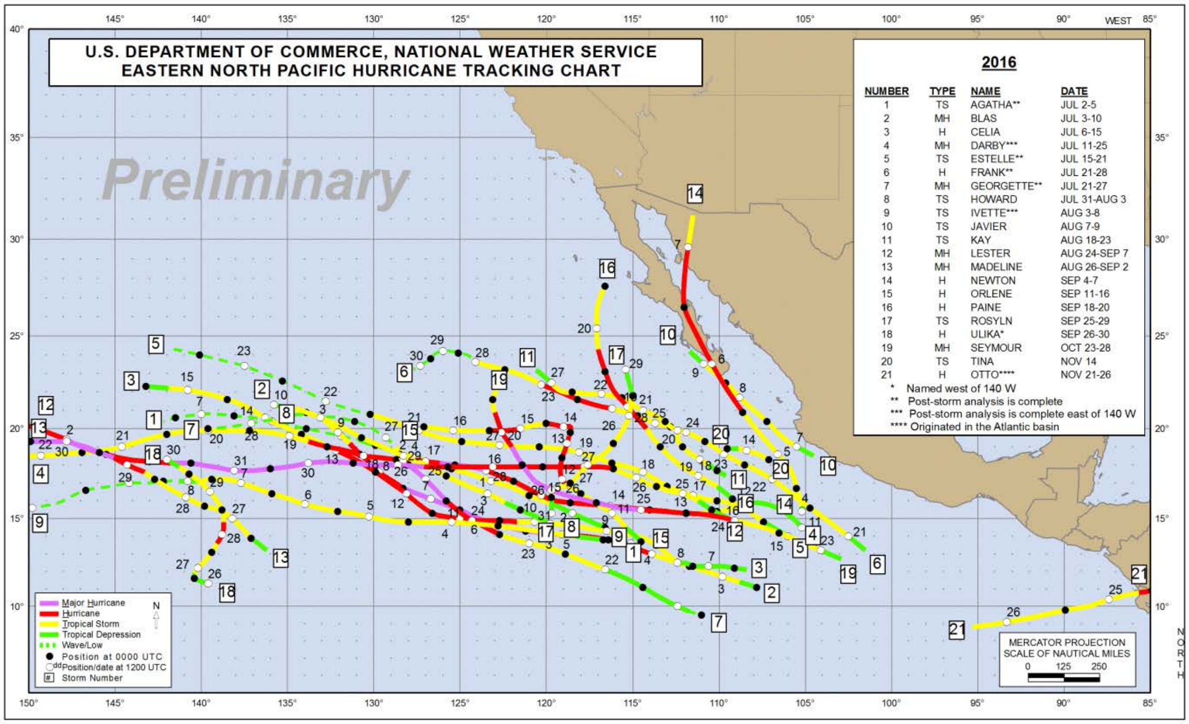


Figure 27. 2016 Eastern Pacific Tropical Cyclone Tracks. Source: NOAA's National Hurricane Center

The NOAA National Centers for Environmental Information, State of the Climate: Hurricanes and Tropical Storms for Annual 2015, published online January 2016, notes that "the 2015 East Pacific hurricane season had 18 named storms, including 13 hurricanes, nine of which became major. The 1981-2010 average number of named storms in the East Pacific is 16.5, with 8.9 hurricanes, and 4.3 major hurricanes. This is the first year since reliable record keeping began in 1971 that the eastern Pacific saw nine major hurricanes. The Central Pacific also saw an above-average tropical cyclone season, with 14 named storms, eight hurricanes, and five major hurricanes, the most active season since reliable record-keeping began in 1971. Three major hurricanes (Ignacio, Kilo and Jimena) were active across the two adjacent basins at the same time, the first time this occurrence has been observed. The ACE index for the East Pacific basin during 2015 was 158 (x10^4 knots^2), which is above the 1981-2010 average of 132 (x10^4 knots^2) and the highest since 2006. The Central Pacific basin ACE during 2015 was 124 (x10^4 knots^2)."

Inserted from: http://www.ncdc.noaa.gov/sotc/tropical-cyclones/201513

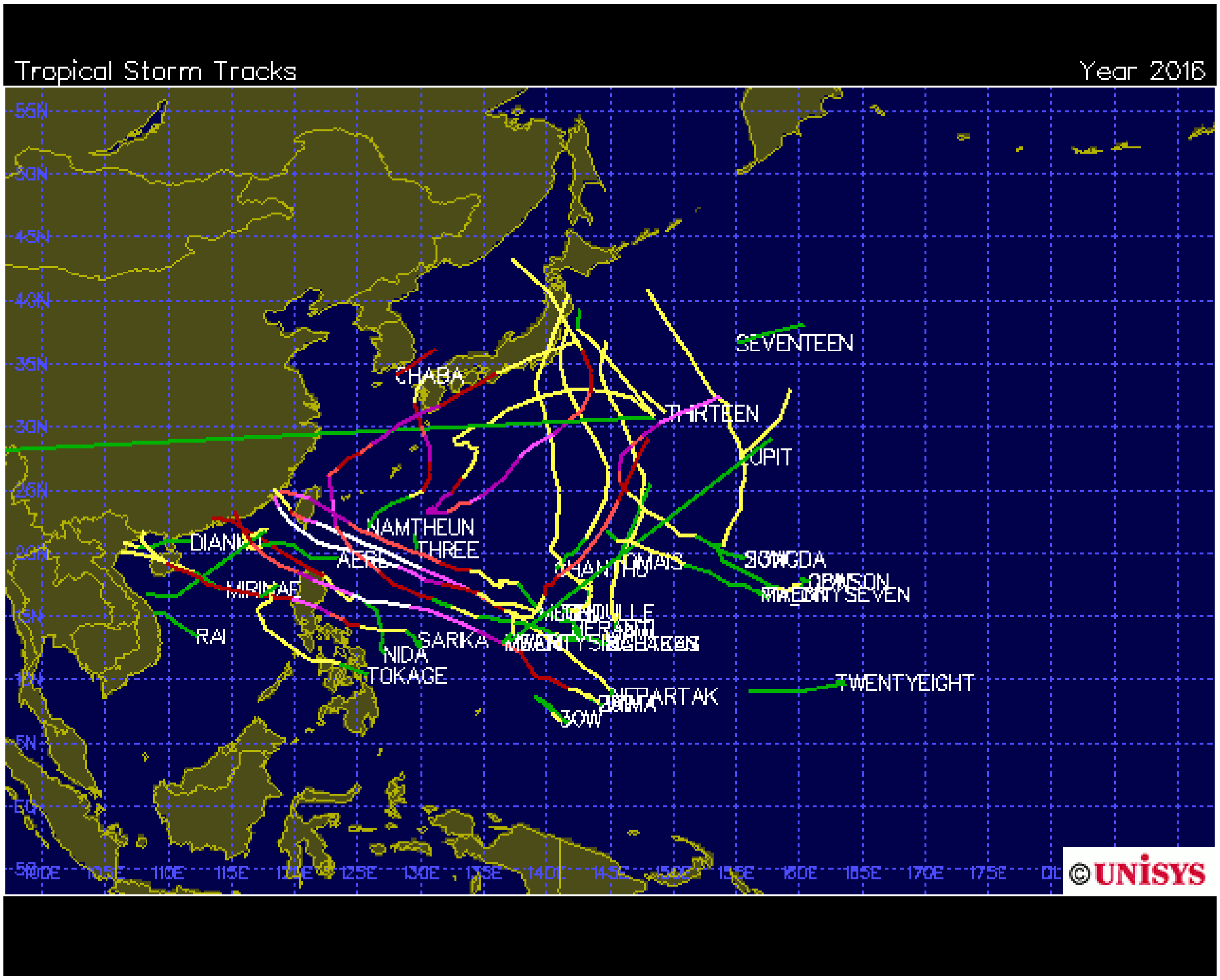


Figure 28. Western Pacific Cyclone Tracks 2016. Source: http://weather.unisys.com/hurricane/w_pacific/2016

References: NOAA National Centers for Environmental Information, State of the Climate: Hurricanes and Tropical Storms for Annual 2016, published online January 2016, retrieved on August 5, 2016 from http://www.ncdc.noaa.gov/sotc/tropical-cyclones/201513.

2.4.3.9 Sea Level (Sea Surface Height and Anomaly)

Description: Monthly mean sea level time series, including extremes

Timeframe: Monthly

Region/Location: Observations from selected sites within the Hawaiian Archipelago

Data Source/Responsible Party: Basin-wide context from satellite altimetry: http://www.aviso.altimetry.fr/en/data/products/ocean-indicators-products/el-nino-bulletin.html

Quarterly time series of mean sea level anomalies from satellite altimetry: http://sealevel.jpl.nasa.gov/science/elninopdo/latestdata/archive/index.cfm?y=2015

Sea Surface Height and Anomaly from NOAA Ocean Service, Tides and Currents, Sea Level Trends https://tidesandcurrents.noaa.gov/sltrends/sltrends_station.shtml?stnid=1612340

Measurement Platform: Satellite and *in situ* tide gauges

Rationale: Rising sea levels can result in a number of coastal impacts, including inundation of infrastructure, increased damage resulting from storm-driven waves and flooding, and saltwater intrusion into freshwater supplies.

Basin-Wide Perspective

This image of the mean sea level anomaly for February 2016 compared to 1993-2013 climatology from satellite altimetry provides a glimpse into how the 2015-2016 El Niño continues to affect sea level across the Pacific Basin. The image captures the fact that sea level continues to be lower in the Western Pacific and higher in the Central and Eastern Pacific (a standard pattern during El Niño events.) This basin-wide perspective provides a context for the location-specific sea level/sea surface height images that follow.

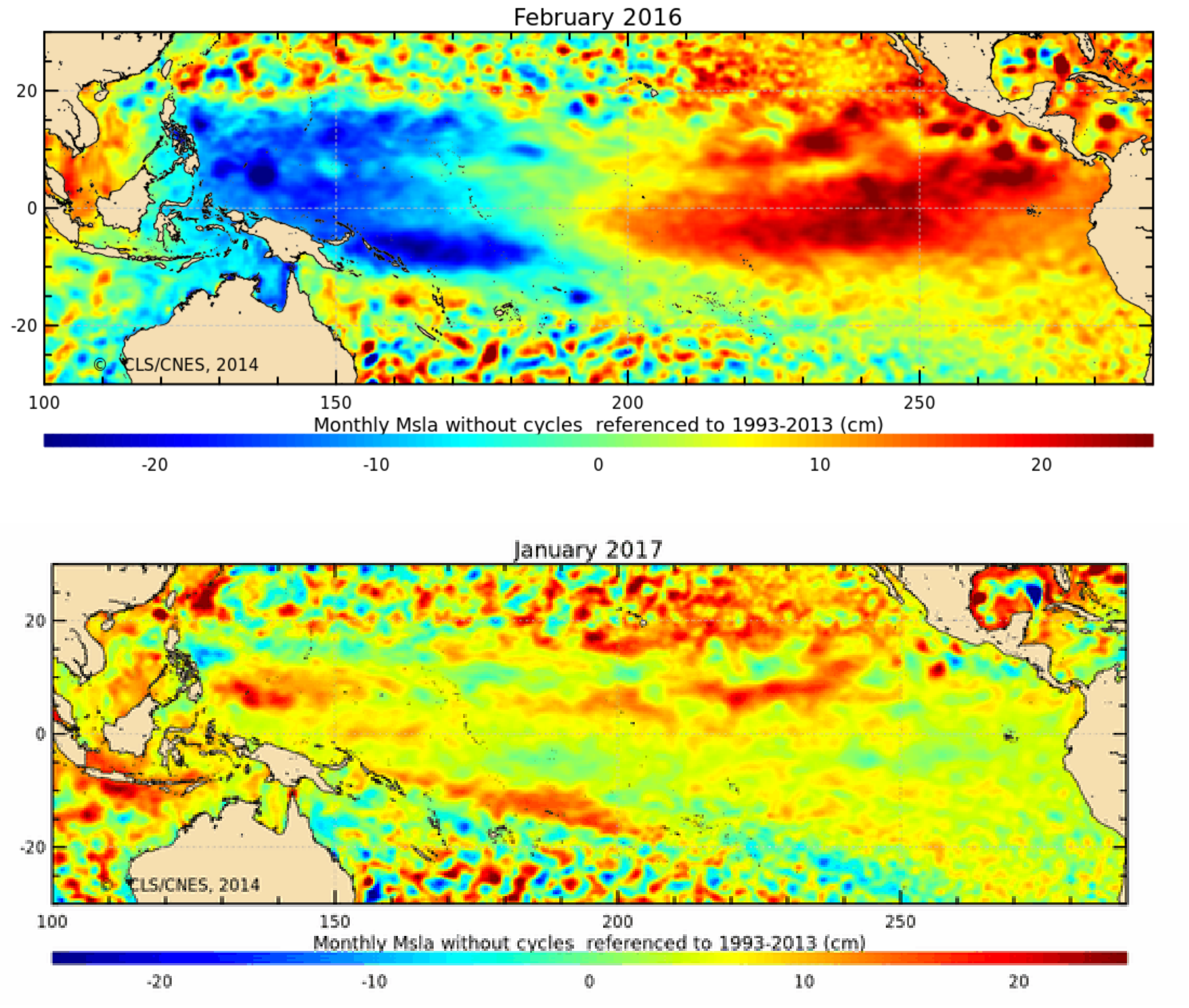
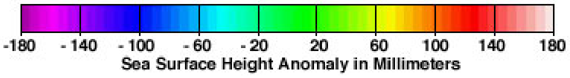
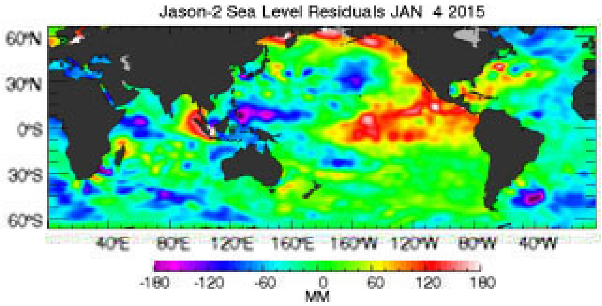
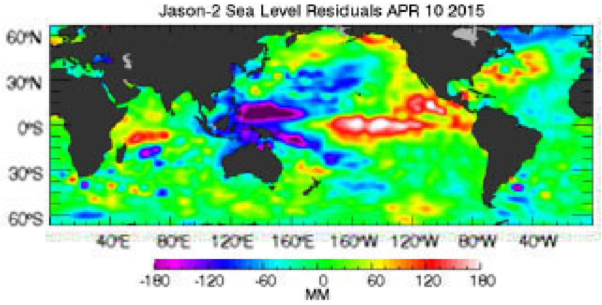


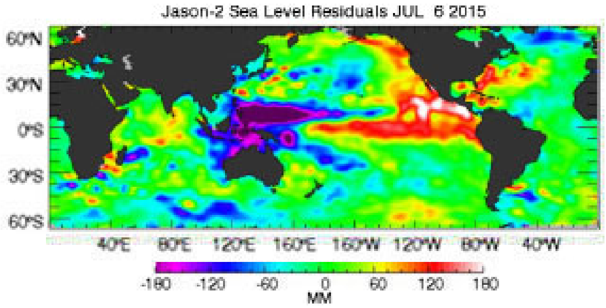
Figure 29. Comparing mean sea level anomaly for February 2016 (El Niño), and January 2017 (Neutral) .

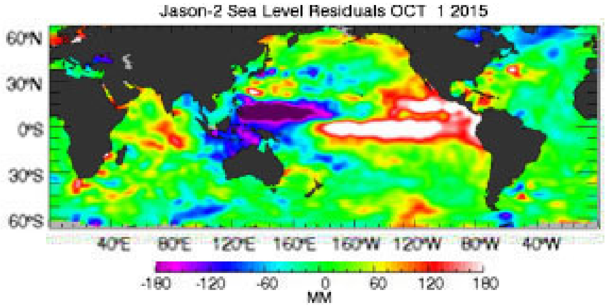
Quarterly time series of mean sea level anomalies during 2015 provide a glimpse into the evolution of the 2015-2016 El Niño throughout the year using satellite altimetry measurements of sea level height (http://sealevel.jpl.nasa.gov/science/elninopdo/latestdata/archive/index.cfm?y=2015**)**

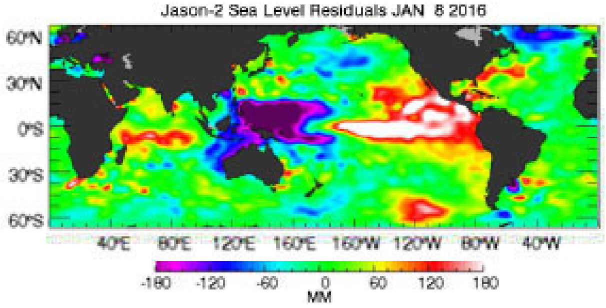


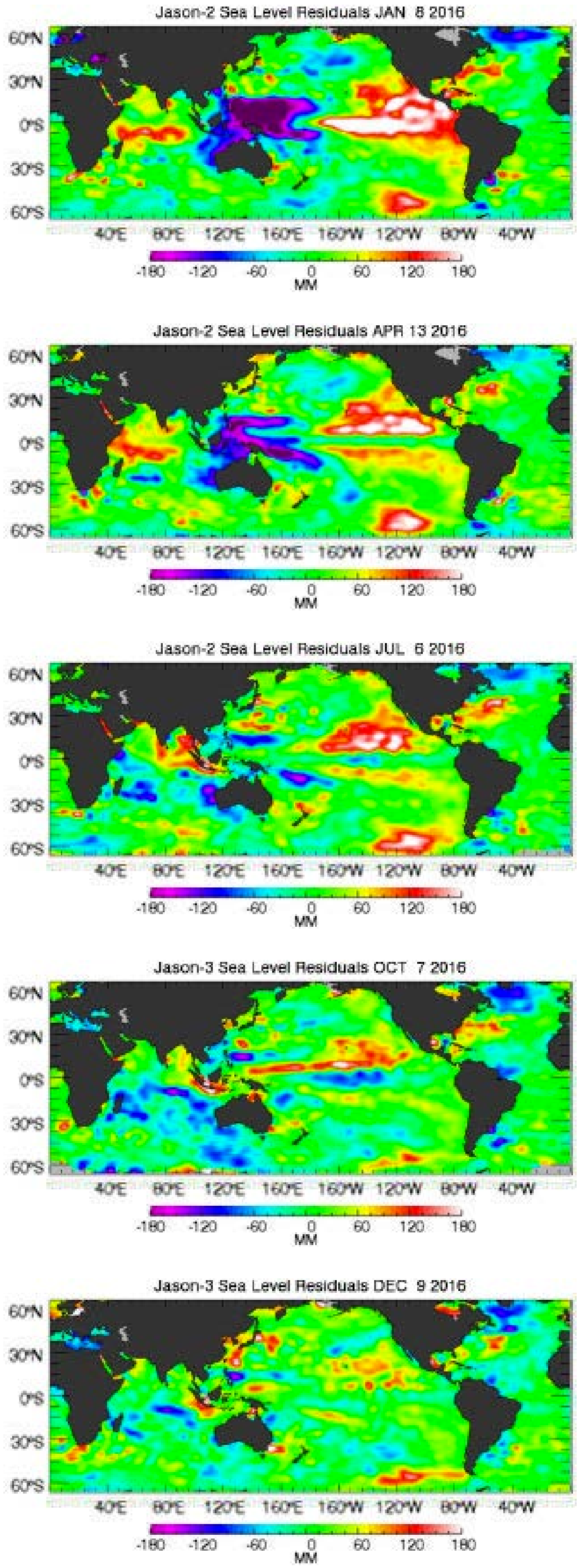




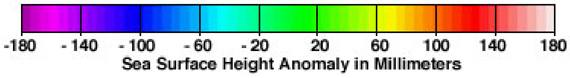








Quarterly time series of mean sea level anomalies during 2016 provide a glimpse into the dissipation of the 2015-2016 El Niño throughout the year using satellite altimetry measurements of sea level height **(**http://sealevel.jpl.nasa.gov/science/elninopdo/lateststdata/archive/index.cfm?y=2016**)**



2.4.3.10 Local Sea Level

These time-series from *in situ* tide gauges provide a perspective on sea level trends within each Archipelago (Tide Station Time Series from NOAA/COOPS). However, the PRIA have no local tide gauges.

2.4.3.11 Wave Watch 3 Global Wave Model

Description: To describe patterns in wave forcing, we present data from the Wave Watch 3 global wave model run by the Department of Ocean and Resources Engineering at the University of Hawaiʻi in collaboration with NOAA/NCEP and NWS Honolulu. PacIOOS describes the model at http://oos.soest.hawaii.edu/pacioos/focus/modeling/wave_models.php: “The global model is initialized daily and is forced with NOAA/NCEP's global forecast system (GFS) winds. This model is designed to capture the large-scale ocean waves, provide spectral boundary conditions for the Hawaiʻi and Mariana Islands regional WW3 model, and most importantly, the 7 day model outputs a 5 day forecast.”

Data presented here come from the global model, but regional WW3 models with higher resolution exist for Hawaii, Marianas and Samoa, and in some cases, very high resolution SWAN models exist for islands within those groups.

Timeframe: 2010-2016, Daily data.

Region/Location: Global.

Data Source: “WaveWatch III (WW3) Global Wave Model”: http://oos.soest.hawaii.edu/erddap/griddap/NWW3_Global_Best.html

Measurement Platform: *Global Forecast System Winds, WW3 model*

Rationale: Wave forcing can have major implications for both coastal ecosystems and pelagic fishing operations.

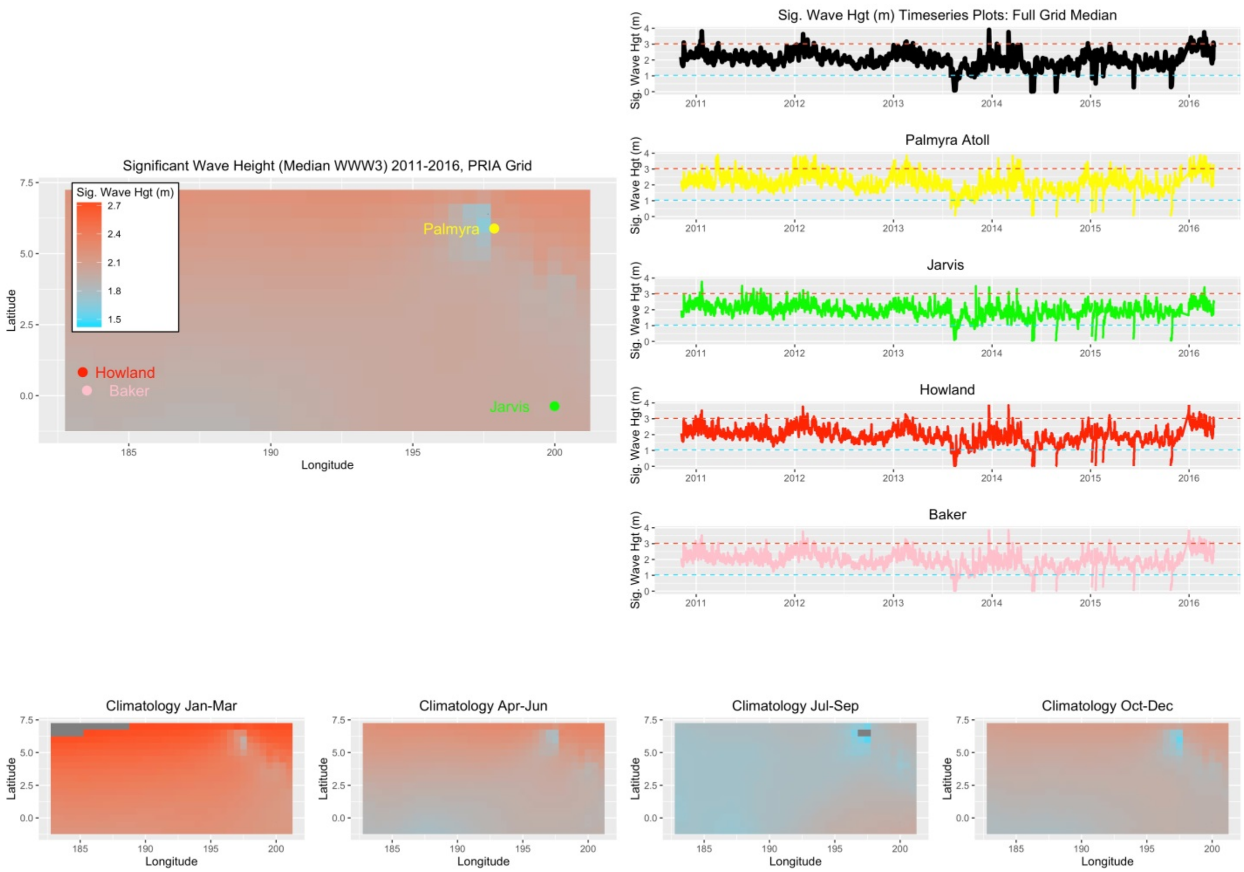


Figure 30. Wave watch summary for Pacific remote island grid.

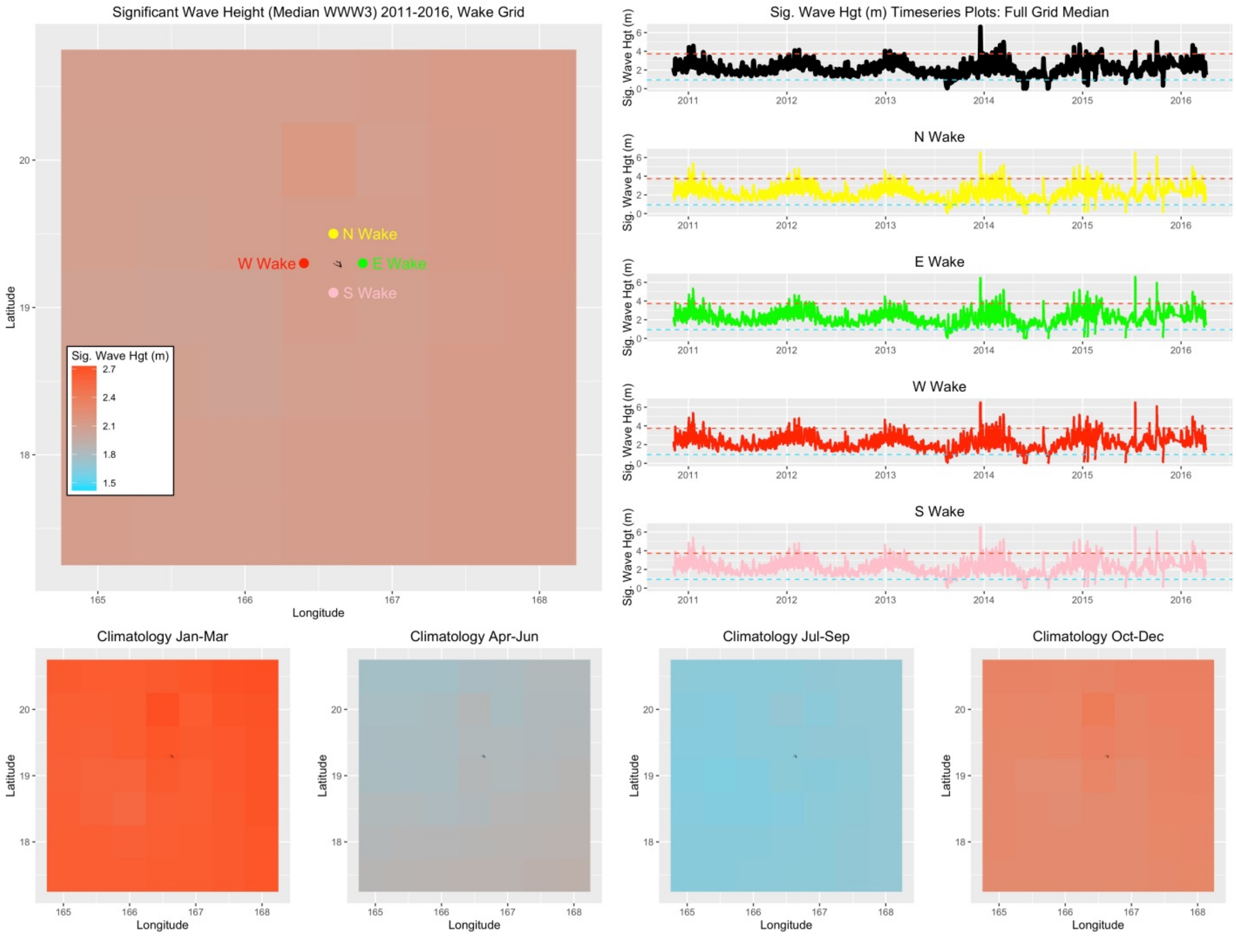


Figure 31. wave watch summary for Wake Island grid.

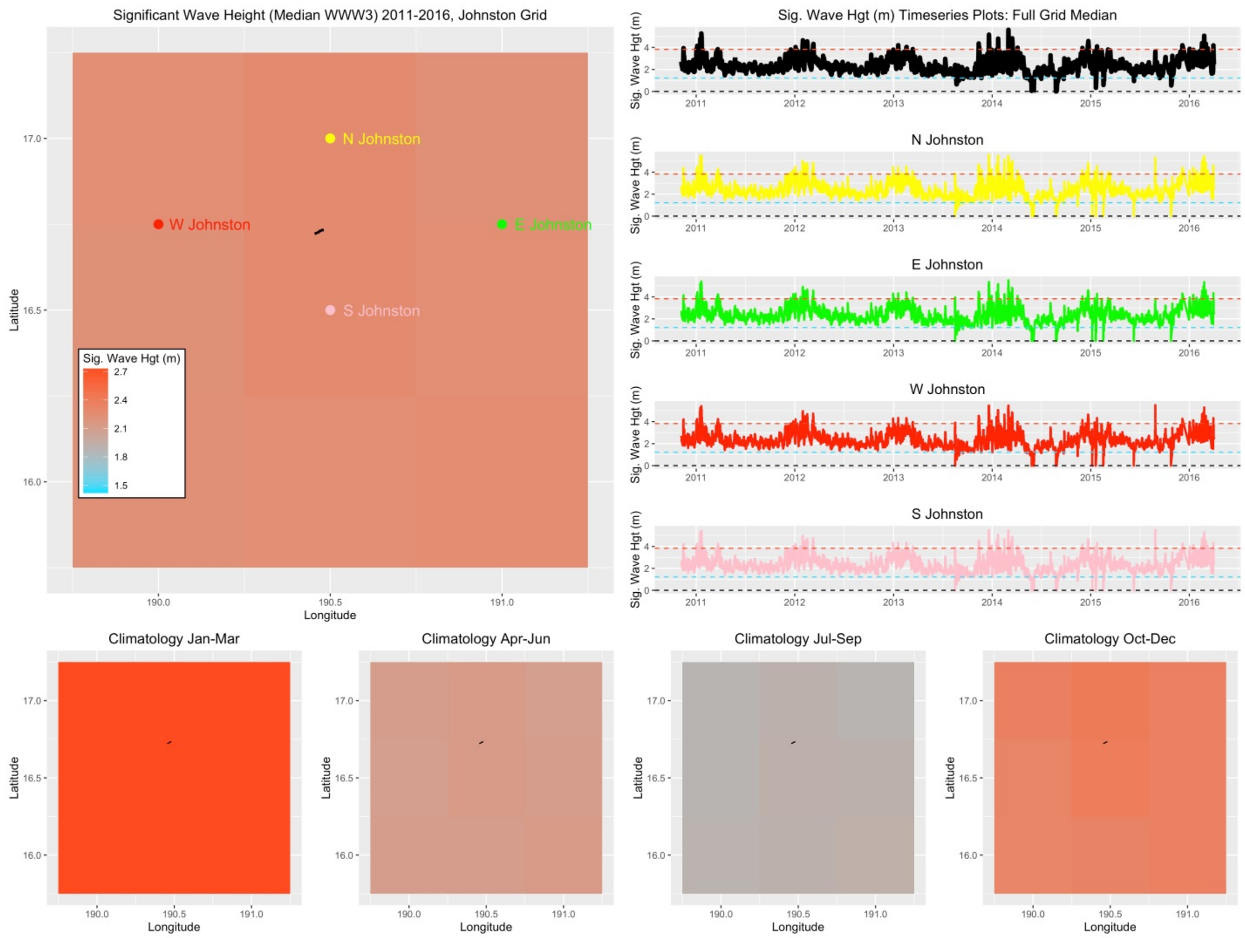


Figure 32. Wave watch summary for Johnston Atoll grid.

2.4.4 Observational and Research Needs

Through preparation of the 2016 Archipelagic Annual Reports, the Council has identified a number of observational and research needs that, if addressed, would improve the information content of future Climate and Ocean Indicators chapters. This information would provide fishery managers, fishing industry and community stakeholders with better understanding and predictive capacity vital to sustaining resilient and vibrant fishery systems in the Western Pacific.

- Emphasize the importance of continuing the climate and ocean indicators used in this report so that a consistent, long-term record can be maintained;
- Develop agreements among stakeholders and research partners to ensure the sustainability, availability and accessibility of climate and ocean indicators, their associated datasets and analytical methods used in this and future reports;
- Improve monitoring and understanding of the impacts of changes in ocean temperature, pH and ocean acidity, ocean oxygen content and hypoxia, and sea level rise through active collaboration by all fishery stakeholders and research partners;
- Develop, test and provide access to additional climate and ocean indicators that can improve the Archipelagic Conceptual Model;
- Explore the connections among sea surface conditions, stratification and mixing;
- Investigate the connections between climate variables and other indicators in the Archipelagic Conceptual Model to improve understanding of changes in physical, biochemical, biologic and socio-economic processes and their interactions in the regional ecosystem;
- Develop predictive models that can be used for scenario planning to account for unexpected changes and uncertainties in the regional ecosystem and fisheries;
- Foster applied research in ecosystem modeling to better describe current conditions and to better anticipate the future under alternative models of climate and ocean change including changes in expected human benefits and their variability;
- Improve understanding of the connections between PDO and fisheries ecosystems beyond the North Pacific;
- Improve understanding of mahi and swordfish size in relation to the orientation of the Transition Zone Chlorophyll Front (TZCF);
- Explore the biological implications of tropical cyclones;
- Standardize fish community size structure data for gear type;
- Clarify and elucidate the interactions among (1) changes in climate, (2) ecosystems and (3) social, economic and cultural impacts on fishing communities;
- Explore the implications and effectiveness of large marine protected areas including intergenerational losses of knowledge due to lack of access to traditional fishing areas;
- Cultural knowledge and practices for adapting to changing climate in the past and how they might contribute to future climate adaptation.
- Enhanced information on social, economic and cultural impacts of a changing climate and increased pressure on the ocean and its resources.

- Analysis of potential relationship between traditional runs of fish and climate change indicators.
- Explore the use of electronic monitoring and autonomous vehicles including small vessel prototypes.
- Explore additional and/or alternative climate and ocean that may have important effects on archipelagic fisheries systems including:
    - Ocean currents and anomalies;
    - Near-surface wind velocities and anomalies;
    - Wave forcing anomalies and wave power;
    - Storm frequency;
    - Estimates of phytoplankton abundance and size from satellite remotely-sensed SST and chlorophyll measurements;
    - Nutrients;
    - Eddy kinetic energy (EKE) which can be derived from satellite and remotely-sensed sea surface height data and can be indicative of productivity-enhancing eddies;
    - Time series of species richness and diversity from catch data which could potentially provide insight into how the ecosystem is responding to physical climate influences;
    - Identifying and monitoring key socio-economic and cultural indicators of the impacts of changing climate on resources, fishing communities, operations and resilience and;
    - Cultural knowledge and practices for adapting to changing climate in the past and how they might contribute to future climate adaptation.

2.4.5 A Look to the Future

Future Annual Reports will include additional indicators as they become available and their relevance to the development, evaluation and revision of ecosystem-fishery plans becomes clear. Working with national and jurisdictional partners, the Council will make all datasets used in the preparation of this and future reports available and easily accessible.

2.5 Essential Fish Habitat

2.5.1 Introduction

The Magnuson-Stevens Fishery Conservation and Management Act includes provisions concerning the identification and conservation of essential fish habitat (EFH), and under the EFH final rule, habitat areas of particular concern (HAPC) (50 Code of Federal Regulations [CFR] 600.815). The Magnuson-Stevens Act defines EFH as “those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity.” HAPC are those areas of EFH identified pursuant to 50 CFR 600.815(a)(8), and meeting one or more of the following considerations: (1) ecological function provided by the habitat is important; (2) habitat is sensitive to human-induced environmental degradation; (3) development activities are, or will be, stressing the habitat type; or (4) the habitat type is rare.

The National Marine Fisheries Service (NMFS) and regional Fishery Management Councils (Councils) must describe and identify EFH in fishery management plans (FMPs), minimize to the extent practicable the adverse effects of fishing on EFH, and identify other actions to encourage the conservation and enhancement of EFH. Federal agencies that authorize, fund, or undertake actions that may adversely affect EFH must consult with NMFS, and NMFS must provide conservation recommendations to federal and state agencies regarding actions that would adversely affect EFH. Councils also have the authority to comment on federal or state agency actions that would adversely affect the habitat, including EFH, of managed species.

The EFH Final Rule strongly recommends regional fisheries management councils and NMFS to conduct a review and revision of the EFH components of fisheries management plans every five years (600.815(a)(10)). The council's FEPs state that new EFH information should be reviewed, as necessary, during preparation of the annual reports by the Plan Teams. Additionally, the EFH Final Rule states "Councils should report on their review of EFH information as part of the annual Stock Assessment and Fishery Evaluation (SAFE) report prepared pursuant to §600.315(e)." The habitat portion of the annual report is designed to meet the FEP requirements and EFH Final Rule guidelines regarding EFH reviews.

National Standard 2 guidelines recommend that the SAFE report summarize the best scientific information available concerning the past, present, and possible future condition of EFH described by the FEPs.

2.5.2 EFH Information

The EFH components of fisheries management plans include the description and identification of EFH, lists of prey species and locations for each managed species, and optionally, habitat areas of particular concern. Impact-oriented components of FMPs include federal fishing activities that may adversely affect EFH; non-federal fishing activities that may adversely affect EFH; non-fishing activities that may adversely affect EFH; conservation and enhancement recommendations; and a cumulative impacts analysis on EFH. The last two components include the research and information needs section, which feeds into the Council's Five Year Research Priorities, and the EFH update procedure, which is described in the FEP but implemented in the annual report.

The Council has described EFH for five management unit species (MUS) under its management authority: pelagic (PMUS), bottomfish (BMUS), crustaceans (CMUS), coral reef ecosystem (CREMUS), and precious corals (PCMUS). The Pacific Remote Island Area (PRIA) FEP describes EFH for the BMUS, CMUS, CREMUS, and PCMUS.

EFH reviews of the biological components, including the description and identification of EFH, lists of prey species and locations, and HAPC, consist of three to four parts:

- Updated species descriptions, which can be found appended to the SAFE report. These can be used to directly update the FEP.
- Updated EFH levels of information tables, which can be found in Section **Error! Reference source not found.**.

- Updated research and information needs, which can be found in Section **Error! Reference source not found.**. These can be used to directly update the FEP.
- An analysis that distinguishes EFH from all potential habitats used by the species, which is the basis for an options paper for the Council. This part is developed if enough information exists to refine EFH.

2.5.2.1 Habitat Objectives of FEP

The habitat objective of the FEP is to refine EFH and minimize impacts to EFH, with the following sub objectives:

a. Review EFH and HAPC designations every five years based on the best available scientific information and update such designations based on the best available scientific information, when available

b. Identify and prioritize research to: assess adverse impacts to EFH and HAPC from fishing (including aquaculture) and non-fishing activities, including, but not limited to, activities that introduce land-based pollution into the marine environment.

This annual report reviews the precious coral EFH components and non-fishing impacts components, resetting the five-year timeline for review. The Council's support of non-fishing activities research is monitored through the program plan and five year research priorities, not the annual report.

2.5.2.2 Response to Previous Council Recommendations

At its 168th meeting held in Honolulu, HI, the Council adopted the EFH Agreement and directed staff to incorporate it into the Regional Operating Agreement, as necessary. The habitat expert on the plan team is ideally the PIFSC staffer with 5 year EFH responsibilities outlined in the EFH Agreement. The Plan Team reviews EFH information as necessary and recommends update to the Council.

2.5.3 Habitat Use by MUS and Trends in Habitat Condition

The Pacific Remote Island Areas comprise the U.S. possessions of Baker Island, Howland Island, Jarvis Island, Johnston Atoll, Kingman Reef, Wake Island, Palmyra Atoll, and Midway Atoll (Figure 33). However, because Midway is located in the Hawaiian archipelago, it is included in the Hawaii Archipelago FEP[2]. Therefore, neither the "Pacific Remote Island Areas" nor "PRIA" include Midway Atoll, for the purpose of federal fisheries management.

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[2] Midway is not administered civilly by the State of Hawaii.

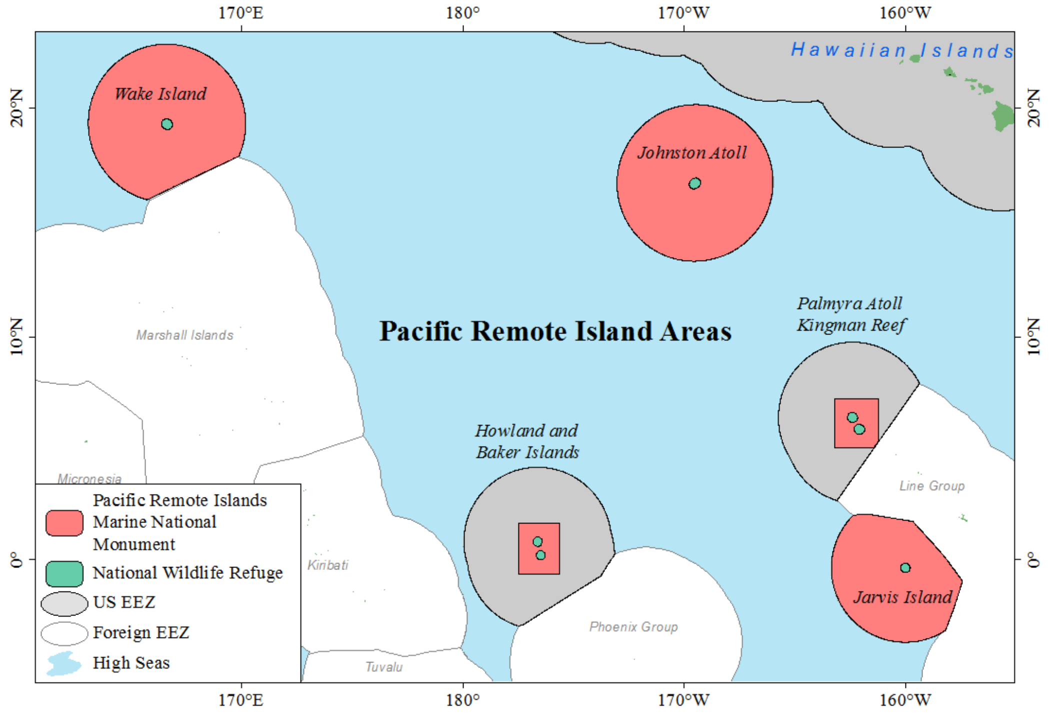


Figure 33. Pacific Remote Island Areas.

Baker Island is part of the Phoenix Islands archipelago. It is located approximately 1,600 nautical miles to the southwest of Honolulu at 0° 13' N and 176° 38' W. Baker is a coral-topped seamount surrounded by a narrow-fringing reef that drops steeply very close to the shore. The total amount of emergent land area of Baker Island is 1.4 square kilometers.

Howland Island lies approximately 35 miles due north of Baker Island and is also part of the Phoenix Islands archipelago. The island, which is the emergent top of a seamount, is fringed by a relatively flat coral reef that drops off sharply. Howland Island is approximately 1.5 miles long and 0.5 miles wide. The island is flat and supports some grasses and small shrubs. The total land area is 1.6 square kilometers.

Jarvis Island, which is part of the Line Island archipelago, is located approximately 1,300 miles south of Honolulu and 1,000 miles east of Baker Island. It sits 23 miles south of the Equator at 160° 01' W. Jarvis Island is a relatively flat, sandy coral island with a 15–20-ft beach rise. Its total land area is 4.5 square kilometers. It experiences a very dry climate.

Palmyra Atoll is a low-lying coral atoll system comprised of approximately 52 islets surrounding three central lagoons. It is approximately 1,050 nautical miles south of Honolulu and is located at 5° 53' N and 162° 05' W. It is situated about halfway between Hawaii and American Samoa. Palmyra Atoll is located in the intertropical convergence zone, an area of high rainfall.

Kingman Reef is located 33 nautical miles northwest of Palmyra Atoll at 6° 23' N and 162° 24' W. Along with Palmyra, it is at the northern end of the Line Island archipelago. Kingman is actually a series of fringing reefs around a central lagoon with no emergent islets that support vegetation.

Wake Island is located at 19° 18' N and 166° 35' E, and is the northernmost atoll of the Marshall Islands group, located approximately 2,100 miles west of Hawaii. Wake Island has a total land area of 6.5 square kilometers and comprises three islets: Wake, Peale, and Wilkes.

Johnston Atoll is located at 16° 44' N and 169° 31' W and is approximately 720 nautical miles southwest of Honolulu. French Frigate Shoals in the NWHI, about 450 nautical miles to the northwest, is the nearest land mass. Johnston Atoll is an egg-shaped coral reef and lagoon complex comprised of four small islands totaling 2.8 square kilometers. The complex resides on a relatively flat, shallow platform approximately 34 kilometers in circumference. Johnston Island, the largest and main island, is natural, but has been enlarged by dredge and fill operations. Sand Island is composed of a naturally-formed island on its eastern portion and is connected by a narrow, man-made causeway to a dredged coral island at its western portion. The remaining two islands, North Island and East Island, are completely man-made from dredged coral.

All commercial activity is prohibited within the Pacific Remote Island Area Marine National Monument, which is 50 nautical miles surrounding Palmyra Atoll and Kingman Reef and Howland and Baker Islands, and the entire US EEZ surrounding Johnston Atoll, Wake, and Jarvis Island.

Essential fish habitat in the PRIA for the four MUS comprises all substrate from the shoreline to the 700 m isobath (Figure 34). The entire water column is described as EFH from the shoreline to the 700 m isobath, and the water column to a depth of 400 m is described as EFH from the 700 m isobath to the limit or boundary of the exclusive economic zone (EEZ). While the coral reef ecosystems surrounding the islands in the PRIA have been the subject of a comprehensive monitoring program through the PIFSC Coral Reef Ecosystem Division (CRED) biennially since 2002, surveys are focused on the nearshore environments surrounding the islands, atolls and reefs (PIBHMC).

The mission of the PIFSC Coral Reef Ecosystem Division (CRED) is to “provide high-quality, scientific information about the status of coral reef ecosystems of the U.S. Pacific islands to the public, resource managers, and policymakers on local, regional, national, and international levels” (PIFSC 2011). CRED’s Reef Assessment and Monitoring Program (RAMP) conducts comprehensive ecosystem monitoring surveys at about 50 island, atoll, and shallow bank sites in the Western Pacific Region on a one to three year schedule (PIFSC 2008). CRED coral reef monitoring reports provide the most comprehensive description of nearshore habitat quality in the region. The benthic habitat mapping program provides information on the quantity of habitat.

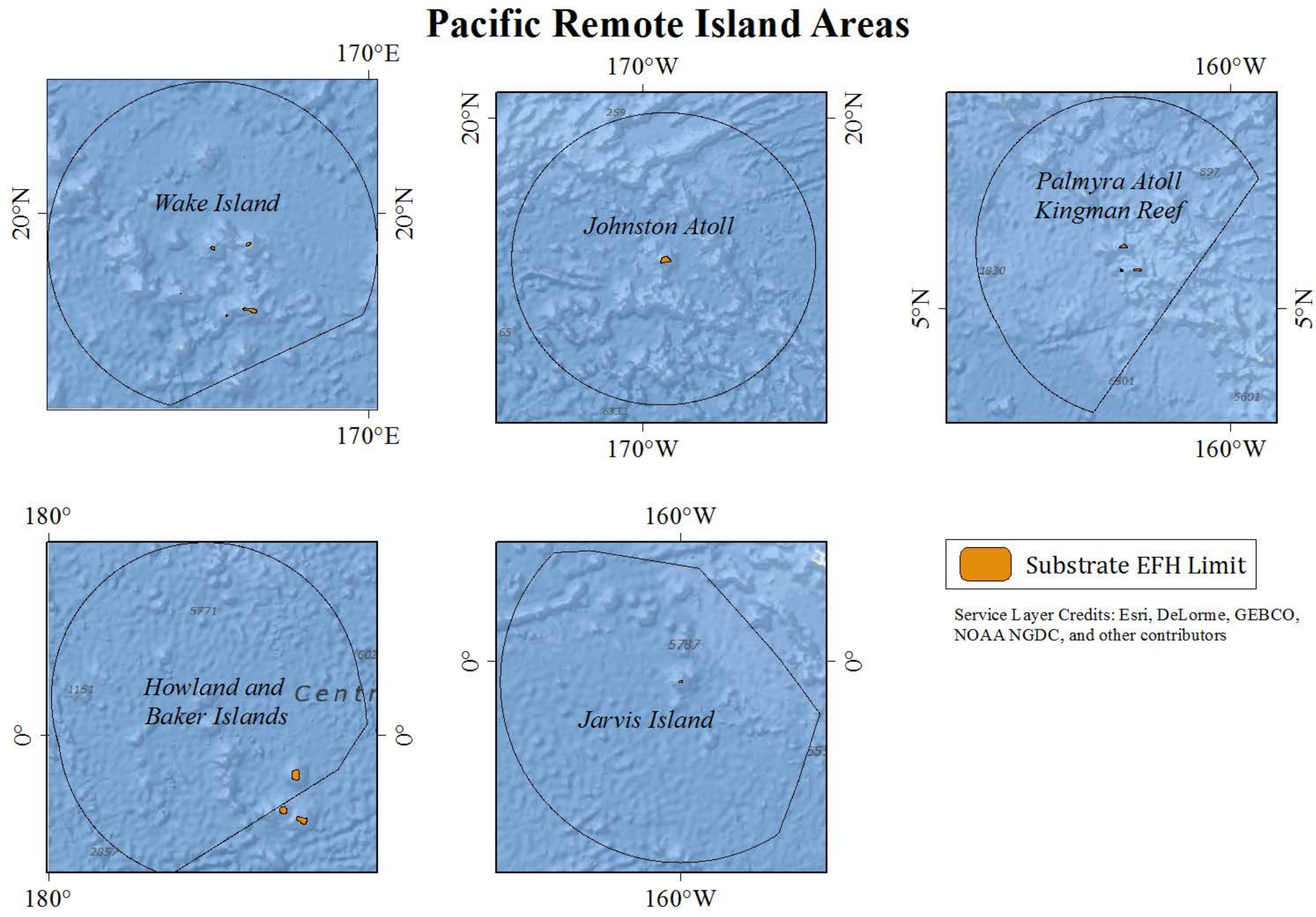


Figure 34. Substrate EFH Limit of 700 m isobath around the PRIA. Data Source: GMRT.

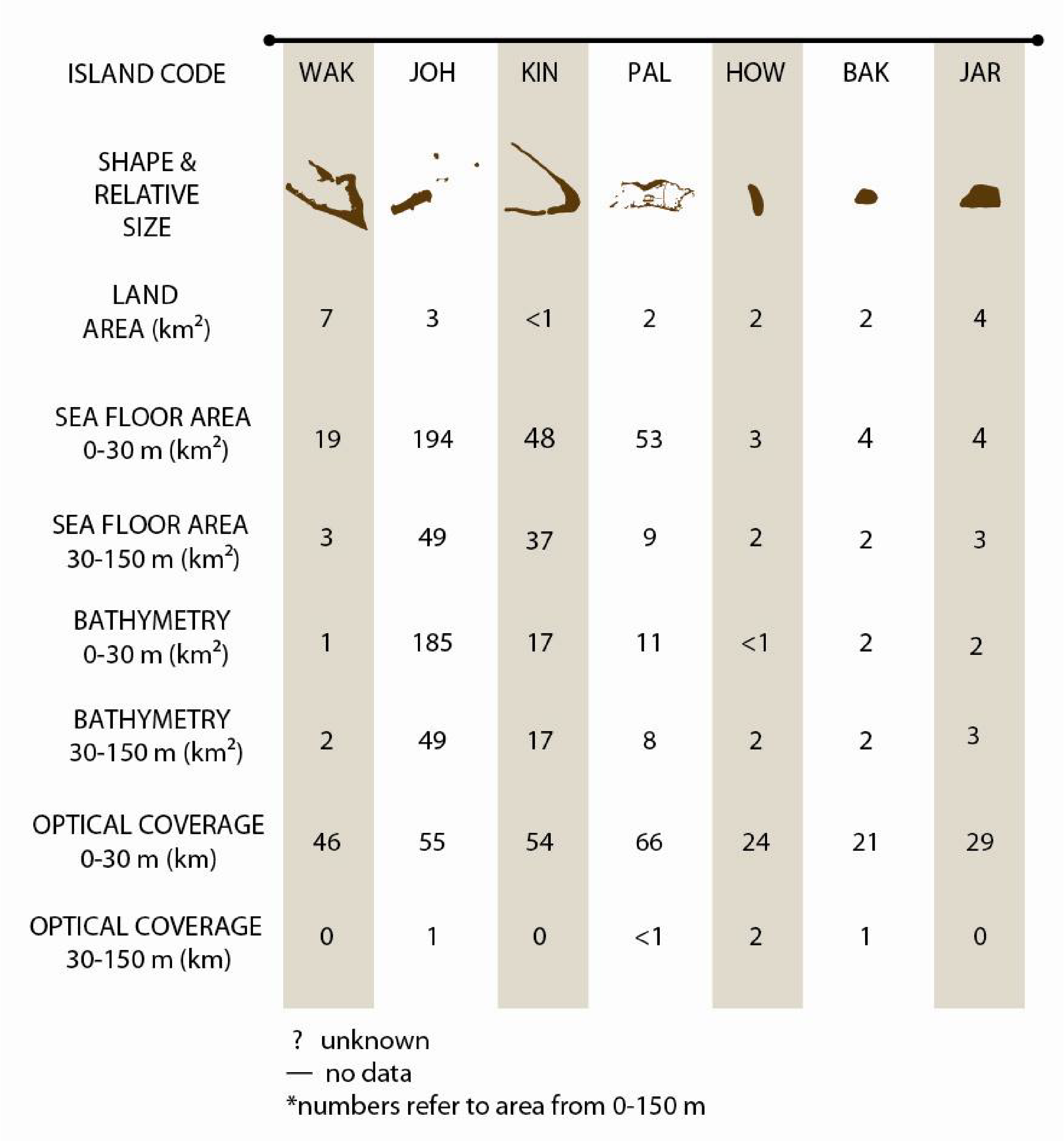
2.5.1.1 Habitat Mapping

Mapping products for the PRIA are available from the Pacific Islands Benthic Habitat Mapping Center.

Table 6. Summary of habitat mapping in the PRIA.

Depth Range	**Timeline/Mapping Product**	**Progress**	**Source**
0-30 m	IKONOS Benthic Habitat Maps	Palmyra only	CRCP 2011
	2000-2010 Bathymetry	67%	DesRochers 2016
	2011-2015 Multibeam Bathymetry		DesRochers 2016
	2011-2015 Satellite Worldview 2 Bathymetry	1%	DesRochers 2016
30-150 m	2000-2010 Bathymetry	79%	DesRochers 2016
	2011-2015 Multibeam Bathymetry	-	DesRochers 2016
15 to 2500 m	Multibeam bathymetry	Complete at Jarvis, Howland, and Baker Islands	Pacific Islands Benthic Habitat Mapping Center
	Derived Products	Backscatter available for all Geomorphology products for Johnston, Howland, Baker, Wake	Pacific Islands Benthic Habitat Mapping Center

The land and seafloor area surrounding the islands and atolls of the PRIA as well as primary data coverage are reproduced from CRCP 2011 in Figure 35.



ISLAND CODE	WAK	JOH	KIN	PAL	HOW	BAK	JAR
SHAPE & RELATIVE SIZE							
LAND AREA (km^2)	7	3	<1	2	2	2	4
SEA FLOOR AREA 0-30 m (km^2)	19	194	48	53	3	4	4
SEA FLOOR AREA 30-150 m (km^2)	3	49	37	9	2	2	3
BATHYMETRY 0-30 m (km^2)	1	185	17	11	<1	2	2
BATHYMETRY 30-150 m (km^2)	2	49	17	8	2	2	3
OPTICAL COVERAGE 0-30 m (km)	46	55	54	66	24	21	29
OPTICAL COVERAGE 30-150 m (km)	0	1	0	<1	2	1	0

? unknown
— no data
*numbers refer to area from 0-150 m

Figure **35**. PRIA Land and Seafloor Area and Primary Data Coverage from CRCP 2011.

2.5.1.2 Benthic Habitat

Juvenile and adult life stages of coral reef MUS and crustaceans including spiny and slipper lobsters and Kona crab extends from the shoreline to the 100 m isobath (64 FR 19067, April 19, 1999). All benthic habitat is considered EFH for crustaceans species (64 FR 19067, April 19, 1999), while the type of bottom habitat varies by family for coral reef species (69 FR 8336, February 24, 2004). Juvenile and adult bottomfish EFH extends from the shoreline to the 400 m isobath (64 FR 19067, April 19, 1999), and juvenile and adult deepwater shrimp habitat extends from the 300 m isobath to the 700 m isobath (73 FR 70603, November 21, 2008). Table 7 shows the depths of geologic features, the occurrence of MUS EFH at that feature, and the availability of long-term monitoring data at diving depths.

Table 7. Occurrence of EFH by feature. 1PIBMHC

Feature	Summit Minimum Depth	Coral Reef/Crustaceans exc. Deepwater Shrimp	Bottomfish	Deepwater Shrimp	CRED Long Term Monitoring
Johnston Atoll	Emergent	✓	✓	✓	✓
Palmyra	Emergent	✓	✓	✓	✓
Kingman Reef	Emergent	✓	✓	✓	✓
Extensive banktop 80 km SW of Kingman		?	?	?	
Jarvis Island	Emergent	✓	✓	✓	✓
Howland Island	Emergent	✓	✓	✓	✓
Baker Island	Emergent	✓	✓	✓	✓
Southeast of Baker	?	?	?	✓	
Wake Island	Emergent	✓	✓	✓	✓
South of Wake	?	?	?	✓	

2.5.1.2.1 RAMP Indicators

Benthic percent cover of coral, macroalgae, and crustose coralline algae from CRED are found in the following tables. CRED uses the benthic towed-diver survey method to monitor changes in benthic composition. In this method, “a pair of scuba divers (one collecting fish data, the other collecting benthic data) is towed about one m above the reef roughly 60 m behind a small boat at a constant speed of about 1.5 kt. Each diver maneuvers a towboard platform, which is connected to the boat by a bridle and towline and outfitted with a communications telegraph and various survey equipment, including a downward-facing digital SLR camera (Canon EOS 50D, Canon Inc., Tokyo). The benthic towed diver records general habitat complexity and type (e.g., spur and groove, pavement), percent cover by functional-group (hard corals, stressed corals, soft corals, macroalgae, crustose coralline algae, sand, and rubble) and for macroinvertebrates (crown-of-thorns seastars, sea cucumbers, free and boring urchins, and giant clams).

Towed-diver surveys are typically 50 min long and cover about two to three km of habitat. Each survey is divided into five-minute segments, with data recorded separately per segment to allow for later location of observations within the ~ 200-300 m length of each segment. Throughout each survey, latitude and longitude of the survey track are recorded on the small boat using a GPS; and after the survey, diver tracks are generated with the GPS data and a layback algorithm that accounts for position of the diver relative to the boat. (PIFSC Website, 2016).

Table 8. Mean percent cover of live coral from RAMP sites collected from towed-diver surveys in the PRIA

	2001	2002	2004	2005	2006	2007	2008	2009	2010	2011	2012	2014	2015	
Baker	35.37	49.47	38.78		32.95		41.2		47.44		42.1		34.48	
Howland	29.06	42.53	36.75		34.69		44.47		50.74		43.26		23.2	
Jarvis	24.22	26.19	30.63		28.54		27.7		26.92		25.38		39.75	
Johnston			5.01		22.95		18.38		7.94		10.89		7.46	
Kingman	39.77	49.51	38.35		24.59		33.13		35.56		37.11		41.92	
Palmyra	24.95	31.99	35.07		22.66		25.02		35.35		31.11		42.77	
Wake				31.98		19.29		22.56		31.4		32.34		

Table 9. Mean percent cover of macroalgae from RAMP sites collected from towed-diver surveys in the PRIA

	2001	2002	2004	2005	2006	2007	2008	2009	2010	2011	2012	2014	2015	
Baker	12.33	2.11	12.63		9.29		8.09		1.6		8.05		2.15	
Howland	2.58	5.34	13.01		3.57		6.14		0.64		6.07		1.08	
Jarvis	28.75	10.88	25.03		38.14		24.01		7.35		7.58		3.94	
Johnston			25.06		6.9		8.82		1.57		8.49		2.49	

Kingman	4.36	5.36	27.04		7.81		7.31		3.97		5.05		2.04	
Palmyra	13.28	10.45	23.14		15.17		11.98		4.76		8.94		4.35	
Wake				22.88		18.74		12		8.3		6.8		

Table 10. Mean percent cover of crustose coralline algae from RAMP sites collected from towed-diver surveys in the PRIA

	2001	2002	2004	2005	2006	2007	2008	2009	2010	2011	2012	2014	2015	
Baker	31.66	37.57	39.61		33.43		23.09		23.4		24.03		32.8	
Howland	36.6	27.4	34.26		22.6		22.59		15.73		18.12		21.25	
Jarvis	29.11	29.56	34.76		24.23		11.82		30.29		24.2		27.48	
Johnston			30.54		19.5		16.07		17.13		17.49		17.45	
Kingman	33.04	16.4	17.49		23.5		13.45		9.2		8.45		9.64	
Palmyra	38.46	24.46	27.26		26.3		18.02		13.87		17.09		10.28	
Wake				1.01		6.43		3.87		4.15		1.13		

2.5.1.3 Oceanography and Water Quality

The water column is also designated as EFH for selected MUS life stages at various depths. For larval stages of all species except deepwater shrimp, the water column is EFH from the shoreline to the EEZ. Coral reef species egg and larval EFH is to a depth of 100 m; crustaceans, 150m; and bottomfish, 400 m. Please see the Ecosystem and Climate Change section for information related to oceanography and water quality.

2.5.4 Report on Review of EFH Information

Two EFH reviews were completed this year:

- Review of precious corals biological components (Appendix C)
- Omnibus review of non-fishing impacts to EFH, cumulative impacts, and conservation and enhancement recommendations (Appendix D)

2.5.5 EFH Levels

NMFS guidelines codified at 50 C.F.R. § 600.815 recommend Councils organize data used to describe and identify EFH into the following four levels:

1. Level 1: Distribution data are available for some or all portions of the geographic range of the species.
2. Level 2: Habitat-related densities of the species are available.
3. Level 3: Growth, reproduction, or survival rates within habitats are available.
4. Level 4: Production rates by habitat are available.

The Council adopted a fifth level, denoted Level 0, for situations in which there is no information available about the geographic extent of a particular managed species' life stage. The existing level of data for individual MUS in each fishery are presented in tables per fishery. Each fishery section also includes the description of EFH, the method used to assess the value of the habitat to the species, description of data sources used if there was analysis, and description of method for analysis. A section summarizing the annual review that was performed follows.

2.5.1.4 Precious Corals

Essential Fish Habitat for precious corals was originally designated in Amendment 4 to the Precious Corals Fishery Management Plan (64 FR 19067, April 19, 1999), using the level of data found in the table.

Table 11. Level of EFH information available for the Western Pacific precious corals management unit species complex.

Note: all observations are from the Hawaiian Islands.

Species	Pelagic phase (larval stage)	Benthic phase	Source(s)
Pink Coral (*Corallium)*			
Pleurocorallium secundum	0	1	Figueroa & Baco, 2014

Species	**Pelagic phase (larval stage)**	**Benthic phase**	**Source(s)**
(prev. *Corallium secundum)*			HURL Database
C. regale	0	1	HURL Database
Hemicorallium laauense (prev. C. laauense)	0	1	HURL Database
Gold Coral			
Kulamanamana haumeaae (prev.	0	1	Sinniger, *et al.* (2013) HURL Database
Callogorgia gilberti	0	1	HURL Database
Narella spp.	0	1	HURL Database
Bamboo Coral			
Lepidisis olapa	0	1	HURL Database
Acanella spp.	0	1	HURL Database
Black Coral			
Antipathes griggi (prev. *Antipathes dichotoma)*	0	2	Opresko, 2009 HURL Database
A. grandis	0	1	HURL Database
Myriopathes ulex (prev. *A. ulex)*	0	1	Opresko, 2009 HURL Database

2.5.1.5 Bottomfish and Seamount Groundfish

Essential Fish Habitat for bottomfish and seamount groundfish was originally designated in Amendment 6 to the Bottomfish and Seamount Groundfish FMP (64 FR 19067, April 19, 1999).

Table 12. Level of EFH information available for the Western Pacific bottomfish and seamount groundfish management unit species complex.

Life History Stage	Eggs	Larvae	Juvenile	Adult
Bottomfish: (scientific/english common)				
Aphareus rutilans (red snapper/silvermouth)	0	0	0	2
Aprion virescens (gray snapper/jobfish)	0	0	1	2
Caranx ignoblis (giant trevally/jack)	0	0	1	2
C lugubris (black trevally/jack)	0	0	0	2
Epinephelus faciatus (blacktip grouper)	0	0	0	1
E quernus (sea bass)	0	0	1	2
Etelis carbunculus (red snapper)	0	0	1	2
E coruscans (red snapper)	0	0	1	2
Lethrinus amboinensis (ambon emperor)	0	0	0	1
L rubrioperculatus (redgill emperor)	0	0	0	1
Lutjanus kasmira (blueline snapper)	0	0	1	1
Pristipomoides auricilla (yellowtail snapper)	0	0	0	2
P filamentosus (pink snapper)	0	0	1	2
P flavipinnis (yelloweye snapper)	0	0	0	2

Life History Stage	Eggs	Larvae	Juvenile	Adult
P seiboldi (pink snapper)	0	0	1	2
P zonatus (snapper)	0	0	0	2
Pseudocaranx dentex (thicklip trevally)	0	0	1	2
Seriola dumerili (amberjack)	0	0	0	2
Variola louti (lunartail grouper)	0	0	0	2
Seamount Groundfish:				
Beryx splendens (alfonsin)	0	1	2	2
Hyperoglyphe japonica (ratfish/butterfish)	0	0	0	1
Pseudopentaceros richardsoni (armorhead)	0	1	1	3

2.5.1.6 Crustaceans

Essential Fish Habitat for crustaceans MUS was originally designated in Amendment 10 to the Crustaceans FMP (64 FR 19067, April 19, 1999). EFH definitions were also approved for deepwater shrimp through an amendment to the Crustaceans FMP in 2008 (73 FR 70603, November 21, 2008).

Table 13. Level of EFH information available for the Western Pacific crustaceans management unit species complex.

Life History Stage	Eggs	Larvae	Juvenile	Adult
Crustaceans: (english common\scientific)				
Spiny lobster (*Panulirus marginatus*)	2	1	1-2	2-3
Spiny lobster (*Panulirus pencillatus*)	1	1	1	2
Common slipper lobster (*Scyllarides squammosus*)	2	1	1	2-3
Ridgeback slipper lobster (*Scyllarides haanii*)	2	0	1	2-3
Chinese slipper lobster (*Parribacus antarcticus*)	2	0	1	2-3
Kona crab (*Ranina ranina*)	1	0	1	1-2

2.5.1.7 Coral Reef

Essential Fish Habitat for coral reef ecosystem species was originally designated in the Coral Reef Ecosystem FMP (69 FR 8336, February 24, 2004). An EFH review of CREMUS will not be undertaken until the Council completes its process of redesignating certain CREMUS into the ecosystem component classification. Ecosystem component species do not require EFH designations, as they are not a managed species.

2.5.6 Research and Information Needs

Based, in part, on the information provided in the tables above the Council identified the following scientific data which are needed to more effectively address the EFH provisions:

2.5.6.1 All FMP Fisheries

- Distribution of early life history stages (eggs and larvae) of management unit species by habitat
- Juvenile habitat (including physical, chemical, and biological features that determine suitable juvenile habitat)
- Food habits (feeding depth, major prey species etc)
- Habitat-related densities for all MUS life history stages
- Growth, reproduction and survival rates for MUS within habitats

2.5.6.2 Bottomfish Fishery

- Inventory of marine habitats in the EEZ of the Western Pacific region
- Data to obtain a better SPR estimate for American Samoa's bottomfish complex
- Baseline (virgin stock) parameters (CPUE, percent immature) for the Guam/NMI deep-water and shallow-water bottomfish complexes
- High resolution maps of bottom topography/currents/water masses/primary productivity
- Habitat utilization patterns for different life history stages and species

2.5.6.3 Crustaceans Fishery

- Identification of post-larval settlement habitat of all CMUS
- Identification of "source/sink" relationships in the NWHI and other regions (ie, relationships between spawning sites settlement using circulation models, genetic techniques, etc)
- Establish baseline parameters (CPUE) for the Guam/Northern Marinas crustacean populations
- Research to determine habitat-related densities for all CMUS life history stages in American Samoa, Guam, Hawaii and NMI
- High resolution mapping of bottom topography, bathymetry, currents, substrate types, algal beds, habitat relief

2.5.6.4 Precious Corals Fishery

- Distribution, abundance and status of precious corals in the PRIA.

2.5.7 References

Annette DesRochers. "Benthic Habitat Mapping." NOAA Fisheries Center, Honolulu, HI. Presentation. April 6, 2016.

Coral Reef Ecosystem Program; Pacific Islands Fisheries Science Center 2016. Benthic Percent

Cover Derived from Analysis of Benthic Images Collected during Towed-diver Surveys of the U.S. Pacific Reefs Since 2003 (NCEI Accession <unassigned>). NOAA National Centers for Environmental Information. Unpublished Dataset. April 5, 2016.

Miller, J; Battista, T.; Pritchett, A; Rohmann, S; Rooney, J. Coral Reef Conservation Program

Mapping Achievements and Unmet Needs. March 14, 2011. 68 p.

Pacific Islands Fisheries Science Center Ecosystem Sciences Coral Reef Ecosystem Survey

Methods. Benthic Monitoring. http://www.pifsc.noaa.gov/cred/survey_methods.php. Updated April 1, 2016. Accessed April 5, 2016.

Pacific Islands Fisheries Science Center (2011) Coral reef ecosystems of American Samoa: a

2002–2010 overview. NOAA Fisheries Pacific Islands Fisheries Science Center, PIFSC Special Publication, SP-11-02, 48 p.

2.6 Marine Planning

2.6.1 Introduction

Marine planning is a science-based tool being utilized regionally, nationally and globally to identify and address issues of multiple human uses, ecosystem health and cumulative impacts in the coastal and ocean environment. The Council's efforts to formalize incorporation of marine planning in its actions began in response to Executive Order 13547, Stewardship of the Ocean, Our Coasts, and the Great Lakes. Executive Order 13158, Marine Protected Areas (MPAs), proposes that agencies strengthen the management, protection, and conservation of existing MPAs, develop a national system of MPAs representing diverse ecosystems, and avoid causing harm to MPAs through federal activities. MPAs, or marine managed areas (MMAs) are one tool used in fisheries management and marine planning.

At its 165th meeting in March 2016, in Honolulu, Hawai`i, the Council approved the following objective for the FEPs: Consider the Implications of Spatial Management Arrangements in Council Decision-making. The following sub-objectives apply:

a. Identify and prioritize research that examines the positive and negative consequences of areas that restrict or prohibit fishing to fisheries, fishery ecosystems, and fishermen, such as the Bottomfish Fishing Restricted Areas, military installations, NWHI restrictions, and Marine Life Conservation Districts.

b. Establish effective spatially-based fishing zones.

c. Consider modifying or removing spatial-based fishing restrictions that are no longer necessary or effective in meeting their management objectives.

d. As needed, periodically evaluate the management effectiveness of existing spatial-based fishing zones in Federal waters.

In order to monitor implementation of this objective, this annual report includes the Council's spatially-based fishing restrictions or marine managed areas (MMAs), the goals associated with those, and the most recent evaluation. Council research needs are identified and prioritized through the 5 Year Research Priorities and other processes, and are not tracked in this report.

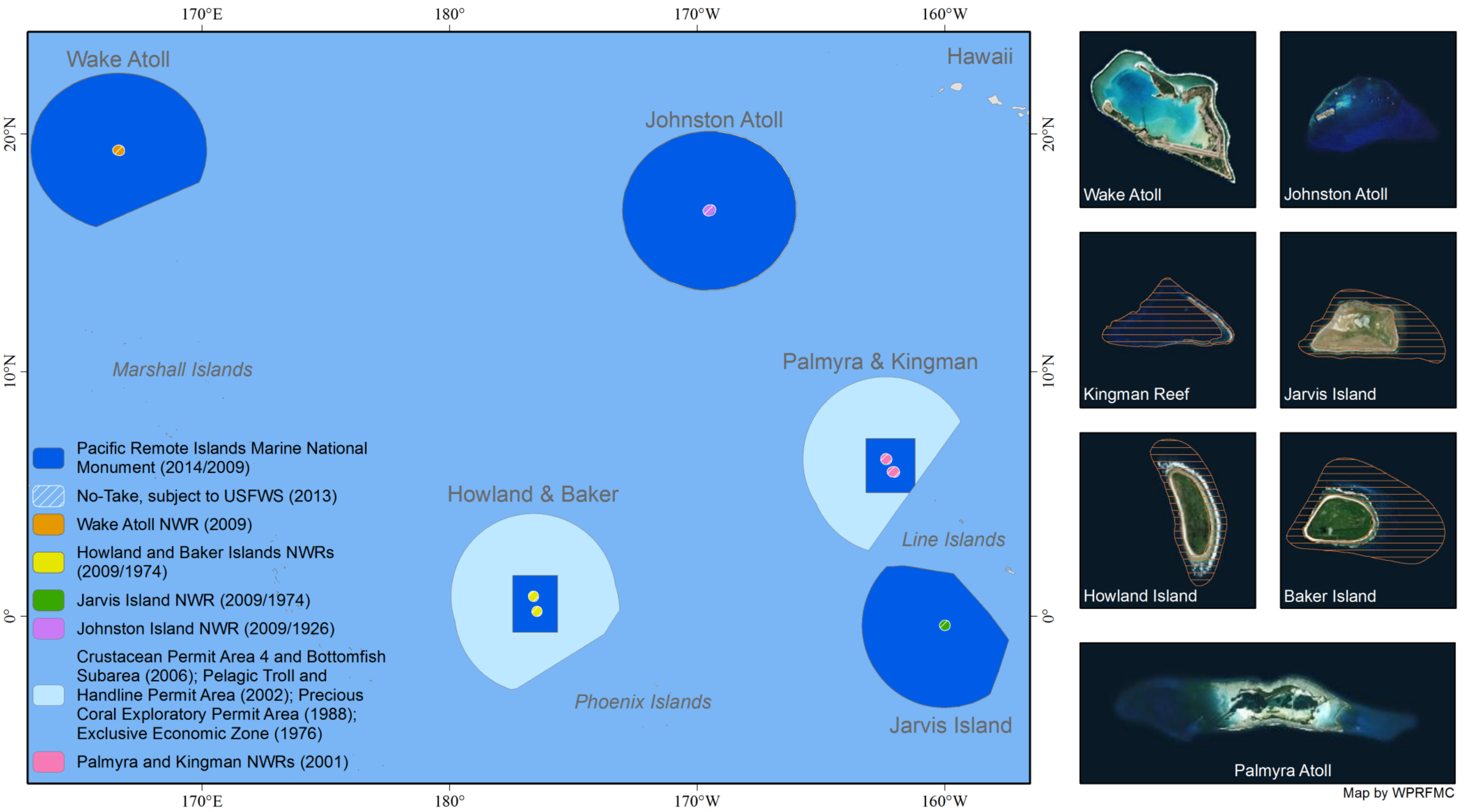
In order to meet the EFH and National Environmental Policy Act (NEPA) mandates, this annual report tracks activities that occur in the ocean that are of interest to the Council, and incidents or facilities that may contribute to cumulative impact. The National Marine Fisheries Service (NMFS) is responsible for NEPA compliance, and the Council must assess the environmental effects of ocean activities for the FEP's EFH cumulative impacts section. These are redundant efforts; therefore, this report can provide material or suggest resources to meet both mandates.

1.2.1.1 Response to Previous Council Recommendations

There are no standing Council recommendations indicating review deadlines for PRIA marine managed areas.

1.2.1.2 MMAs established under FMPs

Council-established marine managed areas (MMAs) were compiled in Table 14 from 50 CFR § 665, Western Pacific Fisheries, the Federal Register, and Council amendment documents. Geodesic areas were calculated in square kilometers in ArcGIS 10.2. All regulated fishing areas and large MMAs, including the Pacific Remote Islands Marine National Monument, are shown in Figure 36.



Map by WPRFMC

Figure 36. Regulated fishing areas of the PRIA.

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Table 14. MMAs established under FEPs from 50 CFR § 665.

Name	**FEP**	**Island**	**50 CFR /FR /Amendment Reference**	**Marine Area (km^2)**	**Fishing Restriction**	**Goals**	**Most Recent Evaluation**	**Review Deadline**
Other Restrictions								
Howland Island No-Take MPA/PRI Marine National Monument	PRIA/ Pelagic	Howland Island	665.599 and 665.799(a)(1) 69 FR 8336 Coral Reef Ecosystem FEP 78 FR 32996 PRIA FEP Am. 2	-	All Take Prohibited	Minimize adverse human impacts on coral reef resources; commercial fishing prohibited within 12 nmi	2013	-
Jarvis Island No-Take MPA/PRI Marine National Monument	PRIA/ Pelagic	Jarvis Island	665.599 and 665.799(a)(1) 69 FR 8336 Coral Reef Ecosystem FEP 78 FR 32996 PRIA FEP Am. 2	-	All Take Prohibited	Minimize adverse human impacts on coral reef resources; commercial fishing prohibited within 12 nmi	2013	-

Name	**FEP**	**Island**	**50 CFR /FR /Amendment Reference**	**Marine Area (km^2)**	**Fishing Restriction**	**Goals**	**Most Recent Evaluation**	**Review Deadline**
Baker Island No-Take MPA/PRI Marine National Monument	PRIA/ Pelagic	Baker Island	665.599 and 665.799(a)(1) 69 FR 8336 Coral Reef Ecosystem FEP 78 FR 32996 PRIA FEP Am. 2	-	All Take Prohibited	Minimize adverse human impacts on coral reef resources; commercial fishing prohibited within 12 nmi	2013	-
Kingman Reef No-Take MPA/PRI Marine National Monument	PRIA/Pelagic	Kingman Reef	665.599 and 665.799(a)(1) 69 FR 8336 Coral Reef Ecosystem FEP 78 FR 32996 PRIA FEP Am. 2	-	All Take Prohibited	Minimize adverse human impacts on coral reef resources; all fishing prohibited within 12 nmi	2013	-

Name	FEP	Island	50 CFR /FR /Amendment Reference	Marine Area (km^2)	Fishing Restriction	Goals	Most Recent Evaluation	Review Deadline
Johnston Atoll Low-Use MPA/PRI Marine National Monument	PRIA/ Pelagic	Johnston Atoll	69 FR 8336 Coral Reef Ecosystem FEP 78 FR 32996 PRIA FEP Am. 2	-	Special Permit Only	Minimize adverse human impacts on coral reef resources; superseded by prohibiting fishing within 12 nmi in Am. 2	2013	-
Palmyra Atoll Low-Use MPAs/PRI Marine National Monument	PRIA/ Pelagic	Palmyra Atoll	69 FR 8336 Coral Reef Ecosystem FEP 78 FR 32996 PRIA FEP Am. 2	-	Special Permit Only	Minimize adverse human impacts on coral reef resources; superseded by prohibiting fishing within 12 nmi in Am. 2	2013	-

Name	**FEP**	**Island**	**50 CFR /FR /Amendment Reference**	**Marine Area (km^2)**	**Fishing Restriction**	**Goals**	**Most Recent Evaluation**	**Review Deadline**
Wake Island Low-Use MPA/PRI Marine National Monument	PRIA/Pelagic	Wake Island	69 FR 8336 Coral Reef Ecosystem FEP 78 FR 32996 PRIA FEP Am. 2	-	Special Permit Only	Minimize adverse human impacts on coral reef resources; superseded by prohibiting fishing within 12 nmi in Am. 2	2013	-

2.6.2 Activities and Facilities

There are no aquaculture, alternative energy facilities, or military training and testing activities occurring in the US EEZ around the PRIAs at this time. The Plan Team will add to this section as new facilities are proposed and/or built.

2.6.3 Pacific Islands Regional Planning Body Report

The Council is a member of the Pacific Islands RPB and as such, the interests of the Council will be incorporated into the regional CMS plan. It is through the Council member that the Council may submit recommendations to the Pacific Islands RPB.

The Pacific Islands RPB met in Honolulu from February 15-16, 2017. The RPB's American Samoa Ocean Planning Team has developed its goals and objectives on which the RPB provided comments and endorsement. The RPB, by consensus, decided to:

- revise its charter with select Maritime Administration comments, a glossary or terms of reference, and handle standard operating procedure concerns through internal documentation rather than amendments to the Charter;
- kick off a Marianas Ocean Planning Team later in 2017; and
- defer the decision on beginning planning in the PRIA until an update is received on the Pacific Remote Islands Marine National Monument Management Plan at the next RPB teleconference.

The American Samoa Ocean Planning Team will continue its work concurrently with a stakeholder assessment. The Data Team will continue its work per the work plan developed in 2016.

2.6.4 References

Fisheries in the Western Pacific. Title 50 *Code of Federal Regulations*, Pt. 665. Electronic Code of Federal Regulations data current as of March 16, 2016. Viewed at http://www.ecfr.gov/cgi-bin/retrieveECFR?gp=&SID=b28abb7da3229173411daf43959fcbd1&n=50y13.0.1.1.2&r=PART&ty=HTML#_top.

Fisheries Off West Coast States and in the Western Pacific; Coral Reef Ecosystems Fishery Management Plan for the Western Pacific, Final Rule. *Federal Register* 69 (24 February 2004): 8336-8349. Downloaded from http://www.wpcouncil.org/precious/Documents/FMP/Amendment5-FR-FinalRule.pdf.

Pelagic Fisheries of the Western Pacific Region, Final Rule. *Federal Register* 56 (18 October 1991): 52214-52217. Downloaded from http://www.wpcouncil.org/pelagic/Documents/FMP/Amendment3-FR-FinalRule.pdf.

Pelagic Fisheries of the Western Pacific Region, Final Rule. *Federal Register* 57 (4 March 1992): 7661-7665. Downloaded from http://www.wpcouncil.org/pelagic/Documents/FMP/Amendment5-FR-FinalRule.pdf.

Western Pacific Fisheries; Fishing in the Marianas Trench, Pacific Remote Islands, and Rose Atoll Marine National Monuments, Final Rule. *Federal Register* 78 (3 June 2013): 32996-33007. Downloaded from http://www.wpcouncil.org/precious/Documents/FMP/Amendment5-FR-FinalRule.pdf.

Western Pacific Regional Fishery Management Council. Fishery Management Plan and Fishery Ecosystem Plan Amendments available from http://www.wpcouncil.org/.

3 DATA INTEGRATION

At the 2016 joint meeting of the Archipelagic and Pelagic Fishery Ecosystem Plan Team, the teams recommended the Council, in coordination with NMFS, organize a workshop in developing the Data Integration Chapter of the Annual/SAFE Report. The workshop was convened on November 30 and December 1, 2017. The goal of the workshop was to identify policy-relevant fishery ecosystem relationships, as well as analytical procedures that can be utilized to examine those relationships, that could be the bases of the data integration chapter ("Chapter 3") of the western Pacific region's (WPR) five annual Stock Assessment and Fishery Evaluation (SAFE) reports. Such variables include, for example, catch, number of fishing trips, primary productivity, and climate and weather attributes.

The Western Pacific Regional Fishery Management Council (Council) hosted the workshop. Participants included staff from the National Marine Fisheries Service (NMFS) Pacific Islands Fisheries Science Center (PIFSC) and Pacific Islands Regional Office (PIRO), the Council, and Triton Aquatics, a Hawaii-based consulting company.

Name	**Affiliation**	**Name**	**Affiliation**
Keith Bigelow	PIFSC	Kevin Kelley	Consultant/PIRO
Chris Boggs	PIFSC	Eric Kingma	Council
Rusty Brainard	PIFSC	Don Kobayashi	PIFSC
Paul Dalzell	Council	Tom Oliver	PIFSC
Joshua DeMello	Council	Michael Parke	PIFSC
Stefanie Dukes	PIFSC	Frank Parrish	PIFSC
Sarah Ellgen	PIRO	Marlowe Sabater	Council
Jamison Gove	PIFSC	Sylvia Spalding	Council
Justin Hospital	PIFSC	Rebecca Walker	Council
Asuka Ishizaki	Council	Mariska Weijerman	PIFSC
Ariel Jacobs	PIRO	Ivor Williams	PIFSC

Several background presentations were given to contextualize the discussions. The following were the background presentations:

1. EBFM and adaptive management in the SAFE report process
2. Examples of fishery ecosystem integration efforts from other regions
3. FEP Objectives and Management Measures

4. Past attempts at Data Integration: Environmental, Social, and Economic Variables Known to Influence Fisheries

Following these background presentations and discussions, participants were segregated into two smaller working groups to brainstorm island and pelagic fishery and environmental/ecological relationships that may be of use in the context of Chapter 3. These relationships could be bivariate or multivariate. Several guided questions were provided for every combination of variables:

1. What can we reasonably expect to learn from or monitor with the results?
2. How does it inform Council decision-making, consistent with the purposes of the FEP?
3. Is it part of an ongoing research initiative?

The archipelagic fisheries group developed nearly 30 relationships to examine across bottomfish, coral reef, and crustacean fisheries, while the pelagic breakout group developed 11 relationships for pelagic fisheries, including protected species. The prioritized relationships are as follows:

Relationships	**FEP**	**Score**	**Rank**
bottomfish catch/effort/cpue/species composition and benthos/substrate (depth, structure)	All	22	3
bottomfish catch/effort/cpue/species composition and PDO	All	20	3
coral reef fish fishery/biomass and temperature-derived variable	All	20	3
akule/opelu and rainfall (HI and GU)	HI	20	3
bottomfish catchability and wind speed	All	19	3
reef fish catch and biomass and Chl-a (with phase lag)	All	19	3
bottomfish catch and CPUE and moon phase	All	19	3
bottomfish catch/effort/cpue/species composition and sea-level height (eddy feature)	All	18	2
coral reef fish fishery/biomass and PDO	All	18	2
green/red spiny lobster catch/cpue vertical relief	HI	18	2
green/red spiny lobster and PDO	HI	18	2
bottomfish catchability and fishing conditions (surface, subsurface current, speed and direction)	All	17	2
coral reef fish abundance and moon phase	All	17	2
coral reef fish fishery/biomass and El Nino	All	17	2
coral reef fish fishery/biomass and sea-level height	All	17	2
coral reef fish fishery/biomass and pH	All	17	2
bottomfish catch/effort/cpue/species composition and temperature-derived	All	16	2

variable (temp. at depth)			
bottomfish catch/effort/cpue/species composition and Chl-a	All	16	2
bottomfish catch/effort/cpue/species composition and rainfall	All	16	2
coral reef fish catch and biomass (family; trophic guilds) and structural complexity and benthic habitat information	All	16	2
bottomfish catch/effort/cpue/species composition and DO	All	15	2
coral reef fish fishery/biomass and rainfall	All	14	2
bottomfish catch/effort/cpue/species composition and pH	All	13	2
bottomfish CPUE and shark/predator biomass/abundance	All	12	2
coral reef fish fishery/biomass and salinity	All	12	2
coral reef fish fishery/biomass and DO	All	12	2
bottomfish catch/effort/cpue/species composition and salinity	All	10	1

The development of the data integration chapter is work in progress that has a 2-3 year timeline. The workshop produced a long list of fishery and ecosystem variable combinations that comprise a significant workload that the participants could not currently take on. The Council hired a contractor that will conduct the exploratory data analysis on the different variable combinations and determine which relationships are worth using in Chapter 3. The contractor is expected to deliver the results at the end of 2017.

Appendix A: Species list table for the PRIA FEP

The PRIA species list and FSSI status will be made available in subsequent reports as resources allow. Please see the PRIA FEP and implementing regulations for the list of managed species.

Appendix B. List of Protected Species and Designated Critical Habitat.

Table B1. Protected species found or reasonably believed to be found in or near PRIA waters.

Common name	Scientific name	ESA listing status	MMPA status	Occurrence	References
Seabirds					
Audubon's Shearwater	*Puffinus lherminieri*	Not Listed	N/A	Breeding	Sala et al. 2014
Band-Rumped Storm-Petrel	*Oceanodroma castro*	Not Listed	N/A	Visitor	Sala et al. 2014
Black Noddy	*Anous minutus*	Not Listed	N/A	Breeding	Sala et al. 2014
Black-Footed Albatross	*Phoebastria nigripes*	Not Listed	N/A	Breeding	Sala et al. 2014
Black-Naped Tern	*Sterna sumatrana*	Not Listed	N/A	Visitor	Sala et al. 2014
Black-Winged Petrel	*Pterodroma nigripennis*	Not Listed	N/A	Visitor	Sala et al. 2014
Blue-Gray Noddy	*Procelsterna cerulea*	Not Listed	N/A	Breeding	Sala et al. 2014
Bonin Petrel	*Pterodroma hypoleuca*	Not Listed	N/A	Visitor	Sala et al. 2014
Bridled Tern	*Onychoprion anaethetus*	Not Listed	N/A	Visitor	Sala et al. 2014
Brown Booby	*Sula leucogaster*	Not Listed	N/A	Breeding	Sala et al. 2014
Brown Noddy	*Anous stolidus*	Not Listed	N/A	Breeding	Sala et al. 2014
Bulwer's Petrel	*Bulweria bulwerii*	Not Listed	N/A	Breeding	Sala et al. 2014
Christmas Shearwater	*Puffinus nativitatis*	Not Listed	N/A	Breeding	Sala et al. 2014
Fairy Tern	*Sternula nereis*	Not Listed	N/A	Breeding	Sala et al. 2014
Flesh-Footed Shearwater	*Ardenna carneipes*	Not Listed	N/A	Visitor	Sala et al. 2014
Gould's Petrel	*Pterodroma leucoptera*	Not Listed	N/A	Visitor	Sala et al. 2014
Great Crested Tern	*Thalasseus bergii*	Not Listed	N/A	Visitor	Sala et al. 2014
Great Frigatebird	*Fregata minor*	Not Listed	N/A	Breeding	Sala et al. 2014
Gray-Backed Tern	*Onychoprion lunatus*	Not Listed	N/A	Breeding	Sala et al. 2014
Hawaiian Petrel	*Pterodroma sandwichensis (Pterodroma phaeopygia sandwichensis)*	Endangered	N/A	Visitor	32 FR 4001, Sala et al. 2014
Herald Petrel	*Pterodroma heraldica*	Not Listed	N/A	Visitor	Sala et al. 2014
Kermadec Petrel	*Pterodroma neglecta*	Not Listed	N/A	Visitor	Sala et al. 2014
Laysan Albatross	*Phoebastria immutabilis*	Not Listed	N/A	Breeding	Sala et al. 2014

Common name	Scientific name	ESA listing status	MMPA status	Occurrence	References
Lesser Frigatebird	*Fregata ariel*	Not Listed	N/A	Breeding	Sala et al. 2014
Little Shearwater	*Puffinus assimilis*	Not Listed	N/A	Visitor	Sala et al. 2014
Masked Booby	*Sula dactylatra*	Not Listed	N/A	Breeding	Sala et al. 2014
Murphy's Petrel	*Pterodroma ultima*	Not Listed	N/A	Visitor	Sala et al. 2014
Newell's Shearwater	*Puffinus newelli (Puffinus auricularis newelli)*	Threatened	N/A	Visitor	40 FR 44149, Sala et al. 2014
Phoenix Petrel	*Pterodroma alba*	Not Listed	N/A	Former breeder	Sala et al. 2014
Polynesian Storm-Petrel	*Nesofregetta fuliginosa*	Not Listed	N/A	Visitor	Sala et al. 2014
Northern Fulmar	*Fulmarus glacialis*	Not Listed	N/A	Breed and range across North Pacific Ocean.	Hatch & Nettleship 2012
Sooty Shearwater	*Ardenna grisea*	Not Listed	N/A	Breed in the southern hemisphere and migrate to the northern hemisphere.	BirdLife International 2017
Short-Tailed Albatross	*Phoebastria albatrus*	Endangered	N/A	Breed in Japan and NWHI, and range across the North Pacific Ocean.	35 FR 8495, 65 FR 46643, BirdLife International 2017
Sea turtles					
Green Sea Turtle	*Chelonia mydas*	Endangered (Central South Pacific DPS)	N/A	Occur at Wake Island and Palmyra Atoll. Few sightings around Howland, Baker, Jarvis, and Kingman reef.	43 FR 32800, 81 FR 20057, Balazs 1982
Green Sea Turtle	*Chelonia mydas*	Threatened (Central North Pacific DPS)	N/A	Forage around Johnston Atoll.	43 FR 32800, 81 FR 20057, Balazs 1985
Loggerhead Sea Turtle	*Caretta caretta*	Endangered (North Pacific DPS)	N/A	No known sightings. Found worldwide along continental shelves, bays, estuaries and lagoons of tropical, subtropical, and temperate waters.	43 FR 32800, 76 FR 58868, Dodd 1990, NMFS & USFWS 1998
Loggerhead Sea Turtle	*Caretta caretta*	Endangered (South Pacific DPS)	N/A	No known sightings. Found worldwide along continental shelves, bays, estuaries and lagoons of tropical, subtropical, and temperate waters.	43 FR 32800, 76 FR 58868, Dodd 1990, NMFS & USFWS 1998

Common name	Scientific name	ESA listing status	MMPA status	Occurrence	References
Olive Ridley Sea Turtle	*Lepidochelys olivacea*	Threatened (Entire species, except for endangered breeding population on the Pacific coast of Mexico).	N/A	No known sightings. Occur worldwide in tropical and warm temperate ocean waters.	43 FR 32800, Pitman 1990, Balacz 1982
Hawksbill Sea Turtle	*Eretmochelys imbricata*	Endangered[a]	N/A	No known sightings. Occur worldwide in tropical and subtropical waters.	35 FR 8491, Baillie & Groombridge 1996
Leatherback Sea Turtle	*Dermochelys coriacea*	Endangered[a]	N/A	No known sightings. Occur worldwide in tropical, subtropical, and subpolar waters.	35 FR 8491, Eckert et al. 2012
Marine mammals					
Bryde's Whale	*Balaenoptera edeni*	Not Listed	Non-strategic	Distributed widely across tropical and warm-temperate Pacific Ocean.	Leatherwood et al. 1982
Blue Whale	*Balaenoptera musculus*	Endangered	Strategic	Extremely rare. Distributed worldwide in tropical and warm-temperate waters.	35 FR 18319, McDonald et al. 2006, Stafford et al. 2001, Bradford et al. 2013, Northrop et al. 1971, Thompson & Friedl 1982
Fin Whale	*Balaenoptera physalus*	Endangered	Strategic	Found worldwide.	35 FR 18319, Hamilton et al. 2009
Humpback Whale	*Megaptera novaeangliae*	Delisted Due to Recovery (Hawaii DPS)	Strategic	Breed in waters around MHI during the winter.	35 FR 18319, 81 FR 62259, Childerhouse et al. 2008, Rice & Wolman 1978, Wolman & Jurasz 1976, Herman & Antinoja 1977,
Humpback Whale	*Megaptera novaeangliae*	Delisted Due to Recovery (Oceania DPS)	Strategic	Breed in Oceania waters during the winter.	35 FR 18319, 81 FR 62259, Guarrige et al. 2007, SPWRC 2008
Humpback Whale	*Megaptera novaeangliae*	Endangered (Western North Pacific DPS)	Strategic	Small population of about 1,000 that breeds in Asian waters during the winter.	35 FR 18319, 81 FR 62259, Eldredge et al. 2003; Barlow et al. 2011; Calambokidis et al. 2001, 2008

Common name	Scientific name	ESA listing status	MMPA status	Occurrence	References
Sei Whale	*Balaenoptera borealis*	Endangered	Strategic	Generally found in offshore temperate waters.	35 FR 18319, Barlow 2003, Bradford et al. 2013
Bottlenose Dolphin	*Tursiops truncatus*	Not Listed	Non-strategic	Distributed worldwide in tropical and warm-temperate waters.	Perrin et al. 2009
False Killer Whale	*Pseudorca crassidens*	Not Listed	Non-strategic	Two stocks found in or near PRIA waters: 1) Palmyra Atoll stock found within US EEZ waters around Palmyra Atoll, and 2) Hawaii pelagic stock which includes animals in waters more than 40 km from the MHI. Little known about these stocks. Found worldwide in tropical and warm-temperate waters.	Barlow et al. 2008, Bradford & Forney 2013, Stacey et al. 1994, Chivers et al. 2010
Pygmy Killer Whale	*Feresa attenuata*	Not Listed	Non-strategic	Found in tropical and subtropical waters worldwide.	Ross & Leatherwood 1994
Risso's Dolphin	*Grampus griseus*	Not Listed	Non-strategic	Found in tropical to warm-temperate waters worldwide.	Perrin et al. 2009
Rough-Toothed Dolphin	*Steno bredanensis*	Not Listed	Non-strategic	Found in tropical to warm-temperate waters worldwide.	Perrin et al. 2009
Common Dolphin	*Delphinus delphis*	Not Listed	Non-strategic	Found worldwide in temperate and subtropical seas.	Perrin et al. 2009
Short-Finned Pilot Whale	*Globicephala macrorhynchus*	Not Listed	Non-strategic	Found in tropical to warm-temperate waters worldwide. Found in waters around Johnston and Palmyra Atolls.	Shallenberger 1981, Baird et al. 2013, Bradford et al. 2013
Spinner Dolphin	*Stenella longirostris*	Not Listed	Non-strategic	Found worldwide in tropical and warm-temperate waters. Occur in shallow protected bays during the day, feed offshore at night.	Norris and Dohl 1980, Norris et al. 1994, Hill et al. 2010, Andews et al. 2010, Karczmarski 2005, Perrin et al. 2009
Spotted Dolphin	*Stenella attenuata attenuata*	Not Listed	Non-strategic	Found in tropical and subtropical waters worldwide. Sighted in waters around Palmyra and Johnston atolls.	Perrin et al. 2009, NMFS PIR unpub. Data

Common name	Scientific name	ESA listing status	MMPA status	Occurrence	References
Striped Dolphin	*Stenella coeruleoalba*	Not Listed	Non-strategic	Found in tropical to warm-temperate waters throughout the world.	Perrin et al. 2009
Guadalupe Fur Seal	*Arctocephalus townsendi*	Threatened	Strategic	No known sightings. Little known about their pelagic distribution. Breed mainly on Isla Guadalupe, Mexico.	50 FR 51252, Gallo-Reynoso et al. 2008, Fleischer 1987
Hawaiian Monk Seal	*Neomonachus schauinslandi*	Endangered[a]	Strategic	Endemic tropical seal. Occurs throughout the Hawaiian archipelago. Occasional sightings on Johnston atoll.	41 FR 51611, Antonelis et al. 2006
Northern Elephant Seal	*Mirounga angustirostris*	Not Listed	Non-strategic	Females migrate to central North Pacific to feed on pelagic prey.	Le Beouf et al. 2000
Sperm Whale	*Physeter macrocephalus*	Endangered	Strategic	Found in tropical to polar waters worldwide, most abundant cetaceans in the region.	35 FR 18319, Rice 1960, Lee 1993, Barlow 2006, Mobley et al. 2000, Shallenberger 1981
Blainville's Beaked Whale	*Mesoplodon densirostris*	Not Listed	Non-strategic	Found worldwide in tropical and temperate waters.	Mead 1989
Cuvier's Beaked Whale	*Ziphius cavirostris*	Not Listed	Non-strategic	Occur worldwide.	Heyning 1989
Sharks					
Scalloped hammerhead	*Sphyrna lewini*	Endangered (Eastern Pacific DPS)	N/A	Found in coastal areas from southern California to Peru.	Compagno 1984, Baum et al. 2007, Bester 2011
Scalloped hammerhead	*Sphyrna lewini*	Threatened (Indo-West Pacific DPS)	N/A	Occur over continental and insular shelves, and adjacent deep waters, but rarely found in waters < 22°C. Range from the intertidal and surface to depths up to 450–512 m.	Compagno 1984, Schulze-Haugen & Kohler 2003, Sanches 1991, Klimley 1993
Corals					
N/A	*Acropora globiceps*	Threatened	N/A	Occur on upper reef slopes, reef flats, and adjacent habitats in depths ranging from 0 to 8 m	Veron 2014

Common name	Scientific name	ESA listing status	MMPA status	Occurrence	References
N/A	*Acropora retusa*	Threatened	N/A	Occur in shallow reef slope and back-reef areas, such as upper reef slopes, reef flats, and shallow lagoons, and depth range is 1 to 5 m.	Veron 2014
N/A	*Acropora speciosa*	Threatened	N/A	Found in protected environments with clear water and high diversity of Acropora and steep slopes or deep, shaded waters. Depth range is 12 to 40 meters, and have been found in mesophotic habitat (40-150 m).	Veron 2014

[a] These species have critical habitat designated under the ESA. See Table A3.

Table B2. ESA-listed species' critical habitat in the Pacific Ocean[a].

Common name	Scientific name	ESA listing status	Critical habitat	References
Hawksbill Sea Turtle	*Eretmochelys imbricata*	Endangered	None in the Pacific Ocean.	63 FR 46693
Leatherback Sea Turtle	*Dermochelys coriacea*	Endangered	Approximately 16,910 square miles (43,798 square km) stretching along the California coast from Point Arena to Point Arguello east of the 3,000 meter depth contour; and 25,004 square miles (64,760 square km) stretching from Cape Flattery, Washington to Cape Blanco, Oregon east of the 2,000 meter depth contour.	77 FR 4170
Hawaiian Monk Seal	*Neomonachus schauinslandi*	Endangered	Ten areas in the Northwestern Hawaiian Islands (NWHI) and six in the main Hawaiian Islands (MHI). These areas contain one or a combination of habitat types: Preferred pupping and nursing areas, significant haul-out areas, and/or marine foraging areas, that will support conservation for the species.	53 FR 18988, 51 FR 16047, 80 FR 50925
North Pacific Right Whale	*Eubalaena japonica*	Endangered	Two specific areas are designated, one in the Gulf of Alaska and another in the Bering Sea, comprising a total of approximately 95,200 square kilometers (36,750 square miles) of marine habitat.	73 FR 19000, 71 FR 38277

[a] For maps of critical habitat, see http://www.nmfs.noaa.gov/pr/species/criticalhabitat.htm.

Appendix C: Precious Corals Species Descriptions

1 PRECIOUS CORAL SPECIES

This section is an update of Appendix 1 to the Western Pacific FEPs, "Essential Fish Habitat Species Descriptions for Western Pacific Archipelagic, and Remote Island Areas Fishery Ecosystem Plan Management Unit Species" for precious corals. Important new references and data points have been added to the original documentation. Many older observations continue to be cited because no newer studies have been completed, with a few notable exceptions. While the original sources are still relevant, new research has revealed important distribution, life history, growth rate, age, and abundance information that is relevant to precious coral management. Some progress has also been made toward clarifying some of the vexing taxonomic challenges presented by these organisms. First, the name of the most important species of gold coral, *Gerardia* sp., has been updated to *Kulamanamana haumeaae* by Sinniger, *et al.* (2013). Second, two of the most important species in the family Coralliidae, *Corallium secundum* (pink coral) and *Corallium regale* (red coral) have been placed into separate genera, the latter also becoming a different species (Figueroa & Baco, 2014). Their new names are now *Pleurocorallium secundum* and *Hemicorallium laauense*, respectively. Third, two changes have taken place in the black corals. *Antipathes dichotoma* is now *Antipathes griggi* and *Antipathes ulex* has been moved to a different genus and is now *Myriopathes ulex* (Opresko, 2009). These changes are shown in Table 1.

1.1 General Distribution of Precious Corals

Most research related to precious corals has been limited to the Hawaiian archipelago, and the majority of the more recent efforts have been directed at taxonomy or simply documenting species distributions, with a few works on growth and life history (*Parrish et al.*, 2015). However, significant new insights have been gained into the genetics (Baco and Cairns, 2012; Sinniger, *et al.*, 2013; Figueroa and Baco, 2014), reproductive biology (Waller and Baco, 2007; Wagner, *et al.*, 2011; Wagner *et al.*, 2012; Wagner *et al.*, 2015), growth and age (Parrish and Roark 2009; Roark *et al.*, 2009; Putts, *pers. comm.*, 2017), and community structure (Kahng *et al.*, 2010; Long and Baco, 2014; Parrish, 2015; Wagner, *et al.*, 2015; Putts, *pers. comm.*, 2017) of precious coral and black coral species.

The U.S. Pacific Islands Region under jurisdiction of the Western Pacific Regional Fisheries Management Council consists of more than 50 oceanic islands, including the Hawaiian and Marianas archipelagos, American Samoa, Johnston, Wake, Palmyra, Kingman, Jarvis, Baker and Howland, and numerous seamounts in proximity to each of these groups. These islands fall under a variety of political jurisdictions, and include the State of Hawaii, the Commonwealth of the Northern Mariana Islands (CNMI), and the territories of Guam and American Samoa, as well as nine sovereign Federal territories—Midway Atoll, Johnston Atoll, Kingman Reef, Palmyra Atoll, Jarvis Island, Howland Island, Baker Island, Rose Atoll, and Wake Island. Precious corals (with currently accepted species names) are known to exist in American Samoa, Guam, Hawaii and the Northern Mariana Islands, as well as throughout the other US islands in the Pacific (Tables 1 and 2), but the only detailed assessments of precious corals have been in Hawaii (Parrish and Baco, 2007, Parrish *et al.*, 2015; Wagner, *et al.*, 2015). Over the last 10 years, we have begun to better understand the distribution and abundance of these corals, but many areas remain unexplored,

Table 1. Precious coral management unit species with updated species names

Species	Common name
Pleurocorallium secundum (prev. *Corallium secundum*)	Pink coral
Hemicorallium laauense (prev. *C. regale*)	Red coral
Kulamanamana haumeaae (prev. *Gerardia* sp.)	Gold coral
Narella sp.	Gold coral
Calyptrophora sp.	Gold coral
Callogorgia gilberti	Gold coral
Lepidisis olapa	Bamboo coral
Acanella sp.	Bamboo coral
*Antipathes griggi (*prev. *A. dichotoma)*	Black coral
Antipathes grandis	Black coral
*Myriopathes ulex (*prev. *Antipathes ulex*)	Black coral

and conditions which lead to their settlement, growth and distribution are still uncertain. Modelling efforts have provided some insight into the global distribution and habitat requirements of deep-water corals (Rogers *et al.*, 2007; Tittensor *et al.*, 2009, Clark *et al.*, 2011, Yesson *et al.*, 2012, Schlacher *et al.,* 2013), but have provided little certainty regarding localized distribution or the specific conditions required for growth of precious corals. Antipatharians, commonly known as black corals, have been exploited for years, but are still among the taxonomic groups containing precious corals that have been inadequately surveyed, as evidenced by the high rates of species discoveries from deep-water surveys around the Hawaiian Islands (Opresko 2003b; Opresko 2005a; Baco 2007; Parrish & Baco 2007; Parrish *et al.*, 2015; Roark, 2009; Wagner *et al.*, 2011, 2015; Wagner, 2011, 2013). Despite this ongoing research, only a few places are known to have dense agglomerations of precious corals. A summary of the known distribution and abundance of precious corals in the central and western Pacific Islands region follows.

American Samoa

There is little information available for the deepwater species of precious corals in American Samoa. Much of the information available comes from the personal accounts of fishermen. In the South Pacific there are no known commercial beds of pink coral (Carleton and Philipson 1987). Survey work begun in 1975 by the Committee for Co-ordination of Joint Prospecting for Mineral Resources in South Pacific Offshore Areas (CCOP/SOPAC) identified three areas of *Corallium* off Western Samoa: off eastern Upolu, off Falealupo and at Tupuola Bank (Carleton and Philipson 1987). Pink coral has been reported off Cape Taputapu, but no information concerning the quality or quantity of these corals or the depths where they occur is available. Unidentified precious corals have also been reported in the past off Fanuatapu at depths of around 90 m. Precious corals are known to occur at an uncharted seamount, about three-fourths of a mile off the northwest tip of Falealupo Bank at depths of around 300 m.

Commercial quantities of one or more species of black coral are known to exist at depths of 40 m and deeper within the territorial waters of American Samoa. Wagner (*pers. comm.*, 2015) has tentatively identified as many as 12 species (not previously catalogued in Am. Samoa) of black corals in depths between 50m and 90m, with 6 of these potential new species exhibiting growth forms that could lead to harvestable sizes. However, Wagner did not find any locations with the types of densities and sizes that would support any commercial harvest of these corals.

Guam and the Commonwealth of the Northern Marianas

There are no known commercial quantities of precious corals in the Northern Mariana Islands archipelago (Grigg and Eldredge 1975). In the past, Japanese fishermen claimed to have taken some *Corallium* north of Pagan Island and off Rota and Saipan. Preliminary results from surveys conducted throughout the Marianas Islands in 2016 indicate a scattered distribution with no areas of large agglomerations of precious corals found in waters deeper than 250 m.

U.S. Pacific Island Remote Areas

There are no known commercial quantities of precious corals in the remote Pacific Island areas, though individual colonies of precious corals have been seen at Jarvis, Palmyra, Kingman (Parrish and Baco, 2007) and Johnston Atoll, and planned surveys in 2017 may provide more information about abundance and distribution of precious corals found in waters deeper than 250 meters in these areas.

Hawaii

In the Hawaiian Archipelago there are seven legally-defined beds of pink, gold and bamboo corals, which are shown in Table 2. It is difficult to determine from the publication record exactly why these particular areas were singled out for legal recognition, other than the fact that they contain some unspecified densities of precious corals within their geographic boundaries. In the MHI, the Makapuu bed is located off Makapuu, Oahu, at depths of between 250 and 575 meters. Discovered in 1966, it the precious coral bed that has been most extensively surveyed in the Hawaiian chain. Its total area is about 4.5 km^2. Its substrate consists largely of hard limestone

(Grigg, 1988). Careful examination during numerous dives with submersibles has determined that about 20% of the total area of the Makapuu bed is comprised of irregular lenses of thin sand, sediments and barren patches (WPRFMC, 1979). These sediment deposits are found primarily in low lying areas and depressions (Grigg, 1988). Thus, the total area used for extrapolating coral density is 3.6 km^2, or 80% of 4.5 km^2 (WPRFMC, 1979).

Table 2. Location of Hawaii FEP precious coral beds

Area Name	Description
Makapu'u (Oahu)	**includes the area within a radius of 2.0 nm of a point at 21°18.0′ N. lat., 157°32.5′ W. long.**
Auau Channel, Maui	**includes the area west and south of a point at 21°10′ N. lat., 156°40′ W. long., and east of a point at 21° N. lat., 157° W. long., and west and north of a point at 20°45′ N. lat., 156°40′ W. long.**
Keahole Point, Hawaii	**includes the area within a radius of 0.5 nm of a point at 19°46.0′ N. lat., 156°06.0′ W. long.**
Kaena Point, Oahu	**includes the area within a radius of 0.5 nm of a point at 21°35.4′ N. lat., 158°22.9′ W. long.**
Brooks Banks	**includes the area within a radius of 2.0 nm of a point at 24°06.0′ N. lat., 166°48.0′ W. long.**
180 Fathom Bank, north of Kure Island	**N.W. of Kure Atoll, includes the area within a radius of 2.0 nm of a point at 28°50.2′ N. lat., 178°53.4′ W. long.**
WesPac Bed, between Nihoa and Necker Islands	**includes the area within a radius of 2.0 nm of a point at 23°18′ N. lat., 162°35′ W. long.**

Precious coral beds have also been found in the deep inter-island channels such as Auau, Alalakeiki, and Kolohi channels off of Maui, around the edges of Penguin Banks, off promontories such as Keahole Point, on older lava flows south from Keahole to Ka Lae, and off of Hilo Harbor, and off of Cape Kumukahi on the Big Island of Hawaii (Oishi, 1990; Grigg, 2001, 2002; Putts, *pers. comm.*, 2017). On Oahu, there is a bed off Kaena Point, and multiple precious coral observations have been made from offshore Barber’s Point extending to offshore

Pearl Harbor, Oahu. On Kauai, a bed of black corals has been identified offshore of Poipu (WPRFMC, 1979).

A dense bed has been located on the summit of Cross Seamount, southwest of the island of Hawaii. This bed covers a pinnacle feature on the top of the summit, but does not contain numbers of corals large enough to sustain commercial harvests (Kelley, pers. comm., 2015).

In the NWHI, a small bed of deepwater precious corals have been found on WestPac bed, between Nihoa and Necker Islands and east of French Frigate Shoals. This bed is not large enough to sustain commercial harvests. Precious coral beds have also been discovered at Brooks Banks, Pioneer Bank, Bank 8, Seamount 11, Laysan, and French Frigate Shoals (Parrish and Baco, 2007; Parrish *et al*., 2015). ROV surveys conducted throughout the NWHI by the Okeanos Explorer during 2015 discovered multiple places that had dense colonies of deep-sea corals. Few of these colonies were precious corals, but these dives were mostly conducted in waters deeper than normal distributions of precious corals (>1500 meters). However, large areas of potential habitat exist in the NWHI on seamounts and banks near 400 m depth. Based on the abundance of potential habitat, it is thought that stocks of precious corals may be more abundant in the northwestern end of the island chain. All precious coral stocks within the boundaries of the Papahānaumokuākea National Marine Monument or Coral Reef Ecosystem Reserve are reserved from harvest, and most habitat suitable for precious corals growth falls within the boundaries of the monument.

Precious corals have also been discovered at the 180 Fathom Bank, north of Kure Island. The extent of this bed is not known. Precious corals have been observed during submersible and ROV dives throughout the Northwestern Hawaiian Islands, and in EEZ waters surrounding Johnston, Jarvis, Palmyra, and Kingman atolls, but little can be definitively said about the overall distribution and abundance of precious corals in the central Pacific region.

In addition to these legally defined areas of precious corals, many other sites have been discovered that sustain populations of precious corals (Parrish and Baco, 2007; Parrish *et al*., 2015; Wagner *et al*., 2015). The map below (Figure 1) provides a color-coded illustration of some of these 8600 observations (Kelley and Drysdale, 2012, *unpublished data*). Given the number of observations and the wide distribution of precious corals in the main Hawaiian Islands, it is almost certain that undiscovered beds of precious corals exist in the EEZ waters of the region managed by the WPRFMC. Whether these beds would contain organisms at sufficient densities and size distributions to support commercial harvests is yet to be determined.

1.2 Systematics of the Deepwater Coral Species

Published records of deep corals from the Hawaiian Archipelago include more than 137 species of gorgonian octocorals and 63 species of azooxanthellate scleractinians (Parrish and Baco, 2007). A total of 6 new genera and 20 new species of octocorals, antipatharians, and zoanthids have been discovered in Hawaii since the 2007 report (Parrish *et al*., 2015). These are either new to science, or new records for the Hawaiian Archipelago (Cairns & Bayer 2008, Cairns 2009, Opresko 2009, Cairns 2010, Wagner *et al.,* 2011a, Opresko *et al.,* 2012, Sinniger *et al.,* 2013).

Taxonomic revisions currently underway for several groups of corals, e.g., isidids, coralliids, plexaurids and paragorgiids, are also likely to yield additional species new to science and new records for Hawaii (Parrish *et al.*, 2015). Only a handful of these deep coral species are considered economically *precious* and have any history of exploitation.

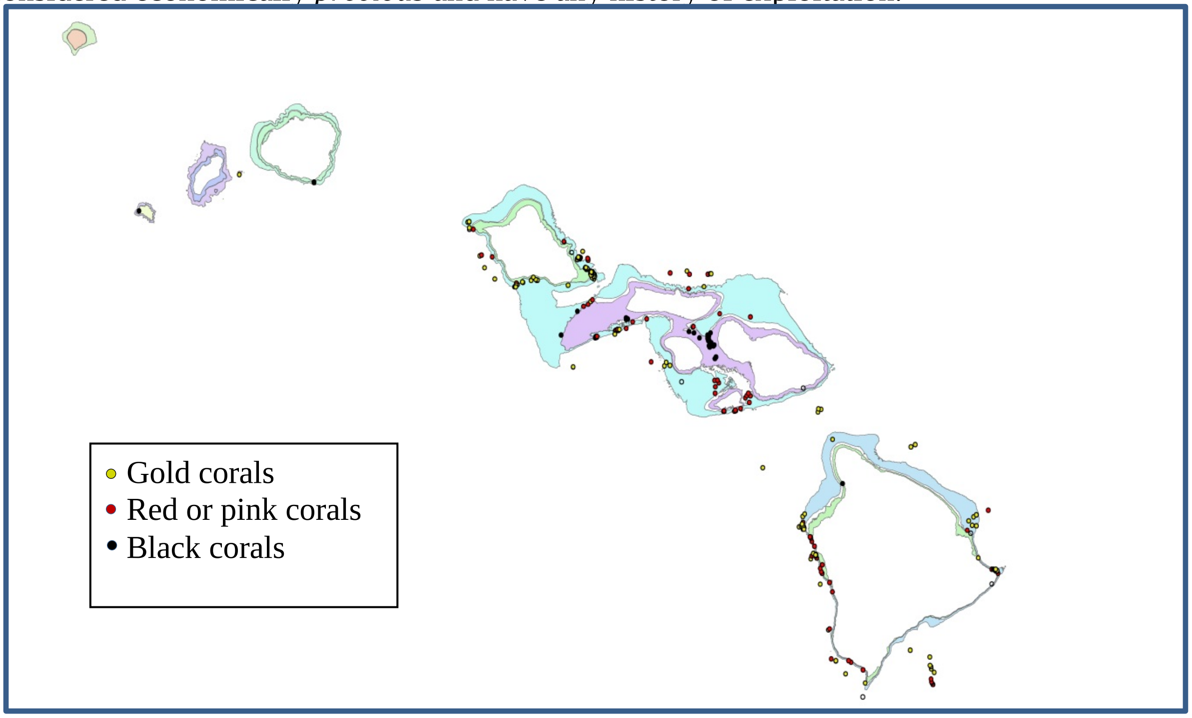

Figure 1. Observations of precious corals in the main Hawaiian islands

Recent molecular phylogenetic and morphologic studies of the family Coralliidae, including Hawaiian precious corals, have illuminated taxonomic relationships. These studies synonymized *Paracorallium* into the genus *Corallium*, and resurrected the genera *Hemicorallium* (Ardila *et al*., 2012; Figueroa & Baco, 2014; Tu *et al*., 2015) and *Pleurocorallium* (Figueroa & Baco, 2014; Tu *et al*., 2015) for several species, including several species in the precious coral trade. A molecular and morphological analysis of octocoral-associated zoanthids collected from the deep slopes in the Hawaiian Archipelago revealed the presence of at least five different genera including the gold coral (Sinniger *et al*., 2013). This study describes the five new genera and species and proposes a new genus and species for the Hawaiian gold coral, *Kulamanamana haumeaae*, an historically important species harvested for the jewelry trade and the only Hawaiian zoanthid that appears to create its own skeleton.

Precious corals are found principally in three orders of the class Anthozoa: Gorgonacea, Antipatharia, and Zoanthiae (Grigg, 1984). In the western Pacific region, pink coral (*Pleurocorallium secundum*), red coral (*Hemicorallium laauense*), gold coral (*Kulamanamana haumeaae*), black coral (*Antipathes* sp.) and bamboo coral (*Lepidisis olapa*) are the primary species/genera of commercial importance. Of these, the most valuable precious corals are species of the genera *Pleurorallium* and *Hemicorallium,* the pink and red corals (Grigg, 1984). Pink coral (*P. secundum*) and Midway deep-sea coral (*Corallium* sp. nov,) are two of the principal species of commercial importance in the Hawaiian and Emperor Seamount chain (Grigg, 1984). *P. secundum* is found in the Hawaiian archipelago from Milwaukee Banks in the Emperor

Seamounts (36^{o}N) to the Island of Hawaii (18^{o}N); *Corallium* sp. nov. is found between 28^{o}–36^{o}N, from Midway to the Emperor Seamounts (Grigg, 1984). In addition to the pink corals, the bamboo corals, *Lepidistis olapa* and *Acanella* sp., are commercially important precious corals in the western Pacific region (Grigg, 1984). Pink coral and bamboo coral are found in the order Gorgonacea in the subclass Octocorallia of the class Anthozoa, in the Phylum Coelenterata (Grigg, 1984).

The final two major groups of commercially important precious corals, gold coral and black coral, are found in separate orders, Zoanthidea and Antipatharia, in the subclass Hexacorallia, in the class Anthozoa and the phylum Coelenterata. The gold coral, *Kulamanamana haumeaae* (prev. *Gerardia* sp.) (Sinneger, *et.al.*, 2013), is endemic to the Hawaiian and Emperor Seamount chain (Grigg 1984). It inhabits depths ranging from 300–400 m (Grigg 1974, 1984). In Hawaii, gold coral, *Kulamanamana haumeaae,* grows mostly on bamboo hosts (e.g. *Acanella, Keratoisis*) as a parasitic overgrowth (Brown, 1976; Grigg, 1984; Parrish, 2015). Gold coral is, therefore, only found growing in areas that were previously inhabited by colonies of *Acanella* (Grigg, 1993) and possibly other bamboo corals (Parrish, 2015). Despite its ecological significance and long history of exploitation, the Hawaiian gold coral has never been subject to taxonomic studies or a formal species description. As a result of this, the nomenclature concerning the Hawaiian gold coral has been relatively confused. Symptomatic of the order, a suite of other zoanthids, besides the Hawaiian gold coral, have been observed and collected in Hawaii, but far less is known of their biology and ecology and they have not been described taxonomically (Sinnegar *et al.,* 2013).

Grigg (1984) classified black corals in the order *Antipatharia*, and identified fourteen genera of black corals reported from the Hawaii-Pacific region with species found in both shallow and deep habitats Grigg, 1965). Wagner (2015) noted that there are over 235 known species of black coral that occur in the oceans of the world, and of this total, only about 10 species are of commercial importance (Grigg, 1984). Wagner (2011) confirmed 8 species of black corals in Hawaii, including (1) *Antipathes griggi* Opresko, 2009, (2) *Antipathes grandis* Verrill, 1928, (3) *Stichopathes echinulata* Brook, 1889, (4) an undescribed *Stichopathes* sp., (5) *Cirrhipathes* cf. *anguina* Dana, 1846, (6) *Aphanipathes verticillata* Brook, 1889, (7) *Acanthopathes undulata* (Van Pesch, 1914), and (8) *Myriopathes* cf. *ulex* Ellis & Solander, 1786. A new name for the Hawaiian species of antipatharian coral previously identified as *Antipathes dichotoma* (Grigg and Opresko, 1977) is described as *Antipathes griggi* (Opresko, 2009).

Many species of gorgonian corals are known to occur within the habitat of pink, gold and bamboo corals in the Hawaiian Islands. At least 37 species of precious corals in the order Gorgonacea have been identified from the Makapuu bed (Grigg and Bayer, 1976). In addition, 18 species of black coral (order Antipatharia) have been reported to occur in Hawaiian waters (Grigg and Opresko, 1977; Oishi, 1990; Wagner, 2011.), but only 3 of these species have been subject to commercial harvest (Oishi, 1990; Wagner *et al.,* 2015).

1.3 Biology and Life History

The management and conservation of deep-sea coral communities is challenged by international harvest with non-selective gear types for the jewelry trade and the paucity of information to inform management strategies. In light of their unusual vulnerability, a better understanding of deep-sea coral ecology and their interrelationships with associated benthic communities is needed to inform coherent international conservation strategies for these important deep-sea habitat-forming species (Bruckner, 2013). Millennia are probably required for a precious coral community to form with full diversity, high evenness, and mature size structure (Putts, *pers. comm*., 2017). Most of the interior of the global ocean remains unobserved. This leaves questions of trophic connectivity, longevity, and population dynamics of many deep-sea communities unanswered. Deep-sea megafauna provide a complex, rich, and varied habitat that promotes high biodiversity and provides congregation points for juvenile and adult fish (Freiwald *et al.*, 2004; Husebo *et al.*, 2002; Smith *et al.*, 2008).

Precious corals may be divided primarily into two groups of species based on their depth ranges: the deepwater species (200-600m) and the shallow water species (20-120m). Other precious corals can be found in depths down to 2000 m, but these species are not exploited in the United States for commercial purposes. Deep-sea corals are found on hard substrates on seamounts and continental margins worldwide at depths of 300 to 3,000 m.

Deep Corals

The Pacific Islands deepwater precious coral species include pink coral, *Pleurocorallium secundum* (prev. *Corallium secundum*), red coral, *Hemicorallium laauense* (prev. *C. regale or C. laauense*), gold coral, *Kulamanamana haumeaae (*prev. *Gerardia sp*.) and bamboo coral, *Lepidistis olapa*. As previously discussed, the most valuable precious corals are gorgonian octocorals (Grigg, 1984). There are seven varieties of pink and red precious corals in the western Pacific region, six of which used to be recognized as distinct species of *Corallium* (Grigg, 1981), but have been reclassified (Parrish *et al.*, 2015). The two species of commercial importance in the EEZ around the Hawaiian Islands are the pink coral *Pleurocorallium secundum* (prev. *Corallium secundum*), and the red coral, *Hemicorallium laauense* (prev. *C. laauense*). The Gorgonian octocorals are by far the most abundant and diverse corals in the Hawaiian Archipelago. Two species, *Pleurocorallium secundum* and *Hemicorallium laauense* are known to occur at depths of 300-600 m on islands and seamounts throughout the Hawaiian Archipelago (Grigg 1974, 1993; Parrish *et al.*, 2015; Parrish and Baco, 2007). Parrish (2007) surveyed *Pleurocorallium secundum* and *Hemicorallium laauense* at 6 precious coral beds in the lower Hawaiian chain, from Brooks Bank to Keahole Point, Hawaii, in depths ranging from 350m to 500m. He found corals on summits, flanks, and shallow banks, with bottom substrate and relief at these sites ranging from a homogenous continuum of one type to a combination of many types at a single site. The survey results show that all three coral taxa colonize both carbonate and basalt/manganese substrates, and the corals favor areas where bottom relief enhances or modifies flow characteristics that may improve the colony’s feeding success.

These corals can grow to more than 30 cm in height, and are often found in large beds with other octocorals, zoanthids, and sometimes scleractinians (Parrish *et al.*, 2015; Parrish and Baco, 2007). These species are relatively long lived, with some of the oldest colonies observed within Makapuu Bed about 0.7 m in height and at least 80 years old (Grigg, 1988b, Roark, 2006). Populations of *P. secundum* appear to be recruitment limited, although in favorable environments (e.g., Makapuu Bed) populations are relatively stable, suggesting that recruitment and mortality are in a steady state (Grigg, 1993). During surveys of lava flows off the western flanks of Hawaii

Island, Putts (*pers. comm*., 2017) found that Coralliidae dominated the early successional stages, and using dates established for those flows, determined that a mature Corallidae community can be established within 150 years. A study by Roark *et al.* (2006) showed that the radial growth rate for specimens of *P. secundum* in the Hawaiian Islands is ~170 µm yr^{-1} and average age is 67 to 71 years, older than previously calculated. Individual colonies have been measured as tall as 28 cm. Bruckner (2009) suggested that the minimum allowable size for genus *Corallium* for harvest should be increased, and supported a potential listing for *Corallium* within the Appendices of the Convention on International Trade in Endangered Species (CITES). The current size restriction in the 2010 Code of Federal Regulations for Pacific Islands Region is 10 in (25.4 cm).

In Cairn's reviews (2008; 2009; 2010), he summarized the research conducted on Hawaiian Octocorallia taxa, including three gold coral PCMUS genuses, *Narella*, *Calyptrophora* and *Callogorgia*. Octocorallia are distributed over all ocean basins, found in depths ranging from shallow (~ 50m) to deep (~ 4,600) in Alaska. All gold PCMUS in Hawaii were collected in deep water (> 270m), throughout the Hawaiian archipelago and adjacent seamounts. Although these octocorals are managed as PCMUS, the only commercially exploited gold coral is the zoantharian, *Kulamanamana haumeaae* (prev. *Gerardia* sp.). It is probably the most common and largest of the zoanthids in Hawaii, and is widely distributed throughout the Hawaiian Archipelago and into the Emperor Seamount Chain at depths of 350–600 meters (Parrish *et al.*, 2015; Parrish and Baco, 2007). While subject to commercial exploitation from the 1970's until 2001 with an interruption between 1979 and 1999 (Grigg, 2001), the gold coral is not currently exploited in Hawaii due to a moratorium on the fishery. The Hawaiian gold coral is one of the largest and numerically dominant benthic macro-invertebrates in its depth range on hard substrate habitats of the Hawaiian Archipelago, and plays an important ecological role in Hawaiian seamount benthic assemblage (Parrish, 2006; Parrish and Baco, 2007; Parrish, *et al.*, 2015). The Hawaiian gold coral has also been found to be one of the longest-lived species on earth. Earlier ageing attempts on the gold coral focused on ring counts (Grigg, 1974; Grigg, 2002) and led to a maximal estimated age of 70 years and a radial growth rate (increase in branch diameter) of 1 mm/year. Recent studies using radiometric data suggest colonies of Hawaiian gold coral are as old as 2740 year with a radial growth rate of only 15 to 45 µm/year (Roark *et al.*, 2006; Roark *et.al.*, 2009; Parrish and Roark, 2009).

Parrish (2015) has found the host of the parasitic *Kulamanamana haumeaae* to be primarily the bamboo corals (e.g. Acanella, Keratoisis). *K. haumeaae* secretes a protein skeleton that over millennia can grow and more than double the original mean size of the host colony. It is relatively common and even dominant at geologically older sample sites, but recruitment is probably infrequent (Parrish, 2015). Although it can be relatively common compared to some other deep corals, it grows very slowly. Parrish and Roark (2009) determined that the Hawaiian gold coral *Kulamanamana haumeaae* has a mean life span of 950 yrs with an overall radial growth of ~41 µm yr^{-1}, and a gross radiocarbon linear growth rate of 2.2 ± 0.2 mm yr^{-1}. This is a much slower growth rate and longer life span than given in previous studies. Grigg (2002) reported a 1 mm yr^{-1} radial growth rate, equivalent to a 6.6 cm yr^{-1} linear growth for a maximum life span of roughly 70 yrs. This means these corals are growing much slower than previously thought, and have much longer life spans if undisturbed. Newly applied radiocarbon age dates from the deep water proteinaceous corals *Gerardia* and *Leiopathes* show that radial growth rates are as low as 4 to 35 micometers per year and that individual colony longevities are on the order of thousands of years (Roark *et al.*, 2009, 2006). The longest-lived *Gerardia* sp. and *Leiopathes* specimens were estimated to be 2,742 years old and 4,265 years old, respectively. *Gerardia* sp. is a colonial zoanthid with a hard skeleton of hard proteinaceous matter that forms tree-like structures with heights of several meters and basal diameters up to 10s of a centimeter. Black corals of *Leiopathes sp*. also has a hard proteinaceous skeleton and grows to heights in excess of 2 m. In Hawai'ian waters, these corals are found at depths of 300 to 500 m on hard substrates, such as seamounts and ledges.

The two bamboo coral PCMUS in the Pacific Islands Region are classified under two genera, *Acanella* and *Lepidistis*. Not much work has been done specifically on these genera, but Parrish (2015) identified branched bamboo colonies such as *Acanella* as a preferred host for *Kulamanamana haumeaae*. Because of the long colony life span of >3000 yrs and the bony hard bodied calcareous internodes of bamboo corals (family Isididae), geochemists are interested in using them to analyze paleo-oceanographic events and long-term climate change (Hill *et al*. 2011), while biologists use them to size and age deep-sea coral populations. Recent studies show that the subfamily Keratoisidinae (family Isididae) consists of four genera (*Acanella*, *Isidella*, *Lepidistis*, and *Keratoisis*), with two genera (*Tenuisis* and *Australisis*) perhaps belonging elsewhere in the Isididae family (Etnoyer 2008; France 2007). Bamboo corals commonly colonize intermediate to deep water depths (400m to >3000m) of continental slopes and seamounts in the Pacific Ocean.

Shallow Corals

The second group of precious coral species is found in shallow water between 20 and 120 m (Grigg, 1993 and Drysdale, *unpublished data*, 2012; Wagner *et al.*, 2015). The shallow water fishery is comprised of three species of black coral, *Antipathes griggi*, *A. grandis* and *Myriopathes ulex*, which have historically been harvested in Hawaii (Oishi 1990), but over 90% of the coral harvested by the fishery consists of *A. griggi* (Oishi 1990; Parrish *et al.*, 2015; Wagner *et al.*, 2015). Other black coral species are found in the NWHI in a wider depth range (20m to 1,400m), but with lower colony density (Wagner *et al*., 2011). Surveys performed in depths of 40-110 meters in the Au'au Channel in 1975 and 1998, suggested stability in both recruitment and growth of commercially valuable black coral populations, and thus indicated that the fishery had been sustainable over this time period (Grigg, 2001). Subsequent surveys performed in the channel in 2001 indicated a substantial decline in the abundance of black coral colonies, with likely causes including increases in harvesting pressure and overgrowth of black coral colonies by the invasive octocoral *Carijoa sp.* and the red alga, *Acanthophora spicifera*, especially on reproductively mature colonies at mesophotic depths (Grigg 2003; Grigg 2004; Kahng & Grigg 2005; Kahng, 2006). Together, these factors renewed scrutiny on the black coral fishery and raised questions about whether regulations need to be redefined in order to maintain a sustainable harvest (Grigg, 2004). In addition to these challenges, Wagner has suggested that taxonomic misidentification has led to the mistaken belief that there is a depth refuge that exists for certain harvested species (Wagner *et al.,* 2012; Wagner, 2011). All of these uncertainties and lack of basic life history information regarding black corals complicates effective management of the resource (Grigg, 2004).

In Hawaii, *A. griggi* accounts for around 90% of the commercial harvest of black coral (Oishi

1990). *A. grandis* accounts for 9% and *M. ulex* 1% of the total black corals harvested. In Hawaii, roughly 85% of all black coral harvested are taken from within state waters. Black corals are managed jointly by the State of Hawaii and the Council. Within state waters (0–3 nmi), black corals are managed by the State of Hawaii (Grigg, 1993).

A new name for the Hawaiian species of antipatharian coral previously identified as *Antipathes dichotoma* (Grigg and Opresko, 1977) is described as *Antipathes griggi* Opresko, n. sp. (Opresko, 2009). The shallow water black coral *A. dichotoma* (*A. griggi)* collected at 50 m exhibited growth rates of 6.42 cm yr^{-1} over a 3.5 yrs study.

Table 3: Depth zonation of precious corals in the Western Pacific. (Source: Grigg 1993, Baco-Taylor, 2007, HURL and Drysdale, 2012)

Species and Common Name	Depth Range (m)
Paracorallium secundum Angle skin coral	250–575
Hemicorallium laauense Red coral	250–575
Corallium sp nov. Midway deepsea coral	1,000–1,500
Kulamanamana haumeaae (prev. *Gerardia* sp.) Hawaiian gold coral	350–575
Lepidisis olapa, *Acanella* spp. bamboo coral	250–1800
Antipathes griggi (prev. *A. dichotoma*), black coral	20–120
Antipathes grandis, pine black coral	20–120
Cirrhipathes cf. *anguina* (prev. *Antipathes anguina*), wire black coral	20–120
Myriopathes ulex (prev. *Antipathes ulex*), fern black coral	20–220

1.4 Growth and Reproduction

There is very limited published literature regarding coral spawning of the PCMUS in the Pacific Islands Region. However, studies by Gleason, *et al*. (2006) and Waller and Baco (2007) indicate that the gold coral *Kulamanamana haumaae* may have seasonal reproduction, and that two pink coral species have a periodic or quasi-continuous reproductive periodicity. Although limited studies about growth rates and life spans of adult PCMUS in the Pacific Islands Region are available, early life history data on larvae, polyps, and juvenile colonies of the PCMUS are unavailable. Many other questions related to genetic connectivity and spatial distribution across the Pacific also remain unanswered. Recent mesophotic coral reef ecosystem studies provide an outline of essential knowledge for the limited deep water coral ecosystem (Kahng, *et al*. 2010). Slow-growing deep-water coral ecosystems are sensitive to many disturbances, such as temperature change, invasive species and destructive fishing techniques.

While different species of precious corals inhabit distinct depth zones, their habitat requirements are strikingly similar. Grigg (1984) noted that these corals are non-reef building and inhabit depth zones below the euphotic zone. In an earlier study, Grigg (1974) determined that precious corals are found in deep water on solid substrate in areas that are swept relatively clean by moderate to strong bottom currents (>25 cm/sec). Strong currents help prevent the accumulation of sediments, which would smother young coral colonies and prevent settlement of new larvae. Grigg (1984) notes that, in Hawaii, large stands of *Corralium* are only found in areas where sediments almost never accumulate, and *P. secundum* appears in large numbers in areas of high flow over carbonate pavement (Parrish *et al.,* 2015; Parrish and Baco, 2007). *Hemicorallium laauense* grows in an intermediate relief of outcrops; and *Kulamanamana haumaae* is most commonly seen growing in high relief areas on pinnacles, walls, and cliffs. These habitat differences may reflect preferred flow regimes for the different corals (e.g., laminar flow for *P. secundum*, alternating flow for *Kulamanamana haumaae*) (Parrish *et al.,* 2015).

Surveys of all potential sites for precious corals in the MHI conducted using a manned submersible show that most shelf areas in the MHI near 400 m are periodically covered with a thin layer of silt and sand (Grigg, 1984). Precious corals are known to grow on a variety of bottom substrate types. Precious coral yields, however, tend to be higher in areas of shell sandstone, limestone and basaltic or metamorphic rock with a limestone veneer. Grigg (1988) concludes that the concurrence of oceanographic features (strong currents, hard substrate, low sediments) necessary to create suitable precious coral habitat are rare in the MHI. Depth clearly influences the distribution of different coral taxa and certainly there is patchiness associated with the presence of premium substrate and environmental conditions (flow, particulate load, etc.). The environmental suitability for colonization and growth is likely to differ among coral taxa.

The habitat sustaining precious corals is generally in pristine condition. There are no known areas that have sustained damage due to resource exploitation, notwithstanding the alleged heavy foreign fishing for corals in the Hancock Seamounts area. Although unlikely, if future development projects are planned in the proximity of precious coral beds, care should be taken to prevent damage to the beds. Projects of particular concern would be those that suspend sediments or modify water-movement patterns, such as deep-sea mining or energy-related operations.

There has been very little research conducted concerning the food habits of precious corals. Precious corals are filter feeders (Grigg, 1984; 1993). The sparse research available suggests that particulate organic matter and microzooplankton are important in the diets of pink and bamboo coral (Grigg, 1970). Many species of pink coral, gold coral (*Kulamanamana haumeaae (*prev. *Gerardia* sp.*)* and black coral (*Antipathes*) form fan shaped colonies (Grigg, 1984; 1993). This type of morphological adaption maximizes the total area of water that is filtered by the polyps (Grigg, 1984; 1993). Bamboo coral (*Lepidisis olapa*), unlike other species of precious corals, is unbranched (Grigg, 1984). Long coils that trail in the prevailing currents maximize the total amount of seawater that is filtered by the polyps (Grigg, 1984). While clearly, the presence of strong currents is a vital factor determining habitat suitability for precious coral colonies, their role to date is not fully understood.

Light is one of the most important determining factors of the upper depth limit of many species of precious corals (Grigg, 1984).The larvae of two species of black coral, *Antipathes grandis* and *A. griggi,* are negatively phototaxic**.**

Grigg (1984) states that temperature does not appear to be a significant factor in delimiting suitable habitat for precious corals. In the Pacific Ocean, species of *Corallium* are found in temperature ranges of 8^o to 20^oC, he observes. Temperature may determine the lower depth limits of some species of precious coral, including two species of black corals in the MHI. In the MHI, the lower depth range of two species of black corals (*A.griggi* and *A. grandis*) coincides with the top of the thermocline (about 100 m). Although, *A. griggi* can be found to depths of 100 m, it is rare below the 75 m depth limit at which commercial harvest occurs in Hawaiʻi. Thus, the supposed depth refuge from harvest does not really exist, and was probably based on taxonomic misidentification, thereby calling into question population models used for the management of the Hawaiian black coral fishery (Wagner *et al.,* 2012; Wagner, 2011).

In pink coral (*P. secundum*), the sexes are separate (Grigg, 1993). Based on the best available data, it is believed that *P. secundum* becomes sexually mature at a height of approximately 12 cm (13 years) (Grigg, 1976). Pink coral reproduce annually, with spawning occurring during the summer, during the months of June and July. Coral polyps produce eggs and sperm. Fertilization of the oocytes is completed externally in the water column (Grigg, 1976; 1993). The resulting larvae, called planulae, drift with the prevailing currents until finding a suitable site for settlement.

Pink, bamboo and gold corals all have planktonic larval stages and sessile adult stages. Larvae settle on solid substrate where they form colonial branching colonies. Grigg (1993) notes that the lengths of the larval stage of all deepwater species of precious corals is unknown. Clean swept areas exposed to strong currents provide important sites for settlement of the larvae, Grigg adds. The larvae of several species of black coral (*Antipathes*) are negatively photoactic, he notes. They are most abundant in dimly lit areas, such as beneath overhangs in waters deeper than 30 m. In an earlier study, Grigg (1976) found that “within their depth ranges, both species are highly aggregated and are most frequently found under vertical dropoffs. Such features are commonly associated with terraces and undercut notches relict of ancient sea level still stands. Such features are common off Kauai and Maui in the MHI. Both species are particularly abundant off of Maui and Kauai, suggesting that their abundance is related to suitable habitat.” Off of Oahu, many submarine terraces that otherwise would be suitable habitat for black corals are covered with sediments (Grigg, 1976).

A variety of invertebrates and fish are known to utilize the same habitat as precious corals. These species of fish include onaga (*Etelis coruscans*), kahala (*Seriola dumerili*) and deepwater shrimp (*Heterocarpus ensifer*). These species do not seem to depend on the coral for shelter or food.

Densities of pink, gold and bamboo coral have been estimated for an unexploited section of the Makapuu bed (Grigg, 1976). As noted in the FMP for precious corals, the average density of pink coral in the Makapuu bed is 0.022 colonies/m^2. This figure was extrapolated to the entire bed (3.6 million m^2), giving an estimated standing crop of 79,200 colonies. At the 95% confidence limit, the standing crop is 47,500 to 111,700 colonies. The standing crop of colonies was converted to biomass ($3N_iW_i$), resulting in an estimate of 43,500 kg of pink coral in the Makapuu bed.

In addition to coral densities, Grigg (1976) determined the age-frequency distribution of pink coral colonies in Makapuu bed. He applied annual growth rates to the size frequency to calculate the age structure of pink coral at Makapuu Bed (Table 4). More recent work by Roark *et al.* (2006) suggests that annual growth ring dating may underestimate the ages of many species of deep water corals, and that most of the colonies that have been dated using the ring method are probably older and slower growing than first estimated.

Estimates of density were also made for bamboo (*Lepidisis olapa*) and gold coral (*Kulamanamana haumeaae (*prev. *Gerardia* sp.*)* for Makapuu bed. The distributions of both these species are patchy. As noted in the FMP, the area where they occur comprises only half of that occupied by pink coral (1.8 km^2). Estimates of the unexploited abundance of bamboo and gold coral were 18,000 and 5,400 colonies, respectively. Estimates of density for the unexploited bamboo coral and gold coral in the Makapuu bed are 0.01 colonies/m^2 and 0.003 colonies/m^2. Using a rough estimate for the mean weights of gold and bamboo coral colonies (2.2 kg and 0.6 kg), a standing crop of about 11,880 kg of gold coral and 10,800 kg for bamboo for Makapuu bed was obtained.

Growth rates for several species of precious corals found in the western Pacific region have been estimated. Grigg (1976) stated that the height of pink coral (*P. secundum*) colonies increases about 0.9 cm/yr up to about 30 years of age. These growth rates are probably overestimated, and should be revisited using modern methodologies, such as radiometric dating (Roark *et al.*, 2006). As noted in the FMP for precious corals, the height of the largest colonies of *Pleurocorallium secundum* at Makapuu bed rarely exceed 60 cm. Colonies of gold coral are known to grow up to 250 cm tall while bamboo corals may reach 300 cm. The natural mortality rate of pink coral at Makapuu bed is believed to be 0.066, equivalent to an annual survival rate of about 93%.

Table 4: Age-Frequency Distribution of *Pleurocorallium secundum* (Source: Grigg, 1973)

Age Group (years)	Number of Colonies
0–10	44
10–20	73
0–30	22
30–40	12
40–50	7
50–60	0

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Non-fishing effects that may adversely affect essential fish habitat in the Pacific Islands region

FINAL REPORT

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List of Acronyms

AS	American Samoa
ATON	Aids to Navigation
BMP	Best management practice
CCA	Crustose coralline algae
CLB	Continuous-line bucket system
CNMI	Commonwealth of the Northern Mariana Islands
DSHMRA	Deep Seabed Hard Mineral Resources Act
EEZ	Exclusive Economic Zone
EFH	Essential Fish Habitat
ENSO	El Niño-Southern Oscillation
EPAP	Ecosystem Principles Advisory Panel
FEP	Fishery Ecosystem Plan
FMP	Fishery Management Plan
HI	State of Hawaiʻi
ISA	International Seabed Authority
MCE	Mesophotic coral ecosystems
MSA	Magnuson-Stevens Fishery Conservation and Management Act
MUS	Management Unit Species
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
OTEC	Ocean Thermal Energy Conversion
PAR	Photosynthetically Active Radiation
PCB	Polychlorinated biphenyls
PDO	Pacific Decadal Oscillation
POM	Particulate organic matter
PPM	Parts per million
PRIA	U.S. Pacific Remote Island Areas
REE	Rare earth elements
TBT	Tri-butyl tin
UV	Ultraviolet radiation
UXO	Unexploded ordnance
WPWP	Western Pacific Warm Pool
WPRFMC	Western Pacific Regional Fishery Management Council

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Executive Summary

Originally enacted in 1976, the Magnuson-Stevens Fishery Conservation and Management Act (MSA) established a federal system to conserve fishery resources and promote a sustainable commercial and recreational fishing industry within the United States of America. To achieve this in the Western Pacific Region, the Western Pacific Regional Fishery Management Council (WPRFMC) was tasked with making management recommendations to the National Marine Fisheries Service for review and implementation through the regulatory process. Recognizing that both the loss and degradation of important habitat were significant, long-term threats to fisheries, the MSA required Essential Fish Habitat (EFH) be described and identified, that adverse effects on EFH be minimized to the extent practicable, and that actions be implemented to encourage habitat conservation and enhancement.

The MSA requires fishery management plans (FMPs) to identify non-fishing activities that may adversely affect EFH, and to provide conservation and enhancement measures that avoid, minimize, mitigate, or otherwise offset adverse effects for federal activities. The WPRFMC uses fishery ecosystem plans (FEPs) to meet the requirements of FMPs under the MSA. A review of information available on EFH must be completed at least once every five years, and EFH provisions of FMPs must be revised or amended, as warranted.

This report reviews the potential effects (including potential cumulative effects) resulting from a range of non-fishing activities and other potential sources of stress. The purpose of this review is to gather new information on: 1) non-fishing activities that may adversely affect EFH, 2) known and potential adverse effects of these activities on EFH, and 3) options to avoid, minimize, or offset those adverse effects. This information will assist the WPRFMC in determining whether modifications to the existing non-fishing effects sections of the five Western Pacific Region FEPs are warranted.

Due to a lack of specific habitat information for many of the management unit species (MUS), the WPRFMC has broadly defined EFH to include nearly all waters and benthos within the Exclusive Economic Zone (EEZ) and encompass all marine and estuarine ecosystems within the marine waters of the Western Pacific jurisdictions. In this report, effects to EFH are evaluated from the context of individual ecosystem function within a designated EFH because identified EFHs are often comprised of multiple marine and estuarine ecosystems. Additionally, most ecological studies assessing the effects of non-fishing activities are conducted at the organismal and ecosystem scales, and each ecosystem may display a different response to a given activity.

Consistent with the ecosystems included in the Western Pacific Region FEPs, this report examines the effect of non-fishing-related activities on eight marine ecosystems: (1) intertidal, (2) mangrove forests or mangals, (3) seagrasses, (4) coral reefs, (5) deep reef slopes, (6) banks and seamounts, (7) deep-ocean floor, and (8) pelagic.

The implementing regulations of the Sustainable Fisheries Act, which amended the MSA in 1996, focused on a diverse array of human activities that could adversely affect EFH, but failed to distinguish between human actions and ecological processes/stressors that can cause ecosystem change in a meaningful way. This report attempts to clearly delineate human activities and sources of stress from the stressors themselves. Doing so allows for a clearer understanding of potential effects of an activity because different activities often alter the intensity, duration, frequency, timing, and/or scale of the same stressor, which results in similar effects on an ecosystem regardless of the original activity (*e.g.*, reduced light affects seagrass growth in the same way regardless of whether the reduction in light results from a dredging project or a permanent structure). Nine categories of non-fishing activities are identified: (1) climate change, (2) energy production, (3) mining, (4) land-based aquaculture, (5) development/ construction, (6) shipping, (7) marine debris, (8) non-fishing human uses, and (9) wastewater discharge.

EFH is subjected to a range of non-fishing human activities and other sources of stress. These activities can affect EFH by altering the magnitude and direction of potential ecological stressors, which in turn may either: a) directly affect organisms and/or the biological processes that control their population dynamics, or b) indirectly affect organisms by altering interspecies interactions or by affecting the quality or quantity of their environment.

Ecological stressors are factors that alter the productivity, fitness, and the survival of organisms, and/or affect the long-term persistence and the functional and structural capacity of populations, biological assemblages, or ecosystems. Sources of ecological stress can come from natural environmental events (*e.g.*, storms), or may result directly or indirectly from human activities. Some ecological stressors act at a relatively small spatial scale, whereas others are regional or global in effect.

When exposure to environmental stressors changes in intensity, duration, frequency, timing, and/or scale, organisms and/or ecosystems will undergo an ecological response. Species and ecosystems have some inherent capacity to tolerate changes in the exposure to stressors, but there are limits to this ability, which are often represented as tolerance thresholds. When these thresholds are exceeded, substantial ecological change may occur.

Fifteen potential stressors on EFH have been identified for this report, and their effects on the ecosystems within the Western Pacific Region are discussed in detail. These stressors (in bold) have been grouped into the following broad categories:

1. *Environmental stressors* are associated with excessive or insufficient physical or chemical conditions within the marine environment, and in this report, include: **Ocean acidification, Shifts in productivity, Thermal, Salinity, Irradiance, Noise,** and **Hypoxia.**

2. *Biological stressors* are associated with interactions among organisms of the same or different species, and in this report, include: **Invasive species, Disease,** and **Fish aggregating device (FAD) effect.**

3. *Physical stressors* are associated with changes in exposure to kinetic energy, and in this report, include: **Physical damage.**

4. *Pollution stressors* occur when chemicals or other contaminants are present in concentrations large enough to affect organisms and thereby cause ecological change, and in this report, include: **Sediment, Chemicals,** and **Nutrient inputs.**

5. ***Sea level rise*** is a unique marine stressor with important implications in the Western Pacific Region. On casual examination, sea level rise alone might appear to be unimportant to subtidal marine ecosystems, but it is a substantial direct threat to intertidal and mangrove ecosystems, and acts indirectly on certain other ecosystems through often synergistic interactions with other stressors.

In any circumstance—meaning at a particular time and place—organisms are exposed to a complex regime of interacting ecological stressors. In some instances, the exposure to a given stressor is intense, but of short duration (*e.g.*, a storm-driven flood event). In other instances, exposure may be chronic and relatively unchanging over time (*e.g.*, sewage discharge). The complex interactions among stressors, and across their ranges of exposure, are what determine the potential effects on organisms and ecosystems.

The effects of these stressors on EFH will vary broadly by ecosystem type, the organisms affected, and their location, and are discussed in detail in the report. In some cases, little-to-no effect may be observed (*e.g.*, changes in irradiance levels will likely have minor, if any, effects on deep ocean floor ecosystems). However, the effects of other stressors on EFH can be significant, resulting in increased mortality, altered abundances and assemblage composition, and disrupted trophic dynamics. Sub-lethal effects would result in reduced individual fitness, affecting calcification, photosynthesis, growth and metabolism, gene expression, behavior, and interspecific interactions. In many cases, adverse effects will be most pronounced on microscopic organisms and planktonic life history stages of macro-fauna, leading to reproductive failure and shifts in primary productivity leading to significant, and likely adverse, effects cascading through food webs.

Cumulative effects are impacts on the environment that result from the incremental effect of an action when added to other past, present, and reasonably foreseeable future actions, regardless of who undertakes such actions. Cumulative effects can result from individually minor, but collectively significant actions taking place over a period of time, or from the cumulative and interactive effects of multiple actions. The cumulative effect from two or more actions is the result of additive (no interaction), synergistic (increased adverse effect), or antagonistic (decreased adverse effect) interactions.

Crain *et al.* (2008) reviewed over 200 studies examining cumulative effects for multiple stressors in intertidal and nearshore marine ecosystems to elucidate general patterns in cumulative stressor effects. In 62% of all cases, interactions between two stressors resulted in an adverse effect on the species or ecosystem that was at least additive (26%) or synergistic (36%). In cases where a third stressor was considered, over two-thirds of the interaction became more negative, and the number of synergistic interactions increased to 66% of the three-stressor cases. Thus, any activity or set of activities that significantly increases the negative effects of three or more stressors is likely to result in synergistic interactions that increase the likelihood of adverse effects on EFH.

The WPRFMC is tasked with describing ways to avoid, minimize, mitigate, or otherwise offset adverse effects of non-fishing activities to EFH, and for promoting the conservation and enhancement of EFH. Best management practices (BMPs), due to their generalized applicability, are the focus of this report.

To be effective, a BMP must: (1) provide meaningful and measureable minimization of impacts, (2) be properly selected and implemented, (3) be regularly inspected to insure its integrity, and (4) be monitored to assess effectiveness. Failure to meet all four requirements may result in a BMP that is ineffective for its intended purpose.

BMPs that can reduce the potential adverse effects of non-fishing activities on EFH are identified from the scientific literature, recommendations made by federal and state/territorial/ commonwealth agencies, and environmental review documents such as environmental impact statements. BMPs have been recommended for specific activity categories and stressor types. The BMPs recommended by activity category generally contain recommendations on the design, placement and execution of activities with the intention of avoiding and minimizing potential adverse effects on EFH at the development and implementation stage of an activity. The BMPs recommended by stressor type contain recommendations intended to reduce the effect of a specific stressor on EFH, either through reduction of the activities' effect on the stressor or by reducing the effect of the stressor on the ecosystem. As such, these BMPs tend to address temporary issues (*e.g.*, construction-related runoff). The BMPs by stressor are not necessarily specific recommendations for a single category of non-fishing activity, and often can be broadly applied across a range of activities. The resulting list of BMPs is not exhaustive, but represents commonly-employed, proven approaches as well as some common-sense recommendations to reduce adverse environmental effects.

1.0 Background

1.1 Magnuson-Stevens Fishery Conservation and Management Act

The Magnuson-Stevens Fishery Conservation and Management Act (MSA) is the primary federal statute for management of U.S. marine fisheries. Originally enacted in 1976, it established a federal system to govern fishing within the 3- to 200-nautical-mile Exclusive Economic Zone (EEZ). MSA’s fishery management system was established to meet the goals of conserving fishery resources and promoting a sustainable commercial and recreational fishing industry in the United States (U.S.).

The MSA established eight Regional Fishery Management Councils that were charged with developing fishery management plans (FMPs) designed to foster long-term biological and economic sustainability of the nation's marine fisheries, with several key objectives, including preventing the overfishing of stocks, rebuilding overfished stocks, increasing long-term economic and social benefits, and ensuring a safe and sustainable supply of seafood. Recognizing the loss of important habitat was a significant, long-term threat to fisheries, in 1996 the Sustainable Fisheries Act amended the MSA to require that Essential Fish Habitat (EFH) be described and identified. The MSA defines EFH as “waters and substrate necessary to fish for spawning, breeding, feeding or growth to maturity.” Furthermore, the MSA requires that adverse effects on EFH be minimized to the extent practicable, and that federal actions be implemented to encourage habitat conservation and enhancement.

The MSA mandates Regional Fishery Management Councils with making fishery management recommendations to the National Marine Fisheries Service (NMFS) for consideration and incorporation into the regulatory process. These recommendations could include the size of the allowable catch, the length of the fishing season, the allocation of any quotas to states and fishers, provisions for permitting and licensing or other fishery management measures suitable for achieving the management objectives of the FMPs. The Western Pacific Regional Fishery Management Council (WPRFMC) has authority over the fisheries in the Western Pacific Region, including EEZ waters surrounding the State of Hawai‘i (HI), the Territory of American Samoa (AS), the Territory of Guam, the Commonwealth of the Northern Mariana Islands (CNMI), and the U.S. Pacific Remote Island Areas (PRIA).

1.2 Fishery Ecosystem Plans

In 1996, the MSA was reauthorized and called for the creation of an Ecosystem Principles Advisory Panel (EPAP) to develop recommendations to expand the application of ecosystem principles in fisheries management. Fishery ecosystem plans (FEPs) were identified as an important mechanism for implementing ecosystem-based fisheries management (EPAP 1999), and could be used to complement the MSA's existing fishery management framework, which requires Regional Fishery Management Councils to develop FMPs that contain conservation and management measures. Per the EPAP, FEPs should contain a management framework to control the harvest of marine resources based on available information regarding the structure and function of the ecosystem in which the harvests occur.

Between 2005 and 2009, the WPRFMC replaced their FMPs with five FEPs for the Western Pacific Region containing fishery conservation and management measures in accordance with provisions as stipulated in Section 303(a) of the MSA. FEPs were developed for each of the geographical/ jurisdictional areas of the Western Pacific Region (State of Hawaiʻi, the Territory of American Samoa, the Mariana Islands, PRIA) and for Pacific-wide pelagic fisheries. These FEPs include the required provisions of an FMP and support the ecosystem-based management of the fisheries.

1.2.1 Effects of Non-fishing Activities

Fishery species and their habitats are subjected to a range of non-fishing human activities and other sources of stress. These activities can affect EFH by altering the magnitude and direction of potential stressors, which in turn may either: 1) directly affect organisms (*e.g.*, injury, mortality, etc.) and/or the biological processes that control their population dynamics (*e.g.*, reproduction, behavior), or 2) indirectly affect organisms by altering interspecies interactions or by affecting the quality or quantity of their environment through alteration of physical, chemical or ecological processes that ensure ecosystem condition, function, and persistence.

The EFH regulations require FMPs to identify non-fishing activities that may adversely affect EFH (50 CFR §600.815(4)), and to provide conservation and enhancement measures to avoid, minimize, mitigate, or otherwise offset adverse effects for federal activities, including (but not limited to): dredging; filling; excavating; mining; impounding, discharging or diverting water; discharging water with different thermal characteristics; conducting activities that contribute to non-point source pollution and sedimentation, introduce potentially hazardous materials, introduce exotic species; and converting aquatic habitat such that it eliminates, diminishes, or disrupts the functions of EFH. Any federal agency undertaking an activity that may adversely affect EFH is required to consult with the NMFS, who is responsible for issuing appropriate recommendations.

In addition to specific human activities, other “natural” stressors can exert considerable force on EFH, and in this report, are important sources of stress. These include events such as weather cycles, hurricanes/typhoons, and natural climatic variability such as the El Niño Southern Oscillation (ENSO) and Pacific Decadal Oscillation (PDO), and other stressors arising from human activities that have global scale effects, such as climate change and ocean acidification from greenhouse gas emissions. While managers cannot regulate or otherwise control these types of events, their occurrence can often be predicted and appropriate management responses can lessen the adverse effects that do and are reasonably expected to occur.

1.2.2 Cumulative Effects

Cumulative effects are effects on the environment that result from the incremental impact of an action when added to other past, present, and reasonably foreseeable future actions, regardless of who undertakes such actions. Cumulative effects can result from individually minor, but collectively significant effects resulting from two or more actions taking place over a period of time. The EFH regulations require FMPs, to the extent feasible and practicable, to analyze how the cumulative effects of fishing and non-fishing activities influence the function of EFH on an ecosystem scale (50 CFR §600.815(5)).

1.3 Purpose of this Report

Under the MSA, a review of information available on EFH must be completed at least once every five years, and EFH provisions of FMPs must be revised or amended, as warranted (50 CFR §600.815(10)). This five-year review should evaluate published scientific literature, unpublished scientific reports, information solicited from interested parties, and previously unavailable or inaccessible data. The WPRFMC reviews and updates the EFH section of the Western Pacific Region FEPs based on a five-year schedule of rotating reviews through its annual Stock Assessment and Fishery Evaluation report process.

This report is intended to review the potential effects (including potential cumulative effects) resulting from a range of non-fishing activities and other potential sources of stress. This review is intended to gather new information on: (1) non-fishing activities that may adversely affect EFH, (2) known and potential adverse effects of these activities on EFH, and (3) options to avoid, minimize, mitigate, or otherwise offset adverse effects on EFH. This information will assist the WPRFMC in determining whether modifications to the existing non-fishing effects sections of the five Western Pacific Region FEPs are warranted. While this information is highly valuable to inform impacts-analyses, the goal was not to address the approach to EFH consultations.

This review includes the following sections:

1) A brief description of the marine and estuarine ecosystems that comprise EFH in the Western Pacific Region (2.0 EFH in the Western Pacific Region).
2) A discussion, by broad categories, of the non-fishing activities and other sources of stress that could affect EFH in the Western Pacific Region, (3.0 Non-fishing Activities and Other Sources of Stress).
3) An assessment of potential effects of stressors on the marine and estuarine ecosystem that comprise the region’s EFH (4.0 Ecological Stressors in the Marine Environment).
4) A discussion of cumulative effects with specific guidance for assessing the effects of multiple stressors (5.0 Cumulative Effects).
5) A list of conservation measures to avoid, minimize, mitigate, or otherwise offset adverse effects (6.0 Conservation and Enhancement Recommendation).
6) A comprehensive bibliography of relevant references reviewed and cited in this report (
7)
8) 8.0 References).

2.0 EFH in the Western Pacific Region

Regional Fishery Management Councils, with assistance from the NMFS, must identify and describe EFH for all Management Unit Species (MUS). EFH is defined as the waters and substrate necessary to a fishery species (*e.g.*, finfish, mollusks, crustaceans and all other forms of marine animal and plant life other than marine reptiles, marine mammals and birds) for spawning, breeding, feeding, or growth to maturity. EFH for managed fishery resources in the Western Pacific Region has been designated in the FEPs prepared by the WPRFMC and includes designations for five MUS: Bottomfish and Seamount Groundfish, Crustaceans, Precious Corals, Coral Reef Ecosystems, and Pelagic species.

For this report, an ecosystem refers to any taxonomically-diverse assemblage of species and the non-living components of their environment that interact with the unit or system (*e.g.*, a coral reef ecosystem). In contrast, habitat is the physical surroundings that influence and is used by a species (*e.g.*, sandflats are feeding habitat for many goatfishes). Due to a lack of habitat-related data for most MUS, the WPRFMC has broadly defined EFH to include all waters to a depth of 1,000 meters (m) and benthos to a depth of 700 m within the EEZ and encompassing all marine and estuarine ecosystems of the Western Pacific jurisdictions. In this report, effects to EFH are evaluated from the context of individual ecosystem function within a designated EFH because the EFH identified for all MUS are often comprised of multiple marine and estuarine ecosystems (The marine and estuarine ecosystems comprising the EFH designations for the nine species complexes (comprising six MUS groups) in the Western Pacific Region.). In addition, most ecological studies assessing the ecological effects of non-fishing activities are conducted at the organismal and ecosystem scales, and each ecosystem may display a different response to a given activity. As such, the broad definition of EFH in the five FEPs creates management and regulatory challenges due to the range and diversity of non-fishing activities (see

3.0 Non-fishing Activities and Other Sources of Stress) that occurs within these numerous and diverse marine ecosystems, and the potential effects of those activities on the stressors that impact these ecosystems. Additional refinement of the effects of non-fishing activities on EFH, and subsequent management of them, would benefit from a narrowing of the EFH designation to better describe the habitat of species within each MUS group.

Ecosystem structure and function varies over time due to a suite of dynamic and interacting processes (Christensen *et al.* 1996, Kay and Schneider 1994, EPAP 1999). Boundaries of marine ecosystems are often difficult to clearly and unambiguously delineate because most are interlinked by population- and ecosystem-level processes critical to each ecosystems' proper function and persistence. Although marine ecosystems are generally open systems, bathymetric and oceanographic features allow them to be reasonably identified (EPAP 1999), and for management purposes, WPRFMC has delineated them geographically, making them place-based. Each ecosystem type, as defined in the five Western Pacific Region FEPs, is discussed briefly below.

2.1 Benthic Ecosystems

Benthic ecosystems are those found on the bottom of the ocean, beginning at the shore line (*e.g.*, the intertidal, mangroves, etc.) and extending subtidally out to sea. Unlike continental coastal

Table 1. The marine and estuarine ecosystems comprising the EFH designations for the nine species complexes (comprising six MUS groups) in the Western Pacific Region.

MUS Group/Species Complex	Ecosystems within the EFH
Bottomfish and Seamount Groundfish	
Bottomfish	Deep reef slopes (<400 m), banks and seamounts, pelagic
Seamount Groundfish	Banks and seamounts at Hancock Seamounts (80-600 m), pelagic
Crustaceans	
Crustaceans: spiny and slipper lobsters, Kona crab	Coral reef, banks and seamounts, pelagic
Crustaceans: deepwater shrimp	Deep reef slopes, banks and seamounts, pelagic
Precious Coral	
Precious coral: deep-water complex	Deep-reef slopes, deep ocean floor, banks and seamounts, pelagic
Precious coral: shallow-water complex	Coral reef, deep reef slopes (to 100 m)
Currently-harvested Coral Reef Ecosystem	Coral reef, intertidal, seagrasses, mangroves, deep-slopes, banks and seamounts, pelagic

Potentially-harvested Coral Reef Ecosystem	Coral reef, intertidal, seagrasses, mangroves, deep-slopes, banks and seamounts, pelagic
Pelagic	Pelagic (<1,000 m), banks and seamounts

waters, islands within the Western Pacific Region tend to have narrow subtidal shelfs that support species-rich, nearshore marine ecosystems (*e.g.*, coral reefs, seagrass beds, etc.) that slope steeply into deep-water ecosystems (Figure 1). Consistent with those included in the Western Pacific Region FEPs, this section presents a brief description of the following benthic ecosystems: (a) intertidal, (b) mangrove forests or mangals, (c) seagrasses, (d) coral reefs, (e) deep reef slopes, (f) banks and seamounts, and (g) deep-ocean floor.

2.1.1 Intertidal

The intertidal zone exists between the highest and lowest extent of the tides and spends at least part of its time exposed to air. The duration and frequency of exposure is correlated with the vertical position on the shore; areas closer to the high tide mark are more frequently exposed and for longer durations than areas closer to the low tide mark. Intertidal areas can be comprised of hard (*e.g.*, basalt, limestone, etc.) or unconsolidated (*e.g.*, sand, cobble, etc.) substratum, which will dictate the types of associated fauna. Sandy shallows and tidal pools are important nursery areas for many subtidal invertebrate and fish species (Major 1978, Leber *et al.* 1998, Cox *et al.*

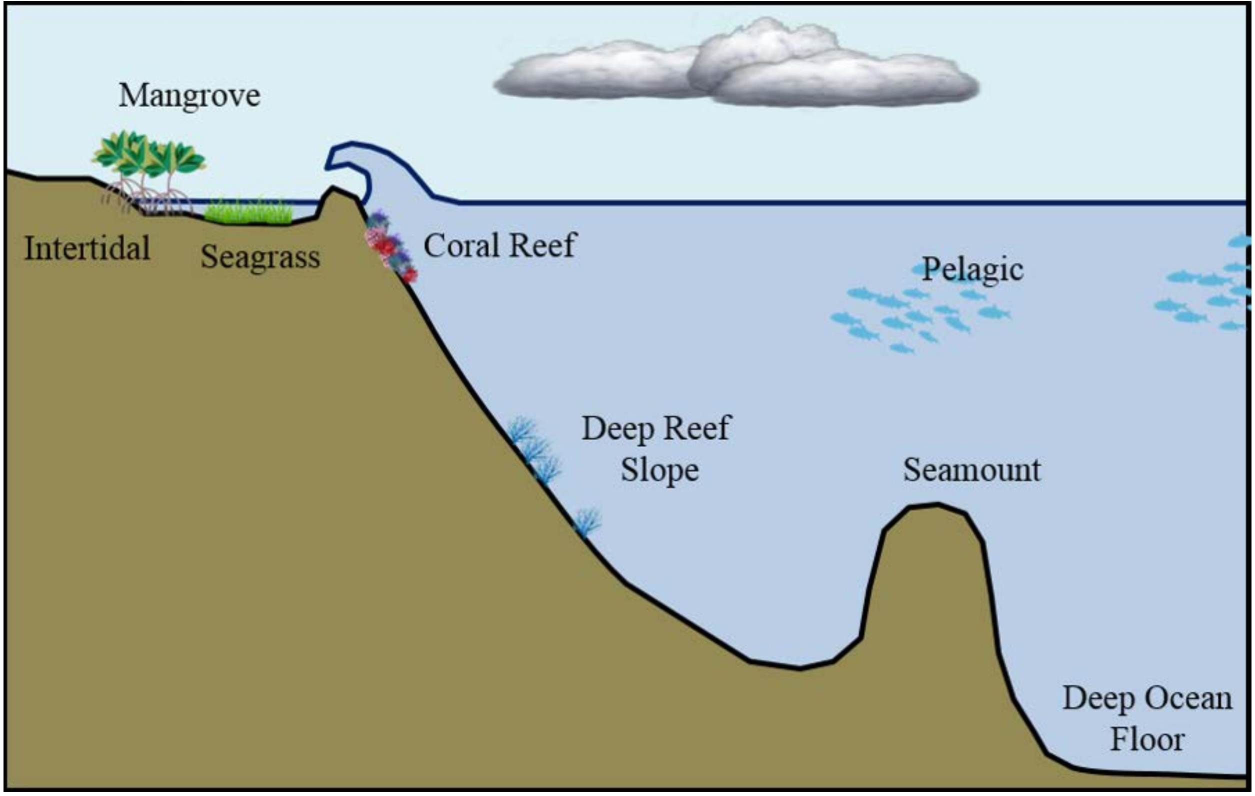

Figure 1. Schematic of the marine ecosystems that comprise the EFH of the Western Pacific Region.

2011, Iglesias 2012), including those that spend their adult life in other marine ecosystem such as coral reefs.

Intertidal organisms often display pronounced vertical zonation, where the lower limits of organisms are often determined by the presence of predators or competing species, and the upper limits are controlled by physiological limits and species' tolerance to temperature and drying (Garrity 1984, Levington 2001), although in the tropics, this may not always be the case (Minton and Gochfeld 2001). Due to challenging environmental conditions, intertidal areas generally have lower species richness and diversity than subtidal areas.

Along tropical rocky intertidal areas, marine algae and epilithic biofilms (comprised of cyanobacteria and diatoms) are the principle primary producers (Williams 1993, Williams *et al.* 2000, Macusi and Ashoka Deepananda 2013). Primary consumers such as snails and sea urchins graze on algae and biofilms, and support an array of secondary consumers that include a variety of invertebrates, sea birds and fish (Williams *et al.* 1993). Sandy intertidal areas usually support lower diversity than rocky intertidal areas, and may include a variety of burrowing mollusks, crustaceans, and worms, depending upon the amount of wave energy, which directly controls sediment grain size. Intertidal organisms are marine, and nearly all have a life history stage—usually a planktonic larval stage—that is dependent upon the ocean.

2.1.2 Mangrove Forests (Mangal)

Mangrove forests, or mangals, are tropical, coastal, forest ecosystems comprised of mangrove trees, which are adapted to grow in saline or brackish water. Mangrove forests are generally characterized as depositional coastal environments (Victor *et al.* 2004), where fine sediment, often high in organic content, collects in areas protected from high-energy wave action (Barbier *et al.* 2011). They help stabilize shorelines and reduce effects of natural disasters such as tsunamis and hurricanes (Scavia *et al.* 2002). Due to their high productivity and relatively sheltered environment, mangroves in some areas serve as important nursery habitat for many ecologically and commercially important coral reef fishery species, although research from several areas in the Pacific suggests that mangroves are less important than other coastal ecosystemns as nursery habitat for certain species (Laegdsgaard and Johnson 1995, Thollot 1992, Tupper 2007). Where mangroves have been found to be important as nurseries, they tend to have water quality conditions (*e.g.*, salinity, turbidity, etc.) similar to coral reefs (Cocheret de la Morinière *et al.* 2002), whereas in areas in which mangroves were not important reef fish nurseries, water tended to be less saline and more turbid. This is consistent with findings that juveniles of reef fishes inhabit the lower, more saline areas of mangals until migrating to the coral reef (Parrish 1989, Mumby *et al.* 2004, Abu El-Regal and Ibrahim 2014). Other fishes and crustaceans remain in the mangal throughout their adult lives, including mangrove crabs, which live in burrows among the mangrove roots. Mangals also provide food, medicine, fuel and building materials for certain local communities (Mumby *et al.* 2004 Gilman *et al.* 2006, Giri *et al.* 2011).

Mangrove trees possess an intricate salt filtration system (Lopez-Hoffman *et al.* 2007) and a complex root system to cope with salt water immersion, anoxic sediment, and wave action (Ball 1988). They can tolerate conditions ranging from brackish water to water with over twice the salinity of ocean water. Mangrove species zonation is generally correlated with soil water salinity (Ball 1988, Ukpong 1994), with less tolerant species located along the landward side of the forest or near freshwater inputs (*e.g.*, rivers). Some mangrove tree species have elaborate prop roots systems that form important substratum on which sessile organisms can settle and grow (MacDonald and Weis 2013), and which provide habitat for a variety of invertebrates and fish (Nagelkerken *et al.* 2010).

The natural eastern limit of mangroves in the Pacific is American Samoa (Ellison 1999), although three species (*Rhizophora mangle*, *Bruguiera gymnorrhiza*, and *Conocarpus erectus*) have become established in Hawaiʻi since their introduction in the early 1900s, with *R. mangle* becoming the dominant plant in protected bays and along coastlines on all of the main islands (Allen 1998). While mangroves are highly regarded in most parts of the tropics for the ecosystem services they provide, in Hawaiʻi they have significant negative ecological and economic effects, including reduction in habitat quality for native coastal wetland and mudflat species, displacement of native species in endemic ecosystems (*e.g.*, in anchialine pools), and overgrowth of native Hawaiian archaeological sites (Allen 1998, Chimner *et al.* 2006). Their values as nursery habitat for juvenile reef fish species is unclear, but generally they are considered detrimental.

Mangrove communities in American Samoa are composed of two species, *Bruguiera gymnorrhiza* and *Rhizophora mangle.* A majority of mangrove areas in American Samoa have been filled for residential and commercial development and roads since the early 1900s, and only five significant mangrove stands remain, covering approximately 52 hectares (ha) (Gillman *et al.* 2006). The role of mangroves in American Samoa as juvenile habitat for coral reef fish is unclear. Although numerous species are known to use areas fringed by mangal, the role of the forest themselves are unclear (Volk 1993).

In the Mariana Islands, mangroves cover an estimated 80 ha (Gillman *et al.* 2006) and comprise four species (*Rhizophora mucronata, R. apiculata, Bruguiera gymnorrhiza, Avicennia marina*). Only a single species is present in the CNMI (*Bruguiera gymnorrhiza*). Some mangrove areas on Guam (*e.g.*, Sasa Bay) have been identified as nursery habitat for jacks, barracudas, snappers, groupers, rabbitfish, mojarras, milkfish, and mullets (Wiles and Ritter 1993).

2.1.3 Seagrass Beds

Seagrasses are marine flowering plants widely distributed along tropical coastlines in the Western Pacific Region. Globally, seagrasses have an important role in fisheries production, and sediment accumulation and stabilization (, Jackson *et al.* 1989, Green and Short 2003, Dorenbosch *et al.* 2005, Larkum *et al.* 2006, Unsworth and Cullen 2008, Unsworth *et al.* 2010). Highly productive seagrass ecosystems have a relatively complex physical structure that provides a combination of food and shelter. This results in high biomass and secondary productivity, including for important fishery species in the Indo-Pacific (Parrish 1989, Beck *et al.* 2001, Honda *et al.* 2013, Nadiarti *et al.* 2015). In some area of the Pacific Ocean, seagrasses provide nursery area for species that support adjacent ecosystems, such as coral reefs and mangrove forests (Unsworth *et al.* 2010, Honda *et al.* 2013). While seagrasses may be less important in the Western Pacific Region as nursery habitat for fish and invertebrates, they are used in some jurisdictions by juvenile rabbitfish, goatfish, and snappers (Jones and Roberts 1975).

The role of seagrasses in binding sediment is important. Seagrass shoots baffle currents, thereby encouraging the settlement of sediment and inhibiting its resuspension (Short and Short 1984, Ward *et al.* 1984). By enhancing sediment retention, and through the relatively rapid uptake of nutrients both by seagrasses and their epiphytes, seagrass ecosystems can remove nutrients and other contaminants from the water column (Barbier *et al.* 2011). Once removed, these nutrients can be released more slowly through the eventual decomposition and consumption of leaf matter, thereby reducing problems of eutrophication and organic pollutants (Hemminga and Duarte 2000). Several studies that have documented the importance of seagrasses in reducing erosional forces during storm events (Koch *et al.* 2006, Barbier *et al.* 2011, Ganthy *et al.* 2014).

Seagrass diversity decreases from west to east across the Western Pacific Region. The Mariana Islands have three seagrass species (Lobban and Tsuda 2003), several of which form extensive and dense beds, especially on Saipan. American Samoa (Skelton 2003) and Hawai‘i (McDermid *et al.* 2002) each have two species, both small in stature, which affects their functional ability to baffle currents and provide sediment stabilization and shoreline protection. However, they are still important sources of food for many species, including sea turtles (Russell *et al.* 2003).

2.1.4 Coral Reefs

Coral reefs are carbonate rock structures and associated unconsolidated substratum (*e.g.*, interspersed sand and rubble) that support viable populations of reef-building organisms, including scleractinian corals and coralline algae, and a variety of associated invertebrates and fish. Coral reef ecosystems are among the most abundant and diverse ecosystems on Earth, rivaling tropical rainforests in terms of biomass and species diversity (Roberts *et al.* 2002, Hughes *et al.* 2003). As such, coral reefs are also geologically, evolutionarily, and ecologically complex (Hatcher *et al.* 1989).

Due their reliance on light for photosynthesis, coral and other reef-building organisms are confined to the depths where light sufficient to conduct photosynthesis penetrates—known as the euphotic zone—although some predominately non-reef-building coral species can occur in the deeper ocean zones (see 2.1.5 Deep Reef Slopes, 2.1.6 Banks and Seamounts, and 2.1.7 Deep Ocean Floor). Maximum reef growth and productivity generally occurs between approximately five and 15 m (Hopley and Kinsey 1988), but the maximum depth at which reefs can grow depends on water clarity and photosynthetic capability, which is highly variable among species (Baker 2001, Yentsch *et al.* 2002, Baird *et al.* 2003). Maximum biodiversity of coral reef species usually occurs between 10-30 m (Huston 1985).

Four primary reef types are found in the Western Pacific Region. Fringing reefs grow directly along the shoreline of islands and often include a shallow (<2 m) reef flat before sloping into deeper water. Given their relatively shallow waters and proximity to the shoreline, fringing reefs are often exposed to more human activity than other reef types. Barrier reefs are shallow reef systems that are separated from the shore, generally by a relatively shallow (<10-20 m) lagoon system. Barrier reefs are relatively rare in the jurisdictions of the Western Pacific Region, with the barrier reefs in Kāneʻohe Bay, Hawaiʻi, Cocos Lagoon, Guam, and Saipan Lagoon, Saipan being the most prominent examples. Patch reefs are comparatively small, often circular reef outcroppings that rise up from the bottom of lagoons or other relatively shallow embayments to within a few meters of the surface (*e.g.*, Kāneʻohe Bay, Hawaiʻi and Apra Harbor, Guam). Atolls are continuous barrier reef-like structures that enclose a lagoon and have no central island. Most atolls have one or more channels through the reef that allows water exchange between the lagoon and the ocean. Patch reefs are commonly found within the atoll's lagoon. Atolls may or may not have one or more low-relief, coral and rubble islands atop the reef structure. Atolls are prominent in the Northwestern Hawaiian Islands and the PRIA.

Reef-building corals are the primary providers of physical structure upon which associated organisms depend for food and shelter (Alvarez-Filip *et al.* 2009), and loss of this structure is often referred to as “flattening” of the reef. The symbiotic relationship between coral and algal cells, known as zooxanthellae, is a key feature of reef-building corals (Roth 2014). Zooxanthellae provide much of the polyp’s nutritional needs, and play a critical role in the coral's ability to accrete carbonate from the water column to construct its skeleton, a process called calcification (Colombo-Pallotta *et al.* 2010). The rate at which a reef can calcify is among its most important ecological functions because persistence of the coral reef ecosystem depends on rate of calcification exceeding the rate of erosion (Wilkinson and Buddemeier 1994).

A healthy, functioning coral reef ecosystem is comprised of more than corals. In addition to coral zooxanthellae, other important primary producers on coral reefs include phytoplankton, macro- and micro-algae, benthic bacteria, and seagrasses. Primary consumers include many species of mollusks, crustaceans, echinoderms, gastropods, sea turtles, and herbivorous fish. Secondary consumers include anemones, crustaceans, and fish, including several important fishery species. Tertiary consumers include eels, octopuses, barracudas, sharks (sometimes referred to as apex predators), and monk seals in Hawaiʻi. While many coral reef species rely on the hardbottom areas on which coral colonies grow, associated sand patches and algal and seagrass beds, often serve as important feeding or spawning habitat for many species (*e.g.*, goatfishes, some wrasses, squid, etc.). Some coral reef organisms also use mangroves, seagrass beds, and intertidal ecosystems for nursery areas (*e.g.*, jacks, barracudas, snappers, rabbitfish, etc.), and these coastal ecosystems also play important roles in ecosystem processes on coral reefs, such as nutrient cycling.

The diversity of nearly all coral reef organisms declines in an easterly direction across the Pacific Ocean (Stoddart 1992, Reaka *et al.* 2008). While taxonomy can vary among observers, ~375 species of reef-building corals have been identified from the Mariana Islands (Randall 2003), ~220 species from American Samoa (DiDonato *et al.* 2006), 59 species from Hawaiʻi (Maragos *et al.* 2004) and between 47 and 173 species on each of the PRIA (Kenyon 2010). As coral species richness declines, reefs tend to lose specific coral genera and families and their associated reef functions. For example, the genus *Acropora* is absent from the main Hawaiian Islands (with some rare exceptions, see Walsh *et al.* 2014, Kosaki *et al.* 2013). *Acropora* species, and especially tabular *Acropora*, provide a complex three-dimensional structure, a key ecological feature for coral reefs. Among mollusks, species with large larval forms and/or short planktonic durations are under-represented or absent from Hawaiian reefs (Paulay and Meyer 2006), and more prevalent Western Pacific Ocean reefs such as the Mariana Islands.

2.1.5 Deep Reef Slopes

Unlike continental areas, the jurisdictions in the Western Pacific Region lack extensive shallow water shelves around their perimeter; instead, relatively narrow fringing reefs generally slope steeply into deep water not far from shore. The benthic communities on these deep reef slopes are zoned in relation to light penetration. Where light is still sufficient for photosynthesis, deep-water reef-building corals will continue to grow where appropriate substratum is available. These mesophotic coral ecosystems (MCE), found at depths of nearly 200 m (Baker *et al.* 2016), have been hypothesized to serve as refugia for shallow reef species, especially those subject to significant fishing pressure and/or other non-fishing stresses (Glynn 1996, Blyth-skyrme *et al.* 2013, Lindfield *et al.* 2014, Muir *et al.* 2015). Deep reef slopes are also home to a diversity of marine organisms, including many important fishery species (Lindfield *et al.* 2014) and antipatharian coral, *i.e.*, precious corals.

Relatively little is known about deep reef slope ecosystems, but recent technological advances have made it possible to conduct scientific investigations of MCE, which inhabit the upper boundary of this area, where low levels of light still penetrate. Significant work to characterize these assemblages has recently been undertaken in several of the jurisdictions in the Western Pacific Region (*e.g.*, survey work by the NOAA Coral Reef Ecosystem Program).
At shallower depths (50 to 80 m) in Hawaiʻi, large *Halimeda* meadows and diverse macroalgal assemblages (*Lobophora variegata*, *Dictyota friabilis*, coralline algal rhodoliths, *Mesophyllum mesomorphum*, and *Peyssonnelia rubra*) have been observed covering both hard and soft substrata. These macroalgal communities generally do not comprise significant habitats for large-bodied fishes in the main Hawaiian Islands (Pyle *et al.* 2016), although endemic reef-associated fishes have been found in deep water *Microdicyton* (algae) beds in the Northwestern Hawaiian Islands (Kane *et al.* 2014). At greater depths, abundance of macroalgae declines and hard substratum is often dominated by monospecific stands of the hard coral *Leptoseris* spp. (Rooney *et al.* 2010, Pyle *et al.* 2016). Below approximately 100 m, live benthic cover was uniformly low, but on hardbottom features exposed to currents, precious black corals and the invasive octocoral *Carijoa* sp. could be locally abundant, with the latter often overgrowing large black coral colonies (Kahng and Grigg 2005).

Limited work in American Samoa has confirmed reef-building MCE at depths as great as 110 m. Encrusting corals belonging to the genus *Montipora* and massive corals in the genus *Porites* were most abundant at shallow depths with their cover gradually decreasing as depth increased. At depths of 60 to 70 m, plate corals in the genus *Acropora* dominated the MCE, giving way to species in the genera *Leptoseris*, *Pachyseris*, or *Montipora*. Branching coral cover was high in the 80 to 110 m depth range (Bare *et al.* 2010).

Extensive mesophotic reefs have been observed seaward of the Saipan Lagoon barrier reef, mainly on the Garapan Anchorage. Lindfield *et al.* (2016), using baited camera drops on Guam, Saipan, Tinian, and Rota, found high fish abundance on MCE (35-90 m) compared to inshore reefs (10-35 m), and suggest that MCE represent a depth refuge for many coral reef fish species.

They also noted that coral structure disappeared at depths greater than 70 m and fish abundance decreased. At depths greater than 70 m, unconsolidated sediment was the primary bottom feature (Lindfield *et al.* 2016). In addition to hard scleractinian corals, sea fans, a type of soft coral, were a common feature on hard substrate at mesophotic depths in the Mariana Archipelago (Blythe-Skyrme *et al.* 2013).

Data are insufficient to identify the location or density of MCE in the PRIA, but the presence of deep-water corals (165 m) at Johnston Atoll (Kahng and Maragos 2006), along with the clear oligotrophic waters minimally influenced by terrigenous inputs, suggests that MCE are likely present at most or all islands within the PRIA (Blyth-Skyrme *et al*. 2013).

2.1.6 Banks and Seamounts

In the Western Pacific Region, banks and seamounts are submerged features formed by undersea volcanos. During the formation of seamounts, they never reached the surface of the ocean and thus maintain a generally "mountainous" shape, with steep slopes and relative little flat area on top of them. Banks are less specifically defined, but comprise shallow areas rising up from relatively deep waters that may have been formed by a submerged part of a larger landmass or a submerged atoll. Over 50,000 seamounts may exist in the Pacific Ocean (Rogers 2004), and banks and seamounts are found in all jurisdictions in the Western Pacific Region.

Seamounts can have a significant effect on the pelagic environment. They may deflect major ocean currents (*e.g.*, the Emperor Seamount Chain deflects the Kuroshio Current), and have the potential to form eddies, called Taylor Columns, that may become trapped or shed downstream (White and Mohn 2002, Rogers 2004). Taylor Columns are associated with the upwelling of nutrient-rich water from the deep ocean, and may lead to increased productivity in the upper waters above or downstream of seamounts (Brainard 1986, Rogers 2004), and may help retain pelagic larvae, although evidence for larval retention over seamounts, especially small ones, is sparse (Boehlert and Mundy 1993, Sponaugle *et al.* 2002).

In the Western Pacific Region, coral reef ecosystems tend to be found on the shallower parts of banks and seamounts, but can extend downslope into the mesophotic zone. Deeper parts of seamounts and banks may be composed of rock, coral rubble, sand, or shell deposits. Bank and seamount assemblages tend to be dominated by those found on nearby shallow areas and do not have unusual diversity or endemism (Howell *et al.* 2010). Seamounts and banks are important feeding and reproduction grounds for many deep water or pelagic species of fish. Plankton biomass may be increased over and around seamounts and form a source of prey for seamount-associated species (Rogers 2004). This forms the basis for the WPRFMC’s designation of the water column down to 1,000 m above seamounts with summits shallower than 2,000 m as Habitat Areas of Particular Concern for the Pelagic MUS.

2.1.7 Deep Ocean Floor

The deep ocean (waters and seafloor deeper than ~200 m), supports a high diversity of ecosystems and species (Hessler and Sanders 1967, Grassle and Maciolek 1992, Sogin *et al.* 2006, Ramirez-Llodra *et al.* 2010, Mora *et al.* 2011), as well as abundant mineral resources

(Herzig and Hannington 1995, Kato *et al.* 2011). Relatively little is known about this region due to the challenges associated with studying this environment, limiting our understanding of the resilience of this ecosystem to and its recovery from adverse effects. The deep ocean has a role in nutrient regeneration and global biogeochemical cycling that is essential for sustaining primary and secondary productivity in the oceans, and adverse effects that decrease the biodiversity of the deep ocean could affect this important ecosystem function (Danovaro *et al.* 2008). Pressure to extract deep ocean resources is increasing (Mengerink *et al.* 2014), including fishing, drilling for hydrocarbon extraction, and mining of rare earth elements (*e.g.*, Morato *et al.* 2006, Benn *et al.* 2010).

The deep ocean floor is generally comprised of soft-sediment, but biologically created "hardbottom" can cover tens of square kilometers and provide extensive three-dimension relief (Thurber *et al.* 2014). Probably the best-known example of biogenic habitat in the deep ocean is created by “cold-water” corals. Submersible explorations in Hawaiʻi have revealed that gorgonian-like corals (*e.g.*, "bamboo corals") and other antipatharian corals (*e.g.*, "precious" corals) can form complex hard structures with their skeletons (NOAA 2009). These areas often have high species diversities because of increased access to dietary resources and refuge from predators or physical disturbance, and may provide a nursery habitat for deep-ocean species including fish (Miller *et al.* 2012).

2.2 Pelagic Environment

The entirety of the water column overlying the benthos is the pelagic zone of the ocean, although the description of EFH for the pelagic MUS includes only the uppermost 1000 m. It comprises the largest ecosystem in the Western Pacific Region, and is the primary connection between all benthic marine ecosystems. Nearly all marine organisms spend all or part of their life in the pelagic environment.

Average primary productivity in the tropical open ocean is among the lowest of all marine ecosystems, typically around 40 grams (g) of carbon/m^2/year (Carpenter 1998). Warm conditions in the tropics promote thermal stratification in the upper layer of the ocean and prevent mixing with lower, cooler, nutrient-rich water (Carpenter 1998). However, in upwelling areas, including waters near oceanic islands and some seamounts (from Taylor Columns), nutrients are brought from the deep ocean into the sunlit upper layers, where phytoplankton can access it, thus increasing primary productivity.

Along the equator in the Central Pacific (near several of the PRIA) is an upwelling area caused by the diverging flow of the North Equatorial Current and the Equatorial Countercurrent (Chavez and Barber 1987). Additionally, the Western Pacific Warm Pool (WPWP) is an area of water with surface temperatures consistently above 28°C (Yan *et al.* 1992), creating a highly stratified water column and little vertical mixing. The waters within the WPWP are nutrient poor, and productivity is low. However, along the edge of the WPWP are convergence zones that upwell nutrient-rich waters from depth (Helber and Weisberg 2001), promoting high primary productivity. This edge area has high densities of tuna and is commercially important. In coastal waters (especially around high islands), productivity is greater than the open ocean, primarily because of land-derived nutrient inputs, including from groundwater discharge (Knee 2010).

Phytoplankton represent several different types of microscopic photosynthetic organisms and occur primarily in the upper 100 m of the water column. Phytoplankton includes organisms such as diatoms, dinoflagellates, coccolithophores, and cyanobacteria. Many of these organisms deposit skeletons by precipitating dissolved minerals (primarily silicates and carbonates) from the water column. Although some phytoplankton such as dinoflagellates have structures that allow them to move (especially vertically through the water column), the distribution of many phytoplankton is controlled by oceanic currents.

The secondary productivity from zooplankton in the Western Central Pacific Ocean roughly mirrors the pattern of primary productivity (Carpenter 1998). Highest zooplankton production is found in upwelling areas, but is generally lower than that found in most coastal areas (Carpenter 1998). Zooplankton include organisms such as copepods, cheatognaths, euphasids, ostracods, amphipods, and many other microscopic invertebrates. Larvae and gametes of marine macro-organisms, including pelagic fish and coral reef-associated fish and invertebrates, are also an important component of the zooplankton (King and Demond 1953).

Large-scale oceanographic events (*e.g.*, ENSO, PDO, etc.) change the characteristics of water temperature and productivity across the Pacific, and have a significant effect on open ocean productivity.

3.0 Non-fishing Activities and Other Sources of Stress

Numerous types of non-fishing activities and other sources of stress occur in the Western Pacific Region. These activities affect EFH by altering the magnitude and direction of potential stressors (see 4.0 Ecological Stressors in the Marine Environment for discussion of specific stressors) directly affecting organisms or changing the quality or quantity of their environment (Figure 2). The potential effects of a specific activity on a marine ecosystem are dependent on the location, size, timing, duration, method, etc. of the specific activity. It would be impossible to list and discuss every non-fishing activity in detail; however, many specific activities have sufficient similarities among the stressors they affect to allow them to be grouped into generalized categories to more easily examine their potential effects on EFH.

The implementing regulations for the Sustainable Fisheries Act, which amended the MSA and created the provision for EFH, focused on a diverse array of human activities and stressors (*e.g.*, coastal development projects, mining, sedimentation, nutrient loading, etc.) that could adversely affect EFH, but in doing so created a confusing mixture of human activities and ecological processes that can cause ecosystem change. Additionally, some potentially significant, non-fishing sources of stress were not adequately considered and analyzed in the subsequent FEPs developed by the WPRFMC, including the potential effect of climate change, which the WPRFMC has subsequently required for consideration in its management decisions through its Marine Planning and Climate Change Policy. Climate change is likely to be the most significant source of stress on EFH in the Western Pacific Region in the coming decades.

This report attempts to clearly delineate human activities and sources of stress from the stressors themselves. Doing so allows for a clearer understanding of potential effects because different activities often alter the intensity, duration, frequency, timing, and/or scale of the same stressor, which results in similar effects on a marine or estuarine ecosystem (Figure 2). For example, physical damage to a coral from the anchor chain of a large vessel dragging on the bottom would likely have similar effects to the damage caused from the underwater detonation of ordnance. The human activities and other sources of stress are discussed in subsections, and concluded with a summary table listing the stressors associated with the activity. Detailed information on the stressors themselves is the subject of 4.0 Ecological Stressors in the Marine Environment.

3.1 Climate Change

Climate is the long-term (usually decades or longer) average weather pattern in a specific place or region. These average patterns are subject to natural cycles that contribute to short-term (annual or decadal) variability (*e.g.*, ENSO, PDO), but which do not result in long-term changes in average condition. **Climate change** is a long-term change in the state of climate that may encompass a change in average weather conditions and/or a change in the variability of that average condition, for example, more or fewer extreme weather events (IPCC 2007). The primary source of climate change – atmospheric accumulation of CO_2 – will also directly affect the acidity of the ocean, and thus ocean acidification is often considered a part of climate change

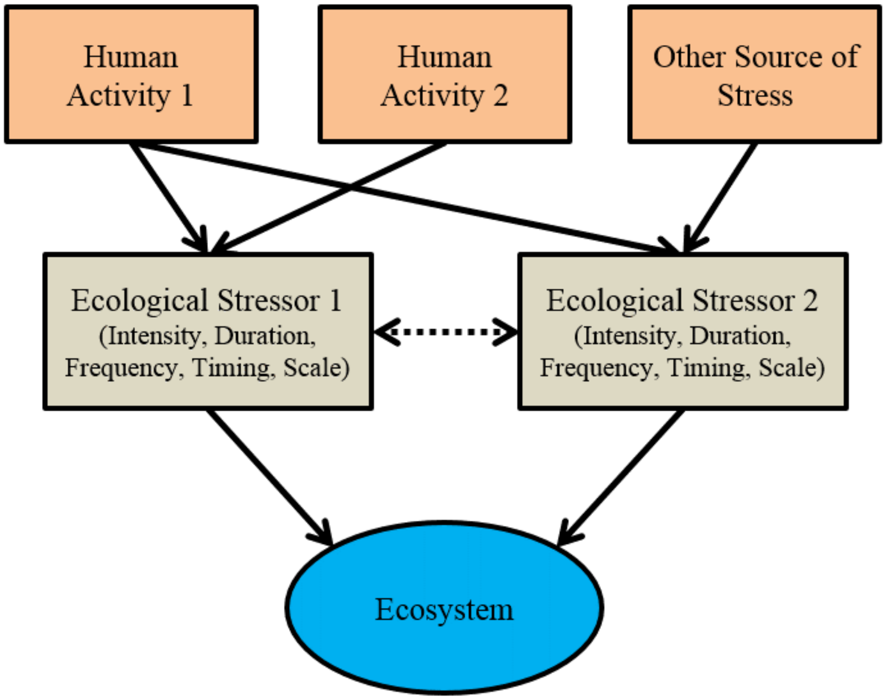

Figure 2. Conceptual flow diagram showing the linkage of human activities and other sources of stress on an ecosystem. Activities and sources of stress alter the intensity, duration, frequency, timing and/or scale of potential ecological stressors, which act directly on species or ecological processes in the ecosystem. Different activities often affect the same ecological stressor(s), and stressors often interact with each other (dotted arrow), resulting in a variety of potential responses (see 5.0 Cumulative *Effects*).

even though it is not actually a climatological feature. The WPRFMC has “adopted the definition of climate change used by the Intergovernmental Panel on Climate Change (IPCC) to include natural climate variability such as ENSO and other patterns of natural variability as well as long-term changes in climate associated with anthropogenic (human) influence on greenhouse gases and other aspects of the Earth's climate system. The definition of climate change in this policy also includes ocean acidification” (WPRFMC 2015). Numerous factors contribute to climate change, including biological processes, variations in solar radiation, geological processes, and some human activities (National Academy of Science 2010).

Climate change is predicted to affect the jurisdictions in the Western Pacific Region in the following ways:

- American Samoa is expected to experience increased surface air temperature and sea-surface temperature, and the intensity and frequency of extreme heat events are expected to increase. Rainfall is expected to stay approximately the same, but the frequency of extreme rain events is expected to increase under current climate change scenarios (PCEP 2016). The number of hurricanes are expected to decline in the south-east Pacific Ocean Basin (Lagomauitumua *et al.* 2010), likely causing a decrease in hurricanes affecting American Samoa. Ocean acidification is expected to increase, and sea level is expected to rise.

- The Hawaiian Archipelago extends across a wide latitudinal range and is comprised of high and low islands. Thus, climate change effects such as rainfall and ocean acidification will likely vary across the archipelago, but to what degree is uncertain. To date research has focused on the southerly high islands, where the archipelago's human population lives. The Hawaiian Islands are expected to experience increased air and sea surface temperatures (Giambelluca *et al.* 2008, Sea Grant 2014). Anticipated decreases in prevailing northeasterly trade winds are expected to result in an overall decline in annual rainfall, which is consistent with observations over the past 40 years (Chu and Chen 2005). Extreme rainfall events and occurrences of drought are also expected to increase (Chu *et al.* 2010), resulting in extended dry periods and more flash flooding. Changes in rainfall patterns will potentially affect aquifer recharge and ground water flow into the coastal marine environment. Ocean acidification is expected to increase across the archipelago, and sea level is expected to rise from 0.3-1 m (1-3 feet (ft)) by the end of the century (Sea Grant 2014).

- The Mariana Islands are expected to experience higher air and sea surface temperatures. It is currently unclear how rainfall in the Mariana Islands will be affected. Guam may experience fewer, but more intense, storms (Lander 2004), but Saipan may see only a small increase in average rainfall and extreme rainfall events, but may experience "wetter" wet and "drier" dry seasons, *i.e.*, increased variability in rainfall (Greene and Skeele 2014). Ocean acidification is expected to increase, and sea level is expected to rise >1 m (>3 ft) by the end of the century (PREL 2014).

- The PRIA are spread across the Pacific Ocean, from south of the equator to the northern extent of coral reef distributions, and from the western to central Pacific. Therefore, the effects of climate change are expected to vary across these geographically dispersed islands, but it may be possible to predict the broader effects based on predicted changes in nearby jurisdictions for which information is currently available. A common feature of most of these island areas is their relatively low topographic relief and extensive coral reef structure. As such, increases in sea surface temperature and ocean acidification (Royal Society 2005, IPCC 2014), and a rise in sea level will affect all island areas within the PRIA, and are expected to be the most serious stressors associated with climate change.

- The open ocean, home to important pelagic fisheries species, is expected to experience warmer surface water temperatures, increased acidification, and increased variability in ENSO events, all of which will have direct effects on current patterns, ocean stratification, seawater chemistry, and productivity (Johnson *et al.* 2013).

Summary Table: Climate Change. See 4.0 Ecological Stressors in the Marine Environment for a detailed discussion of each stressor.

Activity/Source	Potential Jurisdictions	Stressors Direct	 Indirect
Climate change	● HI ● AS ● MI ● PRIA ● Pelagic	● Acidification ● Shift in productivity ● Thermal ● Sea level rise	● Salinity ● Irradiance ● Invasive species ● Disease ● Physical damage ● Sediment ● Nutrient inputs ● Hydrocarbons ● Herbicide/Pesticide ● Metals ● PCBs ● Ordnance† ● Endocrine disruptors

† Mariana Islands and Hawaii

3.2 Energy Production

With the desire to reduce fossil fuel usage and obtain energy independence, a considerable investment has been made to develop and assess the feasibility of alternative energy in the Pacific Islands. The jurisdictions in the Western Pacific Region have no fossil fuel resources, but energy can be obtained from wind, solar, ocean currents (hydrokinetic), ocean thermal, and geothermal means. It is no longer a question of whether alternative energy production will be implemented, but when. In the past decade, numerous utility-scale alternative energy projects have been proposed in the Hawaiian Islands, but only a handful have reached the construction stage. Hawaiʻi has committed to a long-term plan to convert entirely to renewable energy sources by 2050 (DOE 2015); the current proposal, called the Hawaiʻi Clean Energy Initiative, includes 31 types of activities whose specific projects could affect EFH. In American Samoa, an Energy Action Plan (Ness *et al.* 2016) proposes an array of renewable energy projects to be completed by 2020. One of those projects, converting the Island of Ta’u to 100% solar power generation (1.4 megawatts), was completed in 2016 (Heathman 2016). Both Guam (Conrad and Ness 2013a) and the CNMI (Conrad and Ness 2013b) have Energy Action Plans, but have yet to make significant progress in their implementation. Palmyra Atoll currently has a small research station (operated by The Nature Conservancy and the Palmyra Atoll Research Consortium) on its largest island that is powered by a combination of solar and wind power arrays, supported by a diesel generator. When assessing the potential effects on EFH, these renewable energy activities can be divided into two sub-categories: land-based and ocean-based energy activities.

Land-based energy projects include wind turbines, solar, geothermal facilities, and land-based Ocean Thermal Energy Conversion (OTEC). The stressors affected by the land-based portions of these projects would be similar to those found under land-based development/construction category. Some facilities, such as OTEC, require inwater intake and discharge structures which can contribute to direct effects on coastal and nearshore ecosystems. If energy produced through these projects remains on the island where it is generated, likely no additional effects to EFH would be expected, except for OTEC, which is discussed in more detail below. If energy is to be transferred to neighboring islands within an archipelago, the most practical transmission method would use submerged cables, either in surface or (more likely) buried conduits. Buried conduits would likely require removal or disturbance of the substratum, including coral reef, either through mechanical trenching, directional drilling, or a combination of the two.

Ocean-based energy projects include wind turbines and solar facilities placed on platforms in the ocean, and alternative energy approaches that use the physical (*e.g.*, wave or tidal energy) or thermal (*e.g.*, OTEC) properties of the ocean to generate power. Ocean-based energy projects require infrastructure, but it can be free floating or anchored to the bottom. Essential infrastructure feetures include power generating infrastructure and a means to transfer the generated energy to land. Proposals that have been considered in the Western Pacific Region include platform wind turbine farms, hydrokinetic generators (several designs are currently under testing off O‘ahu, Hawai‘i), and ocean-based OTEC. As with land-based projects, energy would be transferred to consumers via either surface or buried conduits.

The energy production potential for OTEC is considered to be much greater than for other ocean energy forms (Arvizu *et al.* 2011), and pilot projects have already been conducted in Hawai‘i. OTEC is considered an attractive and viable energy production method in the Pacific, but it presents specific challenges to EFH that do not occur with other alternative energy production methods. OTEC uses the temperature differential between cold deep and warmer surface waters to generate electricity. OTEC systems may be either closed-cycle or open-cycle. Closed-cycle OTEC uses refrigerants such as ammonia for powering the system’s generators, while open-cycle designs vaporize warm surface seawater in a low-pressure chamber and use it as the working fluid. As a by-product, OTEC produces cold, nutrient-rich water that is generally discharged back into the ocean.

3.3 Mining

Quarries are land-based mining locations that are present in most of the jurisdictions in the Western Pacific Region. Most quarry activity is dedicated to mining limestone for construction material, and likely has little effect on marine ecosystems, although they can potentially contribute to runoff. Unlike some other Pacific Islands (*e.g.*, Yap, Pohnpei, etc.), no direct mining of coral block/aggregate directly from living reefs occurs in the Western Pacific Region.

Currently, **deep ocean mining** is not economically viable on a large-scale, but continued advances in deep ocean mining technology and an increasing demand for rare earth elements (REE), will make it a realistic endeavor across the Pacific in the foreseeable future. Current deep ocean mining practices involve deploying remotely operated vehicles to locate prospective mine sites at depths between 1,400-3,700 m (4,200-8,100 ft) (Ahnert and Borowski 2000). Once a suitable site has been located, a mining ship or station is set up to mine the area (The Economist 2006) and one of two mineral extraction techniques are employed: 1) a continuous-line bucket system (CLB) and/or 2) a hydraulic suction system. The CLB system is the preferred technique and operates much like a conveyor-belt, running from the sea floor to the surface of the ocean

Summary Table: Energy Production. See 4.0 Ecological Stressors in the Marine Environment for a detailed discussion of each stressor.

Activity/Source	Potential Jurisdictions	Stressors	
		Direct	Indirect
Land-based Energy	● HI ● AS ● MI ● PRIA†	● Thermal ● Salinity ● FAD effect ● Physical damage ● Sediment	● Irradiance ● Noise ● Invasive species ● Sediment ● Nutrient inputs ● Hydrocarbons ● Herbicide/Pesticide ● Metals ● PCBs ● Ordnance†† ● Endocrine disruptors
Ocean-based Energy	● HI ● AS ● MI ● Pelagic	● Thermal ● Salinity ● Irradiance ● Invasive species ● FAD effect ● Physical damage ● Sediment ● Nutrient inputs ● Hydrocarbons ● Metals ● Ordnance†† ● Endocrine disruptors	● Noise

† Palmyra
†† Mariana Islands and Hawaii

where a ship or mining platform extracts the desired minerals from material collected by automated harvesters on the bottom, and discharges the tailings and deep ocean water back into the ocean (Nath and Sharma 2000). Hydraulic suction mining lowers a pipe to the seafloor and suction dredges material to the surface where it is processed to extract the desired minerals before a second pipe returns the tailings to the area of the mining site (Nath and Sharma 2000).

The International Seabed Authority (ISA), established as part of the United Nations Conventions on the Law of the Sea, regulates seabed mining in waters outside national jurisdictions, and grants exploration permits for projects. The U.S. is not a signatory to the Law of the Sea and not a party to the ISA. In 1980, Congress enacted the Deep Seabed Hard Mineral Resources Act (DSHMRA) under which U.S. citizens and corporations may apply to the Administrator of the National Oceanic and Atmospheric Administration (NOAA) for 10-year licenses to explore and

20-year permits to mine the deep seabed for hard mineral resources, and specifically REE (DSHMRA 1980). Within the EEZ of Hawaiʻi, commercial mining interests are subject to the Bureau of Ocean Energy Management's regulations governing non-energy mineral prospecting, leasing, and production. It is currently unclear under what authority deep ocean mining would be regulated in the territories, commonwealth or other administered areas outside of a designated Marine National Monument, National Wildlife Refuge, National Park or other such protected area, where mineral resource extraction is already prohibited.

Currently, U.S. mining licenses have been assigned in the mineral-rich Clarion-Clipperton Zone, roughly halfway between Hawaiʻi and Mexico. Additional licenses could be assigned to other mineral rich areas, which are often associated with natural hydrothermal vents. These vents regularly deposit rich concentrations of metals and minerals from the Earth's core to the ocean bottom. Hydrothermal regions are common off the Mariana Islands, and have been found off Hawaiʻi, which present potential opportunities for mineral extraction.

3.4 Land-based Aquaculture

An increasing world population requires a sustainable source of protein, and for many cultures, this has traditionally been derived through the direct harvest of marine organisms. To meet future protein needs, freshwater aquaculture and marine aquaculture (sometimes refer to as aquaculture and mariculture, respectively) will likely continue to expand and become important farming practices throughout the Pacific. In Hawaiʻi, aquaculture production has increased by more than 150% between 2011 and 2015 (DBEDT 2016). Likewise, increasing production has been seen in American Samoa and Guam since 2000 (Knomea 2016). "Fish farming" has a long cultural tradition in many parts of the Pacific (Keala *et al.* 2007), including Hawaiʻi where native

Summary Table: Mining. See 4.0 Ecological Stressors in the Marine Environment for a detailed discussion of each stressor.

		Stressors	
Activity/Source	**Potential Jurisdictions**	**Direct**	**Indirect**
Quarries	● HI ● AS ● MI		● Irradiance ● Sediment ● Nutrient inputs ● Hydrocarbons ● Metals ● PCBs ● Ordnance†† ● Endocrine disruptors
Deep Ocean	● HI ● AS ● MI ● PRIA† ● Pelagic	● Irradiance ● Noise ● Physical damage ● Sediment ● Nutrient inputs ● Hydrocarbons	

		● Metals

[†] Outside protected areas
[††] Mariana Islands and Hawai‘i

Hawaiians developed extensive coastal fishponds to grow species such as moi (*Polydactylus sexfilis*), āholehole (*Kuhlia sandvicensis*), and ‘ama‘ama (*Mugil cephalus*).

Until recently, land-based aquaculture was the primary commercial approach used to rear fish and shellfish, wherein tanks or ponds were placed directly on shore and stocked with desired species[1]. Water (fresh or salt) is pumped into the ponds, and wastewater effluent, is often returned to the nearshore waters, either passively via channels or actively via pumps. Alternative disposal methods, such as ground injection (HDOA 2011), or treatment using reverse osmosis (Qin *et al.* 2005) have been employed in the Western Pacific Region. Cultured organisms were fed to maximize their growth rate, and any excess feed, combined with excretory products would be flushed from the ponds, resulting in elevated nutrient levels in the receiving waters.

3.5 Development/Construction

Given the relatively small size of the islands in the Western Pacific Region, nearly all human development and construction occurs close enough to the coast to potentially affect EFH. Of particular concern are development projects that move earth, alter surface condition (*e.g.*, change ground permeability, erosion rates, etc.), or introduce potential contaminants. Many of these projects require local and/or federal permits and are likely to be subject to environmental review

Summary Table: Land-based Aquaculture. See 4.0 Ecological Stressors in the Marine Environment for a detailed discussion of each stressor.

	Potential	**Stressors**	
Activity/Source	**Jurisdictions**	**Direct**	**Indirect**
Land-based aquaculture	● HI ● AS ● MI	● Thermal ● Salinity ● Irradiance ● Invasive Species ● Disease ● FAD effect ● Sediment ● Nutrient inputs ● Herbicide/Pesticide ● Metals ● PCBs ● Ordnance[†]	● Hypoxia

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[1] In some cases, fish ponds and other support structures such as oyster racks, were placed in coastal waters. In addition, new approaches use anchored and free floating cages. These aquaculture practices and associated facilities will not be covered in this review; the WPRFMC is examining their effects elsewhere.

Activity/Source	Potential Jurisdictions	Stressors Direct	 Indirect
		● Endocrine disruptors	

† Mariana Islands

or other forms of disclosure that involve public and expert review (*e.g.*, NEPA, coastal zone management program, Clean Water Act, and/or the local equivalent).

Land-based development/construction activities include the majority of development projects in the Western Pacific Region, and are projects that have no direct connection with coastal waters, *i.e.*, are not water dependent. This includes the construction of most buildings and associated infrastructure, other structures (*e.g.*, energy production and transmission structures), and most roads, although see coastal roads below for a special case.

Coastal roads are a special case of land-based road construction in which part of the construction requires activities to occur in coastal waters and usually require some placement of fill. This may include construction of bridges, but also include coastal stabilization or hardening structures intended to fortify roads from erosion and/or inundation. In addition, other coastal hardening conducted independent of road construction (*e.g.*, shoreline stabilization, channelizing waterways, etc.) will have similar effects. With rising seas and other anticipated climate change effects, an increase in the number of construction and refurbishments of existing roads using coastal fortifications is expected, as well as an increase in other coastal hardening projects intended to protect shorelines from erosion and infrastructure from inundation.

Unlike land-based projects, waterbased development/construction has a direct connection or nexus with estuarine or marine ecosystems. These structures or projects are "water dependent" and thus cannot be built elsewhere. **Waterbased (dredging)** projects require the removal or addition of material into the waters of the U.S., and may include activities such as dredging to create or maintain navigational channels; trenching, blasting, pile driving, or drilling to install pilings, anchorings or other structures, or to bury conduits, pipelines, or other features; or the release of fill material to create breakwaters and other in-water stabilization/fortification structures. In contrast, **waterbased (non-dredging)** projects do not require dredging or filling, and may include installation of floating structures (*e.g.*, wave or wind turbines, etc.), and possibly construction of harbors or marinas, depending on their size and location.

Artificial reefs are a special case of waterbased construction and are highlighted separately from other waterbased activities due primarily to their designed purpose. These structures are specifically designed and constructed to enhance one or more marine services, and are generally considered to have net positive effects on the marine environment (although this is not always the case). Artificial reefs are often proposed as mitigation for adverse effects on marine ecosystems under federal permitting requirements such as the Clean Water Act. Regardless of their intended purpose and benefits, the placement and design of these features must be individually assessed for their effectiveness to enhance ecosystem services, as well as their potential to adversely affect EFH.

3.6 Shipping/Boating

Beyond the operation of a vessel itself, shipping/boating encompasses a wide variety of activities that could adversely affect marine ecosystems. Many of these activities and sources of stress are covered elsewhere in this report (*e.g.*, dredging and construction projects associated with harbors and safe navigation, marine debris, etc.). Not covered elsewhere are activities including the installation and maintenance of aids-to-navigation and large-scale anchorages, specifically the anchoring of prepositioning ships off the west coast of Saipan, CNMI.
Summary Table: Development/Construction. See 4.0 Ecological Stressors in the Marine Environment for a detailed discussion of each stressor.

	Potential	Stressors	
Activity/Source	**Jurisdictions**	**Direct**	**Indirect**
Land-based	● HI ● AS ● MI		● Thermal ● Salinity ● Irradiance ● Hypoxia ● Invasive Species ● Disease ● Sediment ● Nutrient inputs ● Hydrocarbons ● Herbicide/Pesticide ● Metals ● PCBs ● Ordnance†† ● Endocrine disruptors
Coastal Roads	● HI ● AS ● MI	● Irradiance ● Noise ● Invasive species ● Disease ● FAD effect ● Physical damage ● Sediment ● Nutrient inputs ● Hydrocarbons ● Herbicide/Pesticide ● Metals ● PCBs ● Ordnance†† ● Endocrine disruptors	● Irradiance ● Hypoxia ● Sediment ● Nutrient inputs ● PCBs ● Ordnance††
Waterbased (dredging)	● HI ● AS ● MI ● PRIA†	● Irradiance ● Noise ● Invasive species ● Disease	● Hypoxia

Activity/Source	Potential Jurisdictions	Stressors Direct	Stressors Indirect
		● FAD effect ● Physical damage ● Sediment ● Nutrient inputs ● Hydrocarbons ● Herbicide/Pesticide ● Metals ● PCBs ● Ordnance[††] ● Endocrine disruptors	
Waterbased (non-dredging)	● HI ● AS ● MI ● PRIA[†] ● Pelagic	● Noise ● Invasive species ● FAD effect ● Physical damage ● Hydrocarbons ● Herbicide/Pesticide ● Metals ● PCBs ● Ordnance[††] ● Endocrine disruptors	
Artificial reefs	● HI ● AS ● MI	● Invasive species ● FAD effect ● Physical damage ● Hydrocarbons	● Noise

[†] Palmyra
[††] Mariana Islands and Hawai‘i

Shipping is an essential activity in the Western Pacific Region, and is responsible for the transportation of nearly all imported goods. Maritime-based activities such as boat-based fishing and ocean tourism, are critical to island economies. Hawai‘i and Guam possess large U.S. military bases, from which naval activity and training are regularly conducted. Even for the PRIA, ships are the primary means for accessing the remote islands to conduct research and management activities.

Aids-to-navigation (ATONS) are "road signs" for ship crews and generally include a variety of buoys and beacons, each of which has a purpose to aid boaters in determining location, getting from one place to another, and staying out of danger. As such, ATONS are expected to have a net beneficial effect on EFH. These aids are securely anchored in the nearshore waters of all U.S. jurisdictions where shipping/boating occurs, although the PRIA are a notable exception (except for Palmyra, Wake Islands, and Johnston Islands which have ATONS).

Large-scale **anchorage** sites are rare in the jurisdictions of the Western Pacific Region, although the anchoring of military prepositioning ships off Saipan and military vessels in Apra Harbor are notable exceptions. The mission of these vessels is to quickly and efficiently deliver military cargo and supplies to a designated area in support of two Marine Expeditionary Brigades for up to 30 days and in response to a crisis or humanitarian disaster. Three to five vessels occupy the Garapan Anchorage as part of Maritime Prepositioning Ships Squadron-3 (MPSRON-3), and use large anchors with a considerable scope of heavy chain to hold their position. The vessels use pre-designated anchoring spots identified on NOAA nautical charts. Vessels have been observed to swing in an approximately 60-degree arc depending on the state of the winds and currents, dragging chain along the bottom (Rooney *et al.* 2005).

Summary Table: Shipping/Boating. See 4.0 Ecological Stressors in the Marine Environment for a detailed discussion of each stressor.

Activity/Source	**Potential Jurisdictions**	**Stressors**	
		Direct	**Indirect**
Shipping	● HI ● AS ● MI ● PRIA	● Noise ● Invasive species ● Disease ● FAD effect ● Physical damage ● Sediment ● Nutrient inputs ● Hydrocarbons ● Metals ● Endocrine disruptors	
ATONS	● HI ● AS ● MI ● PRIA[†]	● FAD effect ● Physical damage ● Hydrocarbons ● Metals ● Endocrine disruptors	
Anchorage	● MI[††]	● Noise ● Invasive species ● FAD effect ● Physical damage ● Hydrocarbons ● Metals ● Endocrine disruptors	

[†] Wake and Palmyra
[††] Saipan and Guam

3.7 Marine Debris

Marine debris is comprised of any persistent solid material that has been manufactured by humans and directly or indirectly, intentionally or unintentionally, disposed of or abandoned into the ocean. It can originate from land and be blown or transported via water into coastal waters or it can be directly disposed of into the ocean, generally from ships. Marine debris can include, but is not restricted to, derelict fishing gear, manufactured household and industrial items, metals, plastics, and microplastics. An estimated 4.8 to 12.7 million metric tons of marine debris entered the ocean in 2010 (Jambeck *et al.* 2015).

Once in the ocean, floating debris can be transported by wind and ocean currents thousands of kilometers (Erickson *et al.* 2014) before degrading, sinking, or washing up onto beaches. Due to the configuration of currents, marine debris often collects in specific regions of the ocean, usually referred to as “garbage patches” (NOAA 2011). Marine debris most often approaches islands from the windward side (Tetra Tech 2010), presenting added risk to marine ecosystems along those shores.

Floating debris poses a threat to pelagic animals and once it sinks, it can become entangled around benthic organisms. While ingestion rates may be high among sea turtles and marine mammals, it is considerably lower among fish, with documented ingestion limited to approximately 40 species worldwide, or less than one percent of all species (CBD 2012). Marine debris can serve as floatation and aid species dispersal (Gregory 2009, Donohoue *et al.* 2001). Recently, debris washed into the ocean from the 2011 tsunami in northern Japan has raised concerns for its potential to transport invasive species and contaminants (initial concerns associated with radioactivity have been found to be unwarranted [Smith *et al.* 2015]).

3.8 Other Human non-fishing Use

Humans use the marine environment in a variety of ways and for many purposes. Many of these activities have direct effects on EFH that are not included under other activities in this report. **Military training**, both land-based and ocean-based, is commonly conducted by all branches of the U.S. military throughout the jurisdictions of the Western Pacific Region. Troop and ship maneuvers, amphibious landings, weapons training, active use of sonar, missile launches, underwater demolitions, and coordinated maneuvers with multinational task forces are all important features of military training in the Pacific.

A wide range of civilian, non-fishing activities occur in the Pacific Islands, mostly involving **recreational use**, and including but not limited to scuba diving (and other similar activities), swimming, surfing, boating, and jet skiing. These activities are popular among local island residents and are an important part of the local tourist-based economies of most Wester Pacific jurisdictions.

Scientific research is actively conducted in most jurisdictions in the Western Pacific Region. Within the PRIA, it is likely the most prominent and common human use. Most scientific research has very low impact on the environment relative to the other activities included in this report, and the beneficial effects of scientific research likely outweigh these minimal effects.

Summary Table: Marine Debris. See 4.0 Ecological Stressors in the Marine Environment for a detailed discussion of each stressor.

	Potential	**Stressors**

Activity/Source	Jurisdictions	Direct	Indirect
Marine debris	● HI ● AS ● MI ● PRIA ● Pelagic	● Invasive species ● FAD effect ● Physical damage ● Hydrocarbons ● Herbicide/Pesticide ● Metals ● PCBs ● Endocrine disruptors	

However, sample collection and the installation of instrumentation has the potential to produce cumulative effects, especially if numerous research efforts are spatially and/or temporally concentrated.

3.9 Wastewater Discharge

Most terrestrial-derived "pollutants" are transported to and enter the nearshore ocean via water, whether it is the intentional disposal or through natural processes. For the purposes of this report, wastewater is defined as any water entering the ocean, via point source, groundwater, river system, or runoff that carries some pollutant (*e.g.*, sediment, chemicals, biological contaminants/ organisms) or has different physical properties (*e.g.*, different temperature or salinity) than the receiving body.

Summary Table: Other Human Non-fishing Use. See 4.0 Ecological Stressors in the Marine Environment for a detailed discussion of each stressor.

	Potential	Stressors	
Activity/Source	Jurisdictions	Direct	Indirect
Military training	● HI ● AS ● MI ● Pelagic	● Noise ● Invasive species ● Physical damage ● Nutrient inputs ● Hydrocarbons ● Metals ● PCBs ● Ordnance ● Endocrine disruptors	● Salinity ● Irradiance ● Sediment ● Nutrient inputs ● PCBs ● Ordnance ● Endocrine disruptors
Recreational use	● HI ● AS ● MI	● Noise ● Invasive species ● FAD effect ● Physical damage ● Nutrient inputs ● Hydrocarbons ● Herbicide/Pesticide ● Metals	● Sediment ● Endocrine disruptors

Activity/Source	Potential Jurisdictions	Stressors Direct	 Indirect
		● Endocrine disruptors	
Scientific research	● HI ● AS ● MI ● PRIA ● Pelagic	● Invasive species ● Disease ● FAD effect ● Physical damage ● Hydrocarbons ● Metals	

In the jurisdictions of the Western Pacific Region, effluent from primary and secondary **sewage** treatment plants often discharge directly into the nearshore waters via outfalls. Discharges may be in relatively shallow (~30 m) to deep (>80 m) water. Alternatively, treated effluent can be discharged into upland injection wells, where there is the potential for it to migrate into the groundwater and eventually find its way to the ocean through submarine groundwater discharge. Following large rainfall events, high volumes of stormwater can overburden treatment facilities and result in the discharge of untreated human sewage. Many island communities around the Pacific are not connected to municipal sewage treatment facilities, and rely on cesspools or septic tanks. Cesspools and septic systems are common in many rural and coastal areas of Hawaiʻi, American Samoa and the Mariana Islands (Southwest States and Pacific Islands Regional Water Program 2005). These are prone to leaking, allowing poorly or untreated human sewage to infiltrate into the groundwater, and in some locations, to enter coastal waters. Coastal septic and cesspool systems are particularly susceptible to sea level rise.

Intense or sustained rainfall can result in large discharges of **stormwater**, either through point sources such as stormwater pipes or via non-point sources such as runoff. High sheetwater flow rates can increase erosion and reduce the effectiveness of natural processes that filter pollutants from the stormwater prior to ocean entry. The volume and severity of stormwater discharges are directly related to the intensity, duration, frequency, timing, and/or scale of the rainfall event and the permeability of the surface. Low permeability, such as that associated with many land-based development/construction projects, often results in an increase in sheetwater flow.

Numerous **other activities** are responsible for discharges directly or indirectly into the nearshore marine waters. With some exceptions, agricultural fields (*e.g.*, sugar cane and other agriculture), taro loʻi, and animal lots (*e.g.*, piggeries in American Samoa) produce discharges that are currently excluded from U.S. Clean Water Act regulation, but can be significant sources of pollutants to coastal waters. Fish canning facilities, present in American Samoa, produce nutrient-rich effluent high in suspended solids and oils, whereas other large, managed landscapes, including golf course and residential developments, can be significant sources of nutrients and chemical contaminants, via non-point source runoff. While their point source discharges are regulated, sugar mills, power plants, and OTEC facilities dispose of wastewater from processing or cooling generators into the nearshore marine environment.

Summary Table: Wastewater Discharge. See 4.0 Ecological Stressors in the Marine Environment for a detailed discussion of each stressor.

	Potential	**Stressors**	
Activity/Source	**Jurisdictions**	**Direct**	**Indirect**
Sewage	● HI ● AS ● MI	● Thermal ● Salinity ● Irradiance ● Disease ● Sediment ● Nutrient inputs ● Hydrocarbons ● Herbicide/Pesticide ● Metals ● Endocrine disruptors	● Hypoxia
Stormwater	● HI ● AS ● MI	● Thermal ● Salinity ● Irradiance ● Disease ● Sediment ● Nutrient inputs ● Hydrocarbons ● Herbicide/Pesticide ● Metals ● PCBs ● Endocrine disruptors	● Hypoxia
Other discharges	● HI ● AS ● MI	● Thermal ● Salinity ● Irradiance ● Disease ● Sediment ● Nutrient inputs ● Hydrocarbons ● Herbicide/Pesticide ● Metals ● Endocrine disruptors	● Hypoxia

4.0 Ecological Stressors in the Marine Environment

Ecological stressors are factors that alter the productivity, fitness, and the survival of organisms, and/or affect the long-term persistence and the functional and structural capacity of populations, biological assemblages, or ecosystems. Sources of ecological stress can come from natural environmental events such as storms, or may result directly or indirectly from human activities (Table 2). Some ecological stressors act at a relatively small spatial scale, whereas others are regional or global in effect.

At any particular time and place, organisms are exposed to a complex regime of interacting ecological stressors. In some instances, the exposure to a given stressor is intense, but of short duration (*e.g.*, a storm-driven flood event, a ship grounding). In other instances, exposure may be chronic and relatively unchanging over time (*e.g.*, sewage discharge, nutrient input via groundwater). The complex interactions among stressors, and across their ranges of exposure, are what determine the potential effects on organisms and ecosystems.

Stressors create challenges to the integrity and quality of ecosystems, and by extension, the EFH to which those ecosystems are a component. When exposure to environmental stressors changes in intensity, duration, frequency, timing, and/or scale, organisms and/or ecosystems will undergo an ecological response. For example, disruption of an ecosystem by an intense disturbance could cause the mortality of specific organisms and other ecological damage, followed by a gradual recovery driven by natural processes (*e.g.*, succession). Species and ecosystems have some inherent capacity to tolerate changes to the intensity of stressors, but there are limits to this ability, which are often represented as tolerance thresholds. When these thresholds are exceeded, substantial ecological change may occur, often causing adverse effect to EFH.

Fifteen potential stressors on EFH (Table 2) have been identified for this report, and their effects on the ecosystems within the Western Pacific Region are discussed in greater detail below. These stressors (in bold) have been grouped into the following broad categories:

1. *Environmental stressors* are associated with excessive or insufficient physical or chemical conditions within the marine environment. Environmental stressors can be associated with water temperature, solar radiation, salinity, pH, dissolved oxygen, and any combinations of these, and in this report, include: **Ocean acidification, Shifts in productivity, Thermal, Salinity, Irradiance, Noise,** and **Hypoxia.**

2. *Biological stressors* are associated with interactions among organisms of the same or different species. Biological stressors can result from competition, herbivory, predation, parasitism, and disease, and in this report, include: **Invasive species, Disease,** and **Fish Aggregating Device (FAD) effect.**

Table 2. The potential stressors associated with non-fishing activities and sources of stress. Activity categories (rows) are discussed in detail in the text. Stressors are groups into five general types: environmental (blue), biological (red), physical (green), chemical (purple), and sea level rise (orange). D=activity directly affects the stressor, i=activity indirectly affect the stressor, *=may be a problem in some jurisdictions.

	🟦	🟦	🟦	🟦	🟦	🟦	🟦	🟥	🟥	🟥	🟩	🟪	🟪	🟪	🟪	🟪	🟪	🟪	🟪	🟧
														Chemicals						
	Ocean Acidification	Shift in Productivity	Thermal	Salinity	Irradiance	Noise	Hypoxia	Invasive Species	Disease	FAD Effect	Physical Damage	Sediment	Nutrient Inputs	Hydrocarbons	Herbicide/Pesticide	Metals	PCBs	Ordnance	Endocrine Disruptors	Sea Level Rise
Climate Change	D	D	D	i	i			i	i		i	i	i	i	i	i	i	*	i	D
Energy Production																				
Landbased			D	D	i	i		i		D	D	iD	i	i	i	i	i	*	i	
Waterbased			D	D	D	i		D		D	D	D	D	D		D		*	D	
Mining																				
Quarries					i							i	i	i		i	i	*	i	
Deep Ocean					D	D					D	D		D		D				
Land-based Aquaculture			D	D	D		i	D	D	D		D	D		D	D	D	D	D	
Development/Construction																				

Landbased			i	i	i		i	i	i			i	i	i	i	i	i	*	i	
Coastal roads					iD	D	i	D	D	D	D	iD	iD	D	D	D	iD	*	D	
Waterbased-Dredging					D	D	i	D	D	D	D	D	D	D	D	D	D	*	D	
Waterbased-Non-dredging						D		D		D	D			D	D	D	D	*	D	
Artificial reefs						i		D		D	D			D						
Shipping/Boating																				
Shipping						D		D	D	D	D	D	D	D		D			D	
ATONs										D	D			D		D			D	
Anchorages						D				D	D			D		D			D	
Marine Debris								D		D	D			D	D	D	D		D	
Non-fishing Human Uses																				
Military training				i	i	D		D			D	i	iD	D		D	iD	*	iD	
Recreational use						D		D		D	D	i	D	D	D				iD	
Research								D	D	D	D			D		D				
Wastewater Discharge																				
Sewage			D	D	D		i		D			D	D	D	D	D			D	
Stormwater			D	D	D		i		D			D	D	D	D	D	D		D	
Other activities			D	D	D		i		D			D	D	D	D	D			D	

3. *Physical stressors* are associated with changes in exposure to kinetic energy. This type of ecological disturbance is often acute and episodic, and in this report, include: **Physical damage.**

4. *Pollution stressors* occur when chemicals or other contaminants are present in concentrations large enough to affect organisms and thereby cause ecological change. Pollution can include anthropogenic inputs of pesticides/herbicides, hydrocarbons, metals, and other toxic chemicals, but also can include inputs of sediment and nutrients. This report includes: **Sediment, Chemicals,** and **Nutrient inputs.**

5. ***Sea level rise*** is a unique marine stressor with important implications in the Western Pacific Region. On casual examination, sea level rise alone might appear to be unimportant to subtidal marine ecosystems, but it is a significant direct threat to intertidal and mangrove ecosystems. Additionally, it acts indirectly on other ecosystems through often synergistic interactions with other stressors (see 5.0 Cumulative Effects).

4.1 Environmental Stresses

4.1.1 Ocean Acidification

Ocean acidification is the decrease in the pH of the oceans caused by the uptake of atmospheric carbon dioxide (CO_2) (Caldiera and Wickett 2003). Seawater is slightly basic (pH ~8.2) and acidification shifts it towards a less basic condition, *i.e.*, lower pH. Equally important, acidification decreases the carbonate concentration in seawater, and thus decreases the saturation state of calcium carbonate ($CaCO_3$) (Orr *et al.* 2005, Kleypas *et al.* 2006, Cooley and Doney 2009). This change in the chemical make-up of seawater can directly affect the biological process of calcification, essential for reef-building organisms, mollusks, echinoderms, and many types of plankton.

Over the past two centuries, atmospheric CO_2 has increased by over 43%, from pre-industrial levels of approximately 280 parts per million (ppm) (IPCC2007) to over 400 ppm in 2016 (NOAA 2016), and under "business-as-usual" models which assume continued greenhouse gas emissions at or exceeding current rates, atmospheric CO_2 could exceed 1,000 ppm by the end of the century (Kiehl 2011). This rate of CO_2 increase is driven primarily by human burning of fossil fuels and deforestation (Doney & Schimel 2007), and the current concentration of CO_2 is higher than that experienced on Earth for at least the past 800,000 years (Lüthi *et al.* 2008). Rising atmospheric CO_2 is tempered by oceanic uptake, which can absorb up nearly a third of the anthropogenic carbon added to the atmosphere (Sabine and Feely 2007, Sabine *et al.* 2004).

At the Hawaiʻi Ocean Time-Series (HOT) station ALOHA, the rate of increase of surface water CO_2 and atmospheric CO_2 are strongly correlated (Takahashi *et al.* 2006, Dore *et al.* 2009), indicating uptake of anthropogenic CO_2 is the primary cause of long-term decreases in pH and $CaCO_3$ saturation state. Since preindustrial times, the average ocean surface water (the ocean layer down to approximately 100 m) pH has fallen by approximately 0.1 pH units, from approximately 8.21 to 8.10 (Royal Society 2005) which is due to the logarithmic nature of the pH scale represents about a 30% increase in acidity (Caldiera and Wickett 2003). Buoy data from the equatorial Pacific (covering years 1997-2011) show pH ranged from 7.91-8.12 (Sutton *et al.* 2014), which is consistent with what has been observed in subtropical waters (pH = 8.06-8.14) via the HOT station ALOHA time series (Dore *et al.* 2009). Acidity is expected to decrease to7.88 pH units if the atmospheric CO_2 concentration reaches 1,000 ppm (IPCC 2007), although more current projections suggest pH might be lower under this business-as-usual model (IPCC 2014). Even under modest, likely-to-be-obtained climate change predictions (CO_2 = 560 ppm), oceanic pH is expected to be 7.92 pH units (IPCC 2014), and deep ocean waters and arctic surface waters are expected to be undersaturated ($CaCO_3$ saturation state <1). At pH 7.8, major ecological changes will occur because of the impairment of invertebrate reproduction (Wood *et al.* 2008, Wang *et al.* 2016) and recruitment (Nakamura *et al.* 2011), and shell dissolution of many benthic and planktonic invertebrate taxa (Smith & Buddemeier 1992, Kleypas *et al.* 1999, Hall-Spencer *et al.* 2008, Cooley and Doney 2009). Additionally, acidification will affect biological processes beyond calcification, including gene expression, metabolism, and cell death/regeneration (Kleypas *et al.* 2006, Todgham and Hoffman 2009). Already seasonal acidification events are appearing in upwelled waters along the California coastline in summer, decades earlier than models predict (Feely *et al.* 2008, Gruber *et al.* 2012).

However, the effect of ocean acidification on calcification is complicated by the fact that enhanced levels of CO_2 can increase photosynthetic rates (Behrenfeld *et al.* 2006, Kranz *et al.* 2009), which will affect net primary productivity (Hein and Sand-Jensen 1997, Behrenfeld *et al.* 2006, Jiao *et al.* 2010). In corals, much evidence suggests that under normal conditions, calcification rates generally rise proportionally with increases in rates of primary production, both at the colony and assemblage scale (Gattuso *et al.* 1999), yet in virtually all studies that have measured both photosynthesis and calcification in corals, any stimulation of photosynthesis by increased CO_2 was accompanied by a decrease, rather than an increase, in calcification (Reynaud *et al.* 2003). In Hawai‘i, Langdon and Atkinson (2005) exposed an assemblage of corals (*Porites compressa* and *Montipora capitata*) to two levels of CO_2, and at the higher CO_2 level, observed a 22–26% increase in the rate of net primary production but a 44–80% decrease in calcification, depending on the species and the time of year.

Furthermore, calcification rates in the wild are affected by other stressors such as temperature, light levels, and the availability of trace minerals and nutrients, and several studies have illustrated a complicated relationship between calcification (which affects photosynthesis), and the interactions among ocean acidification and these other stressors. For example, light intensity was shown to be an important factor in laboratory experiments with marine foraminifera, where calcification rates decreased with increasing CO_2 concentrations only under saturating light intensities (Zondervan *et al.* 2002). Trace metal limitation has been shown to affect marine foraminifera calcification and growth (Schulz *et al.* 2004), and iron limitation affected both calcification and productivity, while zinc was limiting to productivity, but not calcification.

Under the “business-as-usual” climate change scenarios, temperate and colder oceans are expected to become undersaturated in both calcite and the more bio-available aragonite (Orr *et al.* 2005), but the warm surface waters of the tropics and subtropics are not expected to become undersaturated over the range of these projected conditions (Fabry *et al.* 2008), except perhaps in some upwelling regions. In these areas aragonite undersaturated waters are pushed upward from the deep ocean into shallower water—a phenomenon frequently referred to as the "shoaling of aragonite saturation horizons”—where it would now impinge on the depth ranges of pelagic animals (Feely *et al.* 2004). Even though tropical surface waters are not expected to become undersaturated, the average aragonite saturation state under “business-as-usual” climate models is expected to be about half its current state in the tropical Pacific (Fabry *et al.* 2008), leading to significantly lower calcification rates.

Reduced calcification rates have been observed following acidification for a variety of calcareous organisms even when aragonite or calcite saturation state is > 1 (Royal Society 2005, Kleypas *et al.* 2006, Fabry *et al.* 2008). Some reef-building corals appear to cease calcification at aragonite saturation state as high as two, but the degree of sensitivity varies among species, and some marine taxa may even show enhanced calcification at elevated CO_2 levels (Iglesias-Rodríguez *et al.* 2008, Ries *et al.* 2009). However, studies of ocean acidification on calcification rates of marine organisms exist for a limited number of species, and we lack sufficient understanding of calcification mechanisms to explain species-specific differences (Doney *et al.* 2009). Regardless, the evidence suggests calcification rates will be significantly reduced for most marine organisms.

Currently, most studies examining the effect of ocean acidification on marine organisms have been of short duration, ranging from hours to weeks. Chronic exposure to increased acidification may have complex effects on the growth and reproductive success of calcifying organisms, and could induce adaptations that are not observed in short-term experiments (Kleypas *et al.* 2006, Doney *et al.* 2009).

Almost every study published to date confirms that calcification rates will decrease in response to decreasing aragonite saturation state and decreasing pH for corals (Gattuso *et al.* 1998, Langdon *et al.* 2000, Marubini & Atkinson 1999, Marubini & Davies 1996), coral reef communities (Langdon *et al.* 2000, 2005, Leclercq *et al.* 2000), and planktonic organisms (Bijma 1991, Riebesell *et al.* 2000). Additionally, in coral reef ecosystems, many other benthic calcifying taxa are ecologically important. Crustose coralline algae (CCA) are a widespread, globally-significant, but often undervalued, benthic marine organism (Foster 2001). CCA have shown declines in both calcification rates and recruitment rates at lower carbonate saturation state (Doropoulos *et al.* 2012), including in Hawai‘i (Kuffner *et al.* 2008). This could have significant cascading effect through the coral reef ecosystem because CCA is an important structure-consolidating organism and a key settlement substratum for many corals. Under lower pH conditions, changes in CCA structure has significantly lowered the settlement density of coral larvae (Doropoulos *et al.* 2012).

Coral reef ecosystems are defined by their ability to produce a net surplus of $CaCO_3$ that produces the topographically complex reef structure necessary to support high marine biodiversity and biomass. Coral reef ecosystems have survived around many Pacific Islands because of their rapid accretion rates, giving them the ability to migrate upward and maintain themselves at a depth that has at least the minimum light levels required for continued growth. Under increasing ocean acidification, coral calcification rates will decrease, and dissolution rates will increase (Langdon *et al.* 2000, Yates and Halley 2006), particularly for those reefs at higher latitudes where seawater saturation state is expected to be closer to an undersaturated state. These reefs are already near the limit for reef growth, and will be further challenged by undersaturated seawater conditions. Interestingly, even though global warming may extend ocean water temperatures conducive to coral survival to higher latitudes, the decrease in reef $CaCO_3$ accretion expected at higher latitudes may restrict reef development to lower latitudes where aragonite saturation levels can support carbonate accumulation (Guinotte *et al.* 2003, Kleypas *et al.* 2001).

Even if calcification continues, reduced rates may impair the ability of calcifying organisms to compete with non-calcifying ones. Such a decrease has been observed in CCA assemblages when exposed to high-CO_2 conditions (Kuffner *et al.* 2008). Given that many taxa appear to exhibit species-specific responses (Fabry 2008, Ries *et al.* 2009, Doropoulos *et al.* 2012), assemblage- and ecosystem-level effects are likely to be complicated and difficult to predict, but are likely to result in major reorganizations of benthic and planktonic assemblages. These alterations will likely affect the physical and chemical structure of reefs. Topographical structure is a key ecological function strongly correlated with biodiversity, abundance, and biomass (Alvarez-Filip *et al.* 2009), and has direct implications on food webs dynamics.

Calcareous skeletal parts are widespread among many groups of benthic invertebrates and studies have reported drops in calcification rates at CO_2 levels below those expected under the current “business-as-usual” models for common species of mussels (*Mytilus edulis*) and oysters (*Crassostrea gigas*), a Pacific conch (*Strombus luhuanus*) and numerous species of sea urchin (Shirayama and Thorton 2005, Dupont *et al.* 2010), many of which occur in the Western Pacific Region. However, these findings cannot be easily generalized across taxa (Kroeker *et al.* 2014); many urchins and crustaceans show surprising resistance to low pH (Hendricks and Duarte 2010, Dupont *et al.* 2010, Kroeker *et al.* 2014), and calcification rates in the arms of a burrowing brittle star increased when they were grown in low pH water (Wood *et al.* 2008), but this finding is complicated in that while brittle stars experienced increased calcification, they also experienced decreased muscle mass in the arms, which would reduce arm movement and likely decrease respiration and feeding, suggesting that over the long-term, the organism would experience a reduction in fitness, highlighting the potential sub-lethal effects that can occur in seemingly resistant taxa (Dupont and Thorndyke 2013).

The effects of acidification may be exacerbated by certain developmental bottlenecks that are affected by low pH, and thus may have a disproportionately large influence on population dynamics that are missed by most experimental investigation (Dupont *et al.* 2010, although see Hendricks and Duarte 2010). The response of early developmental stages of invertebrates to ocean acidification has been investigated across a range of species, including bivalves and sea urchins. Under increasing acidification, sea urchins show reduced fertilization success, developmental rates, larval size, metamorphosis, spicule formation, and in their ability to settle (Kurihara and Shirayama 2004, Dupont *et al.* 2010; Evans and Watson-Wynn 2014). Likewise, developmental abnormalities have been observed in the oyster *C. gigas*, after 24 hours of exposure to high CO_2 levels (>2,000 ppm) and 80% of the larvae displayed malformed shells or remained unmineralized (Kurihara *et al.* 2007). Less dramatic, but still significant, effects have been observed at lower CO_2 levels, and even short exposure at the fertilization stage can carry over into later stage larvae, affecting growth rates and calcification (Barton *et al.* 2012). Greater susceptibility to increased acidification of larval and juvenile compared to adult mollusks is a pattern observed across a range of mollusks that have been studied (Kroeker *et al.* 2013).

In general, marine fish appear to be relatively tolerant to mild increases in CO_2 (Munday 2011a, Kroeker *et al.* 2014). Otolith development is unaffected by moderate increases in acidity (Munday *et al.* 2011b), although sublethal metabolic effects have been identified for some reef fish species (Munday *et al.* 2009). The most significant effects may occur through cellular changes that block olfactory senses, and consequently the ability of adults and juveniles to detect predators (Dixson *et al.* 2010; Munday *et al.* 2013; Heuer and Grosell 2014), and possibly to locate suitable settlement habitat (Dixson *et al.* 2008), which under some ecological conditions could have significant adverse effects on a population.

Deepwater corals in the Western Pacific Region are slow growing and long lived (Roark *et al.* 2006). Their carbonate structure serves as important habitat for many deep sea species and support high biodiversity of invertebrates (Parrish and Baco 2007). The maximum depth of deep water corals and their associated species appears to coincide with the depth of the aragonite saturation state horizon (Guinotte *et al.* 2006), which under the “business-as-usual” climate models is expected to shoal. As such, these deepwater coral systems are expected to be the first to experience a shift to an undersaturated seawater condition (Doney *et al.* 2009). This will likely lead to range/depth contractions, and could force slow-growing deepwater corals into direct competition with shallow water coral species, which are likely superior competitors.

The effects of elevated CO_2 and ocean acidification on primary productivity are complicated by the relationship between carbon uptake (as part of the photosynthetic process), temperature, calcification (where relevant), and nutrient availability. A potentially major consequence of ocean acidification will be significant changes in the inorganic and organic chemistry of seawater. Affected chemical species include biologically important elements such as boron, phosphorus, silicon, and nitrogen, as well as trace elements such as iron, zinc, vanadium, arsenic, and chromium (Doney *et al.* 2009). Concentrations of phosphate, silicate, fluoride, and ammonia species will decrease with increasing acidification (Zeebe and Wolf-Gladrow 2001), and will have far-reaching implications for phytoplankton and other ecological processes. Additionally, many trace elements (*e.g.*, aluminum, iron, chromium, etc.) show reduced bioavailability to organisms as result of hydrolyzation under increasing acidification. The overall effect of ocean acidification on the structure and function of these biologically important compounds is largely unknown, making predicting organismal and ecosystem effects difficult.

Seagrasses show a consistent and dramatic increase in light-saturated photosynthetic rates with increasing acidification (Zimmerman *et al.* 1997, Short and Neckles 1999, Invers *et al.* 2001), although it is possible these benefits could be offset by the negative effects of increased temperature on vegetative growth (Ehlers *et al.* 2008). Interestingly, regions near natural subsurface volcanic CO_2 vents in the Mediterranean Sea showed a marked absence of reef-building corals and reduced abundance of sea urchins, coralline algae, foraminifera, and gastropods. Instead, the benthos was dominated by sea grass, anemones, and non-native invasive algal species (Hall-Spencer *et al.* 2008), consistent with expectations from laboratory experiments.

The mangrove trees *Rhizophora mangle* showed increase photosynthesis under elevated CO_2 levels (Farnsworth *et al.* 1996), but this appears to be mediated by salinity. Trees grown under elevated CO_2 experienced little growth enhancement in high-salinity conditions, but more growth enhancement under low-salinity conditions (Ball *et al.* 1997), an effect that was magnified for less-tolerant species (Ball *et al.* 1997). Likewise, little effect on mangrove seedling growth or survival was found for three species in different mangrove genera when grown under highly acidic conditions (pH=5.0) (Rozainah *et al.* 2016), suggesting that mangrove trees will experience few adverse effects from CO_2 condition expected under "business-as-usual" climate models.

Most studies on the effect of ocean acidification on the calcification rates of non-larval planktonic organisms have focused on coccolithophores (a common tropical planktonic group), and have found inconsistent responses to acidified seawater. The bloom-forming coccolithophore species, *Emiliania huxleyi* and *Gephyrocapsa oceanica,* showed a 25-66% decrease in calcification rate when CO_2 was increased to 560–840 ppm (Riebesell *et al.* 2000, Zondervan *et al.* 2001, Zondervan *et al.* 2002, Sciandra *et al.* 2003, Delille *et al.* 2005, Engel *et al.* 2005). In contrast, other coccolithophore species have exhibited no significant change in calcification or malformations from being cultured in acidified seawater.

In laboratory experiments under conditions of 560 and 740 ppm CO_2, the shell mass of two foraminifera species (*Orbulina universa* and *Globigerinoides sacculifer*) decreased by four to 14% compared with preindustrial CO_2 controls. Finally, the sub-arctic pteropod *Clio pyramidata* showed net shell dissolution in the living organisms when the aragonite saturation state reached <1 (Orr *et al.* 2005, Fabry *et al.* 2008), a level expected to occur over the range of this species under the current "business-as-usual" models.

Most marine phytoplankton tested in single-species laboratory studies and field population experiments showed little change in photosynthetic rates under CO_2 conditions equivalent to ~760 ppm (Tortell *et al.* 1997, Hein and Sand-Jensen 1997, Burkhardt *et al.* 2001, Tortell and Morell 2002, Rost *et al.* 2003, Beardall and Raven 2004, Giordano *et al.* 2005, Martin and Tortell 2006). In contrast, a phytoplankton assemblage dominated by diatoms and coccolithophores showed nearly a 40% increase in carbon uptake at CO_2 levels consistent with the "business-as-usual" climate models (Riebesell *et al.* 2007) indicating increased photosynthesis. Whether species show increased rates of photosynthesis with progressive oceanic uptake of atmospheric CO_2 may depend on nutrient and trace metal availability, light conditions, and temperature. Extrapolating current experimental results to ocean regions presents significant challenges because the ocean warming that accompanies acidification increases stratification of the upper ocean, thereby reducing the upwelling of nutrients, which contributes to decreased phytoplankton biomass and productivity on a global scale (Behrenfeld *et al.* 2006). What is clear is that the species diversity and the composition of phytoplankton assemblages are likely to change, with some species facing a high probability of extinction. The potential for this change at the base of the food web to cascade upward through multiple trophic levels will directly depend on the dietary specialization of secondary and tertiary consumers. However, the potential for severe adverse effects throughout marine food webs is significant and particularly difficult to predict based on available information.

As with other plankton, the effect of ocean acidification on larval fishes appears to be highly variable. Potential effects include reduced growth and survival (Baumann *et al.* 2011), skeletal deformation (Pimentel *et al.* 2014), altered neurological function (Nilsson *et al.* 2012), altered otolith (ear stone) development (Checkley *et al.* 2009, Munday *et al.* 2011b, Hurst *et al.* 2012, Bignami *et al.* 2013), impaired tissue health (Frommel *et al.* 2011), and disrupted behavior (Munday *et al.* 2010, Ferrari *et al.* 2012, Hamilton *et al.* 2014). In contrast, several other studies reported no significant effects of ocean acidification on fish larvae (*e.g.*, Munday *et al.* 2011a, Frommel *et al.* 2013, Bignami *et al.* 2014), illustrating the variability in potential effects.

What is clear is that calcification in marine plankton will be adversely affected when surface waters become undersaturated. While the aragonite saturation state in tropical surface waters is not expected to drop below one under the current “business-as-usual” climate models, saturation state in deeper water layers is expected to be <1 and will likely affect the depth at which plankton can exist without experiencing shell demineralization (Orr *et al.* 2005). This will result in a contraction of marine phytoplankton ranges to shallower depths and lower latitudes. Unfortunately, predicting, and even detecting, such acidification-driven population shifts presents a significant challenge because of a lack of baseline data on the current distributions and abundances of most plankton species.

4.1.2 Shifts in Productivity

Open ocean productivity refers to the production of organic matter through the process of photosynthesis by phytoplankton (primary productivity) and the further production through the consumption and growth of non-photosynthetic heteroplankton (secondary productivity) suspended in the water column (Sigman and Hain 2012). Although productivity is the result of biological activity and the organisms responsible for it are subjected to many of the stressors described in this report, this report considers open ocean productivity as an environmental stressor because the location, diversity, abundance and biomass of pelagic assemblages, including important fishery species, are directly dependent on the amount of productivity in an area (Pauly and Christensen 1995, Chassot *et al.* 2010). Changes in the spatial distribution and amount of open ocean productivity are potentially among the most important non-fishing factors affecting all marine ecosystems, pelagic or benthic, and nearshore or open ocean.

In addition, this report treats open ocean productivity separately from nearshore productivity because the stressors affecting open ocean productivity tend to be regional, basin, or global in scale, all of which lack a strong local terrestrial component (although terrestrial inputs can be important via atmospheric deposition).

In addition to sunlight, phytoplankton require a suite of chemicals with which to grow and conduct photosynthesis, including nitrogen, phosphorous, iron, silicate, $CaCO_3$, and a variety of trace metals (Sigman and Hain 2012). Limitations in the availability of these requirements limit the amount of primary, and by extension secondary, productivity in a region of the ocean.

Open ocean productivity in the tropical Pacific is primarily associated with regions of upwelling, where nutrient-rich, deep-ocean water is brought to the surface. In regions without upwelling, thermal stratification creates a warm, nutrient-poor, or oligotrophic, surface layer (due to a lack

Summary Stressor Table: Potential effects of ocean acidification
Color reflects the relative severity of an adverse effect compared across all stressors: green=none to mild, yellow=moderate, red=severe.

Ecosystem	**Potential Effects**
Intertidal	● Decreased diversity ● Decreased survival of planktonic larval stages of important herbivorous and sessile invertebrates (*e.g.*, urchins, nerites) ● Increased algal photosynthetic activity, potential for a phase shift toward algal-dominated shoreline
Mangrove Forests	● Increased photosynthetic and growth rates for mangroves and other primary producers, but may depend on salinity ● Decreased abundance of calcifying organism ● Decreased survival of planktonic larval stages
Seagrass Beds	● Increased photosynthetic rates and primary productivity ● Denser seagrass beds, although vegetative growth may be tempered by increasing seawater temperature ● Decreased abundance of calcifying organism ● Decreased survival of planktonic larval stage
Coral Reefs	● Reduced calcification rates in reef-building organisms, including corals and coralline algae. ● Increased algal photosynthesis and growth ● Reduced calcification and survival of potentially important invertebrate grazers (*e.g.*, urchins) ● "Flattening" of reef structure leading to loss of species diversity, including important fishery species ● Potential for a phase-shift toward algal-dominated assemblage
Deep Reef Slopes	● Drop in aragonite saturation state <1 under "business-as-usual" climate change predictions ● Dissolution of calcifying organisms ● "Shoaling" of range distributions, potentially leading to increased competitive interactions with shallow-water species ● Extirpation of species likely ● Decreased diversity (including fishery species) associated with loss of structure-producing organisms ● Decreased survival of planktonic larval stages
Banks and Seamounts	See *Coral Reefs* and *Deep Reef Slopes*

Ecosystem	**Potential Effects**
Deep Ocean Floor	- Drop in aragonite saturation state <1 under "business-as-usual" climate change predictions - Dissolution of calcifying organisms - Extirpation of species is likely - Decreased diversity (including fishery species) associated with loss of structure-producing organisms
Pelagic Environment	- Increased photosynthesis in phytoplankton, but mediated by nutrients and trace minerals - Decreased abundance of calcifying organism - Decreased survival of planktonic larval stages - Shifts in species composition, which has potential to disrupt food web dynamics

of mixing with deeper layers) where both primary and secondary productivity are limited. Therefore, regions of productivity are strongly affected by oceanographic processes that alter the position and strength of upwelling. These oceanographic processes are usually the result of basin- or global-scale climatic events. Basin-scale events, including "short" duration ENSO events and longer duration PDO events, result in the shifting of surface water masses of differing temperature, which alters ocean stratification and moves the location of upwelling. At the global scale, climate change is expected to permanently change the amount, location, and quality of productivity.

In general, changing climate is likely to increase vertical stratification, reducing the upward flow of nutrients and lowering both primary (Falkowski *et al.* 1998, Behrenfeld *et al.* 2006, Toseland *et al.* 2013) and secondary (Roemmich and McGowan 1995) productivity. This effect is predicted to be most pronounced in the tropical oceans, including the Western Pacific Region. A six percent reduction in global oceanic primary production has already been observed between the early 1980s and the late 1990s (Gregg *et al.* 2003), and extrapolating into the future, suggests that marine biological productivity in the tropics and mid-latitudes will decline substantially (Cochrane *et al.* 2009). Both statistical and coupled biogeochemical models (Lehodey 2001, Lehodey *et al.*2003) have predicted the slowdown of Pacific meridional overturning circulation and a subsequent decrease of equatorial upwelling, which has been attributed as the cause of the primary production and biomass decrease over the past 40 years (McPhaden and Zhang 2002).

Changes in secondary productivity are likely to be linked closely with changes in primary productivity in the Western Pacific Region, and effects on tropical zooplankton are likely to be more pronounced than those already being observed at higher latitudes. The more heat-tolerant, low-latitude species might be more vulnerable to climate change stressors than less heat-tolerant species because they may live closer to their physiological limits (Tomanek and Somero 1999, Stillman 2002).

An increased in primary productivity has the potential to increase particulate organic matter (POM). Zooplankton, which consume phytoplankton, usually experience a time lag before they can respond to the increase in primary productivity. During this time lag, POM will be exported from the surface waters to the deep waters, where microbial assemblages will recycle it. This process consumes oxygen and can result in hypoxia in deep waters (see
), creating what have been called “dead zones.”

Currently, it is unclear how climate change will affect ENSO and PDO events in the Western Pacific Region (IPCC 2013). Climate change is expected to weaken tropical easterly trade winds, warm the surface ocean, and intensify the subsurface thermocline. ENSO variability is controlled by a delicate balance of competing feedbacks, and it is likely that one or more of the major physical processes that are responsible for determining the characteristics of ENSO will be modified by climate change (Collins *et al.* 2010). Unfortunately, our current understanding of ENSO variability does not make it possible to predict the potential changes that could occur (IPCC 2013). The WPWP, an immense region of warm water along whose eastern edge strong upwelling occurs, is likewise affected by ENSO events. The upwelling region is important to several species of tuna. During ENSO events, the eastern edge, and thus the region of high productivity can shift as much as 4,000 kilometers (km) eastward as a result of weakened easterly trade winds (Lehodey *et al.* 1997). Likewise, it is not clear how climate change stressors will affect the WPWP, but an effect is expected to cause a significant shift in both the amount and location of high productivity areas, which will result in concomitant shifts in pelagic assemblages, including important fishery species.

4.1.3 Thermal

Thermal stress occurs when the temperature of the environment changes such that it can disrupt the normal biological activity of an organism or the processes and/or function of an ecosystem. In the ocean, thermal stress is often associated with increased temperature of the water, but does not necessarily need to be the result of warming; a decrease in water temperature can be a source of thermal stress. Likewise, most current discussion and research of thermal stress has been focused around regional or global processes (*e.g.*, climate change, ENSO events, etc.), but thermal stress can occur at smaller scales (*e.g.*, a discharge for a power plant or OTEC facility). Regardless of the scale, the results of "climate change studies" that examine thermal effects are still relevant when assessing the potential adverse effects of a small-scale thermal stress event.

In the marine environment, much focus has been placed on the large-scale or global effect of climate change on sea surface water temperature, with a significant focus on both organismal response and potential ecosystem level changes. Corals and coral reef ecosystems have received the majority of the attention, as the potential thermal stress responses in these organisms are expected to have far-reaching and dire implications for coral colonies, associated species, and ecosystem level processes. To a lesser extent, thermal stress response has been investigated in other marine organisms.

Summary Stressor Table: Potential effects of shifting productivity
Color reflects the relative severity of an adverse effect compared across all stressors: green=none to mild, yellow=moderate, red=severe.

Ecosystem	Potential Effects
Intertidal	● Altered survival rates for planktonic larvae, especially those with a long larval duration ● Reduced connectivity among insular populations, likely reducing recovery potential
Mangrove Forests	See *intertidal*
Seagrass Beds	See *intertidal*
Coral Reefs	See *intertidal*
Deep Reef Slopes	See *intertidal*
Banks and Seamounts	See *intertidal*
Deep Ocean Floor	● Altered transport of particulate organic material into the deep ocean, which could result in increased hypoxia (in areas with >POM) or fewer nutrient resources (in areas with <POM) ● Decreased diversity and altered assemblage structure ● Altered biochemical cycling, affecting nutrient and chemical composition of upwelled water ● Reduced connectivity among insular populations, likely reducing recovery potential
Pelagic Environment	● Altered survival rates for planktonic larvae, especially those that have a long larval duration ● Altered assemblage composition; likely resulting in a loss of biodiversity ● Altered trophic structure and food web dynamics ● Shifts in species composition, which has potential to disrupt food web dynamics. ● Shift in location and position of pelagic assemblages

The relative thermal tolerance of many marine organisms is roughly correlated with the temperature variability occurring in the organism's natural climate regime (Pörtner *et al.* 2014). The highest temperature tolerances are generally found in species at temperate latitudes, where seasonally-driven temperature changes are often large. In contrast, polar and tropical species have relatively narrow natural thermal ranges and for many of these species, they inhabit waters near their physiological temperature tolerance limits (Storch *et al.* 2014), making even small changes in water temperature problematic. Additionally, the thermal range tolerated by a species can vary among its life history stages, with early stages (*e.g.*, eggs and larvae) generally more sensitive than later ones (Pörtner and Peck 2010). Temperature tolerance can also be affected by the presence of other environmental stressors, such as reduced oxygen or ocean acidification (Pörtner and Peck 2010, Deutsch *et al.* 2015).

The effects of elevated ocean temperature are perhaps best studied in reef-building corals. Elevated water temperatures can cause the symbiotic algae, called zooxanthellae, that are found in coral tissues to leave or be expelled, resulting in coral "bleaching." The loss of zooxanthellae directly affects the coral's energy production, but this loss can be offset to a limited extent by heterotrophic feeding by the coral polyps. If bleaching is prolonged, however, a coral colony will suffer partial or total mortality because of starvation.

Many reef-building corals live close to their upper thermal tolerance and are thus extremely vulnerable to warming (Hughes *et al.* 2003, McWilliams *et al.* 2005). Numerous reports of coral bleaching due to recent warming have been reported (*e.g.*, Hoegh-Guldberg 1999, Sheppard 2003, Reaser *et al.* 2000), including in the Mariana Islands, Hawaiʻi, and Jarvis Island in the PRIA. Bleaching usually occurs when temperatures exceed a "threshold" of about 0.8 to 1 °C above mean summer maximum levels for at least four to six weeks (Hoegh-Guldberg 1999, Pandolfi *et al.* 2011).

Bleaching susceptibility shows high inter- (McClanahan *et al.* 2004, Yee *et al.* 2008) and intra-specific variability (Baird and Marshall 2002) and varies as a consequence of the magnitude of the thermal stress (Kleypas *et al.* 2008), irradiance levels (Mumby *et al.* 2001, Dunne *et al.* 2001), zooxanthellae symbiont types (Berkelmans 2006, Baker *et al.* 2008), species identity (Loya *et al.* 2001), and the thermal history of the organism (Thompson and van Woesik 2009, Oliver and Palumbi 2011). Species identity is one of the best predictors of thermal tolerance due to a predictable hierarchy of susceptibility among coral taxa. Fast growing branching taxa, such as *Acropora* and *Pocillopora*, normally bleach rapidly and experience high rates of whole colony mortality (Baird and Marshall 2002). In contrast, massive taxa such as *Porites* and some faviids take longer to bleach, and often show lower colony mortality (Baird and Marshall 2002). Ultimately, variability in bleaching susceptibility may be driven by the predominant type of zooxanthellae hosted by corals (Glynn *et al.* 2001, Baker *et al.* 2008). For example, increasing thermal tolerance of *Pocillopora* at some locations in the eastern Pacific has been linked to increased prevalence of colonies that host a thermally tolerant clade D symbiont (Glynn *et al.* 2001). Similarly, *Pocillopora* in French Polynesia host a diversity of symbiont types, including clade D (Magalon *et al.* 2007), which may explain their low level of bleaching susceptibility during recent bleaching events compared with many other geographic locations (Pratchett *et al.* 2013).

Corals also show significant variation in their ability to recover following a bleaching event (Baird and Marshall 2002). If sufficient colony tissue survives, recovery can occur within a few years (Diaz-Pulido *et al.* 2009), but recovery often requires a decade or more (Glynn *et al.* 2001, Baker *et al.* 2008, Sheppard *et al.* 2008). In other cases, no appreciable recovery of coral cover has been observed up to a decade following a bleaching event (Graham *et al.* 2007, Somerfield *et al.* 2008). For coral species hosting multiple symbiont strains, shifts to thermally resistant strains are sometimes observed after bleaching events (Thonhill *et al.* 2006, Cunning *et al.* 2016), although reversion to domination by thermally sensitive strains may occur over several years, probably because of a trade-off between bleaching resistance and photosynthetic rate (Jones and Berklmans 2010).

Mass bleaching events, when most of the coral assemblage bleaches, have become more frequent and widespread in the past few decades (Baker *et al.* 2008). These events are often associated with high mortality (Baird and Marshall 2002) and decreased colony growth and reproduction among survivors (Mendes and Woodley 2002). The consistency of the species hierarchy to bleaching susceptibility has led to the prediction that hardier, slow-growing massive species will replace less hardy, fast-growing, branching species on reefs in the future (Loya *et al.* 2001, Hughes *et al.* 2003). Changes in the morphological composition of the coral assemblage (*e.g.*, loss of fast-growing branching and tabular species) would likely result in a loss, or "flattening," of three-dimensional topographic structure (Alvarez-Filip *et al.* 2009), an ecological function that forms a critical part of reef fish habitat. Mass bleaching can be followed by increases in macroalgae, especially when herbivores are absent or avoid consuming macroalgal species (Ledlie *et al.* 2007). Loss of coral diversity and physical structure usually leads to declines in reef community biodiversity (Jones *et al.* 2004, Alvarez-Filip 2009). Fishes and invertebrates that consume or inhabit corals during some part of their life cycle will also likely decline in abundance, although such effects may likely be accompanied by a time lag (Graham *et al.* 2007, Grandcourt and Cesar 2003).

In addition to reef-building corals, zooxanthellae are also found in species of soft-corals, sea anemones, gorgonians, giant clams (*Tridacna* spp.), and some nudibranchs, all of which have the potential to bleach under exposure to stress (Lesser *et al.* 1990, Norton *et al.* 1995, Ishikura *et al.* 1999, Buck *et al.* 2002, Leggat *et al.* 2003, Neo and Todd 2013). As in corals, bleaching reduces photosynthetic rates, alters the metabolism, and affects their growth, ultimately lowering fitness, although the magnitude of the effects varies among species. Following the 1998 mass bleaching event, survival rates of bleached clams were >95% (Leggat *et al.* 2003), compared to some species of coral which experience mortality as great as 99% (Mumby *et al.* 2001). This suggests that *Tridacna* spp. may be better able to cope with bleaching events significantly better than corals.

For non-photosynthetic marine organisms, research is more limited, but the most apparent effects of sub-lethal temperature stress are associated with altered metabolic processes such as growth, changes in the timing and success of reproduction (Walther *et al.* 2002, Walther *et al.* 2005, Parmesan and Yohe 2003), and shifts in the distribution of species (*e.g.*, Thomas *et al.* 2004, Perry *et al.* 2005, Poloczanska *et al.* 2007). For example, laboratory experiments on coral reef fishes have shown that elevated sea water temperatures lead to reductions in critical swimming speeds (Johansen and Jones 2011) and growth (Munday *et al.* 2008), as well as altering the timing of reproduction, reproductive output, and the condition of juveniles and larvae (Munday *et al.* 2008, Donelson *et al.* 2010). Juveniles of many marine fishes are particularly susceptible to changes in temperature, and larvae may succumb to elevated temperatures that their adult stages can survive (Gagliano *et al.* 2007). Shifts in the hatching times of eggs may affect the survival chances of larvae if hatching becomes asynchronous with food availability (Brierley and Kingsford 2009).

Changes in temperature may also change fish behavior, specifically their catchability in the fishery. Increased temperatures are likely to increase metabolic and consumption rates in fish and invertebrates (Kennedy *et al.* 2002), which could lead to higher catch rates using baits and potentially increase the diversity of catch, including unwanted bycatch (Cheung *et al.* 2012). In contrast, increased temperature could also result in increased fish swimming speeds (Peck *et al.* 2006), which could alter the efficiency of towed fishing devices, such as trawl nets (Rijnsdorp *et al.* 2009).

Intertidal species may already exist close to their tolerance limits, and further thermal stress may cause range shifts along continental coastlines (Stillman 2003, Sorte *et al.* 2010), but similar distributional shifts will not be possible on insular shorelines, and may lead to local extirpation of intertidal organisms that cannot adapt to changing conditions. This will result in substantial changes to intertidal assemblages, especially for species that occupy lower vertical positions on the shore because they tend to show lower thermal thresholds (Williams and Morritt 1995, Marshall *et al.* 2015).

The direct effect of increased temperature on seagrasses and macroalgae depends on species-specific thermal tolerances, and the seagrasses' optimal temperature for photosynthesis, respiration, and growth. Warm water species can often increase their photosynthetic rate and respiration over a wide range of temperatures (Perez and Romero 1992, Terrados and Ros 1995). Both respiration and photosynthesis are positively correlated with sea water temperature, but respiration usually increases at a greater rate than photosynthesis, especially at higher temperatures, thus leading to a reduction in net photosynthesis (Bulthuis 1983b; Dennison 1987, Marsh *et al.* 1986, Pérez and Romero 1992, Herzka and Dunton 1997, Masini and Manning 1997, Tait and Schiel 2013, Colvard *et al.* 2014). Thus, species growing near the upper limit of their thermal tolerance, will decrease in net productivity in warming water. Increased thermal stress may also affect flowering (de Cock 1981, McMillan 1982, Durako and Moffler 1987) and seed germination (Harrison 1982, Phillips *et al.* 1983), although the effect of temperature may be complicated by interactions with other stressors, for example, salinity (Caye and Meinesz 1986, Conacher *et al.* 1994). On intertidal shores, photosynthetic biofilms show increased productivity, but net productivity fell as herbivore grazing rates increased under elevated temperature conditions (Russell *et al.* 2013).

While the effects of rising sea temperature on individual species of plankton are not well understood and are likely variable (Huertas *et al.* 2011), rising sea surface temperatures will affect plankton assemblages by upsetting natural carbon dioxide, nitrogen and phosphorous cycling (Toseland *et al.* 2013) through reduced mixing and upwelling brought on by an increase in temperature-driven ocean stratification (see 4.1.2 Shifts in Productivity). This will result in lower primary productivity and decreased diversity, likely resulting in substantial adverse effects which cascade upward through the food chain. For example, increased thermal stress could lead to a decoupling in the timing of reproduction and the timing of plankton blooms (Platt *et al.* 2003), resulting in trophic instability through breaks in food chains (Hipfner 2009, Richardson and Schoeman 2004).

Even species with higher thermal tolerance could be affected by loss of prey species, including commercially important fish species (Beaurgrand *et al.* 2003). Some of these species will themselves shift ranges as a consequence of warming, but this will not necessarily lead to assemblage decline; for example, fish species richness in the North Sea has increased over the last two decades of the 20th century as the region has warmed, but species composition has been significantly altered (Hiddink and Hofstede 2008).

4.1.4 Salinity

Changes in water salinity will have different effects on marine organisms depending upon their ability to osmoregulate. Even minor osmoregulatory stress will result in increased energetic demands, possibly leading to a cascade of effects which are dependent upon the level of metabolic stress incurred. Like temperature tolerances, a species' tolerance, and thus its ability to cope with changes in salinity, is often associated with the natural variability within its habitat; species in estuarine and coastal ecosystems such as mangrove forests tend to display tolerance to a greater range of salinity than organisms found in the nearshore or open ocean ecosystems where salinity fluctuations tend to be small.

Salinity will directly affect estuarine (*e.g.*, mangroves, river mouths) organisms through osmoregulatory stress or indirectly by degrading their habitat, including breeding and nursery areas (Marshall and Elliot 1998). Mangrove trees are facultative halophytes, and tend to grow best when salinity is between five and 75 ppt, although many species can tolerate salinity up to 90 ppt (Krauss *et al.* 2008, Parida and Jha 2010). Mangrove trees do not have a salt resistant metabolism, but instead are equipped with physiological mechanisms that enable them to exclude or excrete salt (Drennan and Pammenter 1982). These mechanisms included one or more of the following (Mohammad and Uraguchi 2013): salt filtration at the root level (Takemura *et al.* 2000, Kahn *et al.* 2001), salt excretion via glands positioned on the undersides of the leaves, and/or salt disposal via accumulation of salt within leaf cells followed by defoliation (Popp *et al.* 1993).

Salinity is directly correlated with the standing crop of mangrove vegetation and productivity (Chen and Twilley 1998, Chen and Twilley 1999, Mall *et al.* 1987, Ukpong 1991), and under normal conditions, the distribution of mangrove species can be explained primarily by salinity gradients (Ball 1988, Ukpong 1994). Therefore, changes in salinity will likely influence the species richness of a mangal, and distributions of species within the forest. Deviations above or below a species' optimal salinity can reduce vegetative growth (Chodhury 2015), likely because of reduced photosynthesis, net photosynthetic rate, stomatal conductance, and transpiration rate (Noor *et al.* 2015). Additionally, changes to salinity can reduce seedling survival and establishment rates (Ye *et al.* 2004, Ye *et al.* 2005), and stunt tree height (Ball and Pidsley 1995, Hao *et al.* 2009).

Summary Stressor Table: Potential effects of thermal stress.
Color reflects the relative severity of an adverse effect compared across all stressors: green=none to mild, yellow=moderate, red=severe.

Ecosystem	Potential Effects
Intertidal	- Increased primary productivity associated with biofilms, but lower net productivity due to temperature-driven increases in grazing rates - Reduced growth due to increased metabolic demands for some animal species - Changed timing and lower success of reproduction for some species - Temperatures above thermal tolerance thresholds could result in extirpation of species unable to migrate due to insular habitat
Mangrove Forests	- Few effects on mangrove trees - Reduced growth due to increased metabolic demands for some animal species - Changed timing and lower success of reproduction for some species - Shifts in species distribution and assemblage composition - Change in behavior of fishes; potentially increased feeding
Seagrass Beds	- Increased photosynthesis and respiration; at higher temperatures a decrease in net productivity, which can alter nutrient cycling - Reduced growth due to increased metabolic demands for some animal species - Increased bleaching in zooxanthellae-bearing invertebrates - Changed timing and lowered success of reproduction for some species - Change in behavior of fishes; potentially increased feeding
Coral Reefs	- Increased bleaching in coral and other zooxanthellae-bearing organisms, resulting in some cases in organism death - Flattening of reef structure leading to loss of diversity, abundance and biomass, including important fishery species - Altered assemblage composition, including the potential for a phase-shift toward algal-dominated assemblage - Changed timing and lowered success of reproduction for some animal species - Reduced connectivity among populations, likely reducing recovery potential

Ecosystem	Potential Effects
Deep Reef Slopes	• Effects likely to be minor due to depth, water movement, and lack of dependency on particulate organic matter from surface waters
Banks and Seamounts	See *Coral Reefs* (shallow) and *Deep Reef Slopes* (deep)
Deep Ocean Floor	• Altered transport of POM into the deep ocean, which could result in increased hypoxia (if >POM) or fewer resources (if <POM) • Altered biochemical cycling, affecting nutrient and chemical composition of upwelled water
Pelagic Environment	• Decreased net primary productivity • Geographic shifts in productivity • Altered survival rates for planktonic larvae, especially those that have a long larval duration • Altered assemblage composition; likely resulting in a loss of biodiversity, leading to changed trophic structure and food web dynamics

While many seagrasses in the Western Pacific Region are primarily marine in nature, they often experience natural fluctuations in salinity because of their shallow, nearshore habitat.

Seagrasses show wide variability in salinity tolerance, which is correlated with the amount of natural variability in salinity found in their habitat. Changes in salinity have been associated with distributional shifts and changes in abundance of seagrasses (Young and Kirkman 1975, Dawes *et al.* 1989, Lazar and Dawes 1991, Quammen and Onuf 1993). For example, vegetative growth of *Zostera capensis*, a mid-saline seagrass in South Africa, is inhibited at high and low salinities, while *Ruppia cirrhosa*, a competing species adapted to fresher water, showed maximum growth near zero salinity (Adams and Bate 1994). Several studies of seagrass seedling survival conducted on a wide range of species have shown that seeds tend to germinate well at relatively low salinities, but optimal seedling growth and development often occur under higher salinity conditions (Caye and Meinesz 1986, Hootsmans *et al.* 1987, Loques *et al.* 1990). Although none of these studies examine species present in the Western Pacific Region, they suggest what may be a general pattern among seagrasses. Salinities that are above optimal can reduce biomass because adjusting osmotic regulation limits seagrass growth by competing for energy, carbohydrate, and nitrogen supplies (Stewart and Lee 1974, Cavalieri 1983, Yeo 1983). In contrast, low salinity has been shown to suppress protein metabolism and alter enzyme activity, again leading to reduced biomass (McGahee and Davis 1971, Haller *et al.* 1974, James and Hart 1993). In addition, salinity has been a major factor influencing the onset and severity of eelgrass diseases (Short *et al.* 1986, Muehlstein *et al.* 1991, Burdick *et al.* 1993), although little is known about tropical seagrass diseases.

Corals have few physiological mechanisms for osmoregulation (Muthiga and Szmant 1987, Mayfield and Gates 2007), so a change in salinity can directly alter metabolic processes and/or cause colony mortality. The effects of salinity changes on coral reefs have not been well-studied, likely because most reefs experience little fluctuation in natural salinity levels, but the response of corals to changing salinity appears to be related to the strength and duration of the exposure and the species affected. As with most other taxonomic groups, considerable inter-specific variation in salinity tolerance is present among coral species. For example, *Stylophora pistillata* is sensitive to small changes in salinity (Sakai *et al.* 1989) whereas *Porites compressa* is more tolerant (Coles 1992). *Platygyra sinensis*, *Acropora millepora*, and *Pocillopora damicornis* have also been found to be relatively tolerant to changes in salinity (Kuanui *et al.* 2015). All of these species are relatively common in the Western Pacific Region. Some coral species have shown evidence of an ability to acclimate to drops in salinity (Ferrier-Pages *et al.* 1999).

Regardless of individual tolerances, high coral mortality has been observed following intense rain events (Sakai *et al.* 1989), including in Hawaiʻi (Jokiel *et al.* 1993 and references therein, Bahr *et al.* 2015). Where mortality did not occur, bleaching, and other metabolic (*e.g.*, increased respiration) and histopathological (swelling and lysis of cells) changes were noted (Glynn 1993, vanWoesik *et al.* 1995, Porter *et al.* 1999, Mayfield and Gates 2007). Severe tissue necrosis, followed by the death of the colonies, has been observed for corals incubated for extended periods in water with relatively small elevations in salinity (Ferrier-Pages *et al.* 1999). Changes in salinity can also adversely affect reproduction (Richmond 1993). Likewise, many coral reef-associated species show low tolerance to salinity changes. Mortality in a wide range of organisms (sea cucumbers, crabs and cryptic fish such as eels) has been observed following freshwater kill events in Hawaiʻi (Jokiel *et al.* 1993, Bahr *et al.* 2015).

At large, oceanic scales, anticipated changes in the ocean's temperature and salinity as a result of climate change will affect circulation patterns. In general, the Pacific Ocean north of the equator is decreasing in salinity, which is expected to affect upwelling strength and location (Bindoff *et al.* 2007). Unfortunately, studies on the effects of salinity changes on non-estuarine phyto- and zooplankton are limited. Estuarine plankton are sensitive to salinity changes, but in many cases, effects associated with temperature, acidification, and nutrient availability are significantly larger. Open ocean plankton assemblages will likely show a similar pattern: the effects of salinity changes on the assemblage will be minor compared to the effects of other stressors. This is reinforced by climate change predictions which predict only small changes in salinity over much of the tropical ocean. Exceptions could include areas where deep ocean mining or OTEC energy production are being conducted, but even under these activities, temperature and nutrient differentials of deep ocean water compared to surface waters are likely to outweigh salinity-related effects. However, more research in this area would be beneficial given the importance of open ocean productivity to broader ecosystem processes.

Summary Stressor Table: Potential effects of salinity
Color reflects the relative severity of an adverse effect compared across all stressors: green=none to mild, yellow=moderate, red=severe.

Ecosystem	Potential Effects
Intertidal	● Organism tend to be extremely tolerant to changes in salinity
Mangrove Forests	● Reduced photosynthesis in mangrove trees and stunted growth at salinities higher or lower than that optimal for the species ● Shifts in mangrove species distributions/zonation based on salinity ● Reduced seedling survival ● Other mangrove associated organisms tend to be salinity tolerant, but will experience sublethal metabolic stress
Seagrass Beds	● Reduced photosynthesis, growth, and biomass at salinities higher or lower than that optimal for the species ● Reduce seedling germination at high salinity ● Reduced seedling growth at low salinity ● Other seagrass-associated organisms tend to be salinity tolerant, but will experience sublethal metabolic stress
Coral Reefs	● Many species have low tolerance to salinity changes ● Increased coral mortality (partial and full) ● Increase mortality among coral reef-associated species (sea cucumbers, crabs and cryptic fish such as eels) that also show low tolerance to salinity changes
Deep Reef Slopes	Unknown (no research available), but likely similar to *Coral Reefs*
Banks and Seamounts	See *Coral Reefs* (shallow) and *Deep Reef Slopes* (deep)
Deep Ocean Floor	Unknown; no research available
Pelagic Environment	● Decreased net primary productivity ● Geographic shifts in productivity ● Altered assemblage composition; likely resulting in a loss of biodiversity, leading to changed trophic structure and food web dynamics

4.1.5 Irradiance

Marine organisms are sensitive to changes in irradiance levels, both photosynthetically active radiation (PAR) and ultraviolet radiation (UV). Decreases in irradiance (often associated with decreased water clarity) generally results in lower photosynthetic rates. Increase irradiance, especially high UV exposure cause cellular damage.

Most research on corals has focused on increased irradiance, which has been linked to coral bleaching (Hoegh-Guldberg 1999, Jones *et al.* 1998) and damage to DNA. High irradiance can amplify the effect of thermal stress on corals (Coles and Jokiel 1978), whereas shading by high islands (Bruno *et al.* 2001), unusually cloudy conditions (Mumby *et al.* 2001), and even increased water turbidity (West and Salm 2003, Anthony *et al.* 2007), can ameliorate the effects of thermal stress on corals. Decreases in irradiance have been shown to affect settlement of coral larvae, and may account for depth zonation in at least five species of Indo-Pacific corals (Mundy and Babcock 1998).

Light limits the distribution and species composition of seagrass beds, and low irradiance levels reduce individual plant biomass and growth rates (Dennison 1987, Abal and Dennison 1996, Ralph *et al.* 2007, Campbell *et al.* 2007). Seagrasses have high respiratory (metabolic) demands needed to support and oxygenate their extensive root and rhizome biomass (Waycott *et al.* 2011), and they use only a limited range of the light spectrum. Seagrasses have a higher minimum light requirement than marine algae and phytoplankton (Dennison *et al.* 1993), making them competitively inferior under reduced light conditions. Thus, seagrasses are generally restricted to shallow coastal areas where ample sunlight can penetrate to the bottom, although considerable species variability exists (Dennison *et al.* 1993). For example, Indo-Pacific species of *Halophila* can grow at greater depth because of a lower minimum light requirement (Erftemeijer and Stapel 1999), a trait usually attributed to the morphology of *Halophila* (Middelboe and Markager 1997).

Seagrasses exhibit several physiological and morphological responses to reductions in irradiance. The magnitude and time required to initiate a response is species-specific, and depends on light intensity and duration, and interactions with other potential stressors, such as water temperature and nutrient availability (Bulthuis 1983a, Bulthuis 1983b, Gordon *et al.* 1994, van Lent *et al.* 1995, Abal 1996, Grice *et al.* 1996, Longstaff and Dennison 1999). Initial effects can include changes in amino acid content and chlorophyll levels (Longstaff and Dennison 1999). Later effects can include reduced biomass, shoot density, leaf production rates, and canopy height (Wiginton and McMillan 1979, Dennison and Alberte 1982, Dennison and Alberte 1985, Neverauskas 1988, Tomasko and Dawes 1989, Abal *et al.* 1994, Lee and Dunton 1997, Peralta *et al.* 2002).

Few studies have looked at the effects of irradiance on tropical Pacific macroalgae. While interspecific variation exists, the minimum light requirements of macroalgae (Sand-Jensen 1988, Duarte 1991, Markager and Sand-Jensen 1992, Dennison *et al.* 1993) and CCA (Littler *et al.* 1985) are lower than those of seagrasses. Thus, marine algae are generally able to survive and outcompete seagrasses under low light conditions, and their distribution (especially their maximum depth) is determined in part by their minimum light requirements for photosynthesis and growth.

Sun light is absorbed and scattered in the ocean, and irradiance decreases exponentially with depth. As with benthic primary producers, spatial and temporal variations in light affect the vertical distribution of phytoplankton. Under climate change forecasts, some areas of the Pacific Ocean are expected to experience increased cloud cover (*e.g.*, Western Pacific Warm Pool, Intertropical Convergence Zone, Pacific Equatorial Divergence), which will reduce irradiance and contribute to declines in primary productivity (Le Borgne *et al.* 2011). Other areas of the Pacific Ocean are expected to experience increased irradiation because of reduced cloud cover (*e.g.*, North and South Pacific Tropical Gyres). Primary productivity is sensitive to both too much and too little light. Photosynthesis can be reduced in the upper water column due to photo-inhibition. Alternatively, photosynthesis rates can drop three-fold if irradiance is reduced to 10% of that present on a sunny day (Le Borgne *et al.* 2011). The potential effects of these changes in irradiance on ocean productivity are unclear, but given that vertical mixing within the surface layer prevents planktonic organisms from staying in the upper photic zone for long, these changes in surface irradiation are expected to have a weak effect on ocean productivity (Le Borgne *et al.* 2011).

4.1.6 Noise

Sounds in the marine environment can originate from abiotic and biotic sources, including the movement of water, geologic events, and the noises generated by fish, marine mammals, and invertebrates. Organisms produce sounds to communicate over short and long distances with mates, offspring and other conspecifics, and/or to find prey or other objects of interest (Popper and Hastings 2009, Simpson *et al.* 2016).

Sources of anthropogenic sounds in the ocean are extensive and varied (Peng *et al.* 2015), and anthropogenic noise covers the full frequency bandwidth that marine animals use, from 1 hertz (Hz) – 200 kilohertz (kHz) (Stocker 2001). It also occurs throughout all ocean ecosystems, from shallow coral reef and seagrass beds down into the deep sea, including the deep ocean floor. Due to the efficiency of sound transmission in the ocean, noise travels great distances and containment is difficult.

Boats of all sizes are a significant source of noise. Pile driving is important in the construction of bridges, wind farms, and seaports. Sonar is used by military, the shipping and fishing industries, and in oceanographic research. Underwater explosions occasionally occur as part of military training, and, while seldom used in the Western Pacific Region, seismic devices such as air guns are used for oil exploration and for studies on undersea geology. Even bubble noise from scuba divers has been linked to altered fish behavior (Lobel 2005).

Noise in the marine environment has a broad range of potential effects, especially when it is very loud, *i.e.*, high amplitude (Casper *et al.* 2016), or when it is less intense but long-lasting (Popper and Hastings 2009). Intense, high amplitude sounds, such as pile driving, underwater explosions, and seismic air guns, can cause immediate death or tissue damage that might or might not directly result in the death of the organism (McCauley *et al.* 2003), but which might lower its fitness (Casper *et al.* 2016). Temporary hearing loss may also occur, which is likely to lower fitness until hearing recovers. Behavioral changes can occur, resulting in animals leaving

Summary Stressor Table: Potential effects of irradiance
Color reflects the relative severity of an adverse effect compared across all stressors: green=none to mild, yellow=moderate, red=severe.

Ecosystem	Potential Effects
Intertidal	● Organism tend to be tolerant to changes in irradiance
Mangrove Forests	● Few effects on mangrove trees unless extreme; leaves are above the water surface so unaffected by reduced water clarity
Seagrass Beds	● Reduced seagrass photosynthesis, biomass, shoot density, leaf production rates, and canopy height under reduced light conditions ● Potential for a phase-shift toward algal-dominated assemblage under low light regimes
Coral Reefs	● Increased risk of coral bleaching at high irradiance; depth dependent sensitivity to UV ● Reduced photosynthesis, calcification, and growth at low irradiance; potential for reduced fitness under prolonged shading ● Potential for a phase-shift toward algal-dominated assemblage under low light regimes
Deep Reef Slopes	● Photosynthetic organisms highly adapted to low light conditions and could experience photo-inhibition under elevated irradiance ● All photosynthetic organisms at the extreme lower irradiance threshold; further reductions would result in mortality, loss of diversity, abundance and biomass of the entire assemblage
Banks and Seamounts	See *Coral Reefs* (shallow) and *Deep Reef Slopes* (deep)
Deep Ocean Floor	Unknown, but the lack of photosynthetic organisms suggested minimal adverse effects would occur
Pelagic Environment	● Decreased primary productivity ● Altered assemblage composition; likely resulting in a loss of biodiversity, leading to changed trophic structure and food web dynamics

feeding or reproduction grounds (Slabbekoorn *et al.* 2010) or becoming more susceptible to mortality through decrease predator-avoidance responses (Simpson *et al.* 2016). Less intense but chronic noise, such as that produced by continuous boating, can cause a general increase in background noise over a large area. Although not likely to kill organisms, chronic noise can mask biologically important sounds and alter the natural soundscape, cause hearing loss, and/or have an adverse effect on an organism's stress levels and immune system.

Little empirical research has been conducted on the effects of noise on tropical marine species, but most of that has focused on marine mammals. Research conducted on model fishes (*e.g.*, tilapia, goldfish, etc.) have shown a wide range of potential effects from excessive noise, most of which were sub-lethal (see Popper and Hastings 2009 for a review). Nichols *et al.* (2015) found that coastal marine fishes secreted stress hormones in the presence of shipping noise. Bluefin tuna showed a disruption in their schooling structure and swimming behavior when exposed to boat noise, as well as an increase in aggressive behavior (Sarà *et al.* 2007). Embryonic clownfish showed increased heart rate in the presence of elevated noise (Simpson *et al.* 2005). Chronic boat noise can reduce the startle response of coral reef fish, increasing their susceptibility to predation (Simpson *et al.* 2015). While it is often assumed that most motile animals will leave noisy areas, this is not always the case (Iafrate *et al.* 2016).

Reef fish use aspects of reef noise to select suitable settlement habitat, and anthropogenic noise that interferes with their "soundscape" could adversely affect their behavior. Simpson *et al.* (2008) found settlement-stage fish of six reef fish families (Pomacentridae, Apogonidae, Lethrinidae, Gobiidae, Syngnathidae, and Blenniidae) preferentially settled into light traps emitting high-frequency reef noised compared to low-frequency reef noise or silent traps. Only the Siganidae showed no preference between any of the sound treatments. High-frequency reef noise is produced mainly by marine invertebrates, and appears to be used by the fish as a means of selectively orienting towards suitable settlement habitats. Masking of natural reef soundscapes by anthropogenic noise could result in changes to the abundances of species and alterations to the structure of reef fish assemblages.

Prawns have been shown to be as sensitive to sound as fish (Lovell *et al.* 2005), and increased metabolic rates have been observed in brown shrimp exposed to elevated noise conditions, causing a reduction in growth and reproduction over three months (Lagardère 1982). Intense noise, such as pile driving and seismic surveying has been shown to reduce feeding rates in mussels (Spiga and Caldwell 2016) and cause larval malformations in scallops (Aguilar de Soto *et al.* 2013). Temperate lobster increased their food consumption for weeks to months after low-level exposure to seismic noise (Payne *et al.* 2007), suggesting increased metabolic demands. Similar effects have also been found in multiple crab species (Edmonds *et al.* 2016, Wale *et al.* 2013a, 2013b), suggesting sub-lethal stress effects in the presence of boat noise might be common in crustaceans.

Anthropogenic noise may mask deep-water invertebrate scavengers' sensitivity to 'micro-seismic' events in the frequency range of 30 Hz – 250 Hz, which they use to detect food-fall up to distances of 100 m (Klages and Muyakshin 1999). Some animals appear to adapt to "threat" sounds; recent anecdotal evidence suggests that schools of pelagic shrimp have adapted evasion strategies toward the sound of shrimp trawlers (Stocker 2001). When the trawlers circle in, the shrimp dive deep, below the nets. Similar behavior has been noted among carangid fish to boats on Midway Atoll, where a catch and release fishery operated for several years (Minton, pers. obs.). The flight response at Midway was opposite that observed at neighboring Pearl and Hermes, where carangids were frequently attracted to small vessel sound, sometimes forming schools of hundreds of individuals.

4.1.7 Hypoxia

In the marine environment, oxygen from the atmosphere and produced as a by-product of photosynthesis dissolves in the water and helps to meet the respiratory demand of all marineorganisms. When the supply of oxygen is diminished or it is removed, or the consumption rate exceeds the resupply rate, dissolved oxygen concentrations can decline below the point that sustains most marine life. This condition of low dissolved oxygen is known as hypoxia. The complete absence of oxygen is called anoxia.

Oxygen solubility in seawater is a function of water temperature, and as the oceans have warmed over the past half century, dissolved oxygen has declined (Garcia *et al.* 2005). By the end of the century, ongoing warming together with rising atmospheric CO_2 will likely result in an expansion of low oxygen zones, perhaps by more than 50% of their present volume (Diaz and Rosenberg 2008, Oschlies *et al.* 2008). This will result in adverse effects on some of the world’s most productive fishery regions.

While temperature controls the amount of oxygen that can dissolve in seawater (fully-saturated seawater at 25 °C [77 °F] has an oxygen concentration of about 8.25 milligrams (mg)/liter (L), water column stratification and increased decomposition of organic matter are two processes that contribute to hypoxic regions in the ocean. Stratification of the water column reduces mixing of oxygen-rich surface layers with deep ocean waters, and microbial decomposition of POM increases respiration in deep ocean waters, resulting in a net decrease in dissolved oxygen at depth. Increased productivity in surface waters, especially in areas with anthropogenic inputs of coastal nutrients, increases the amount of POM that sinks into deep water layers, creating or exacerbating what have been called "dead zones" (Diaz and Rosenberg 2008). Therefore, increased productivity, coupled with increased oceanic stratification, has the potential to result in oxygenated surface waters and a hypoxic deep ocean, leading to the loss of biodiversity.

Most marine organisms experience a hypoxic response when the oxygen concentration falls below 2-3 mg/L (Gray *et al.* 2002, Stramma *et al.* 2008), but considerable interspecific variability exits (Vaquer-Sunyer and Duarte 2008, Seibel 2011). Vaquer-Sunyer and Duarte (2008) suggest this threshold is too low, and noted that many species experience lethal effects below 4.6 mg/L, and significant sublethal effects at oxygen concentrations below 5 mg/L. Crustaceans and fish appear to be particularly susceptible to hypoxic conditions, and mollusks and non-coral cnidarians appeared most tolerant (Vaquer-Sunyer and Duarte 2008). While there is considerable variability among species in a taxonomic group, motile organisms appear to be more sensitive to hypoxic conditions than sessile ones; many fish and motile organisms can detect, and actively avoid hypoxic areas (Pihl *et al.* 1991). Wannamaker and Rice (2000) studied the behavior of six species of fish and one species of shrimp, and all could detect and avoid hypoxic conditions.

Summary Stressor Table: Potential effects of noise
Color reflects the relative severity of an adverse effect compared across all stressors: green=none to mild, yellow=moderate, red=severe.

Ecosystem	Potential Effects
Intertidal	● Effects are expected to be minor for mid-to-high intertidal organisms due to lower exposure ● For low intertidal organisms, high amplitude noise can cause mortality, hearing damage, and disrupted behavior which may reduce fitness ● Chronic low amplitude noise may disrupt behavior ● Individuals may relocate from area of the noise ● Adverse effects generally resolve shortly after the cessation of the noise
Mangrove Forests	● High amplitude noise can cause mortality, hearing damage, and disrupted behavior which may reduce fitness ● Chronic low amplitude noise may disrupt behavior ● Individuals may relocate from area of the noise ● Adverse effects generally resolve shortly after the cessation of the noise
Seagrass Beds	See *Mangrove Forests*
Coral Reefs	See *Mangrove Forests*
Deep Reef Slopes	See *Mangrove Forests*
Banks and Seamounts	See *Mangrove Forests*
Deep Ocean Floor	See *Mangrove Forests*
Pelagic Environment	See *Mangrove Forests*

While little research has been done on the effects of hypoxic conditions on tropical Pacific organisms, in general, marine animals respond to hypoxia by first attempting to maintain oxygen levels through increased respiration rate or increasing the number of oxygen-transporting cells, followed by conserving energy through metabolic depression and down-regulation of protein synthesis and other regulatory enzymes (Holeton and Randall 1967, Burggren and Randall 1978, van den Thillart and Smit 1984, Wu and Woo 1985, Dunn and Hochachka 1986, Boutilier *et al.* 1988, Chew and Ip 1992, Randall *et al.* 1992, Dalla Via *et al.* 1994). Reduction in movement is commonly employed by marine organisms to conserve energy and reduce metabolic demand under hypoxic conditions. For example, swimming of Atlantic cod (*Gadus morhua*) was reduced by ~60% under hypoxic conditions (Schurmann and Steffensen 1994), and digging activity in an Atlantic lobster ceased (Eriksson and Baden 1997).

Hypoxic conditions reduce growth and feeding, which may eventually affect individual fitness. Growth reductions have been shown in brittlestars, oysters (*Crassostrea virginica*), and mussels (*Mytilus edulis*) (Diaz and Rosenberg 1995), as well as in some polycheate worms (Forbes and Lopez 1990). Similarly, reduced growth has been demonstrated in fish subjected to hypoxia (Petersen and Phil 1995), likely a result of reduced feeding (Wu 2002). When subjected to hypoxic conditions, feeding rate was reduced in crabs, gastropods, annelid worms, and lobster, but this effect can vary with life history stage (Das and Stickle 1994, Baden *et al.* 1990a, Baden *et al.* 1990b, Llanso and Diaz 1994).

The effects of hypoxia on reproduction and development of marine animals remains poorly studied, but fish can suffer increased embryo and larval mortality when exposed to hypoxic conditions (Keckeis *et al.* 1996). High mortality and adverse effects on development and growth were found in oyster (*C. virginica*) larvae (Baker and Mann 1992), and mussel (*M. edulis*) embryos experienced delayed development (Wang and Widdows 1991). Hypoxia can also retard gonad development, fertilization success, reproductive output, larval hatching and larval success in the common carp (Wu *et al.* 2003).

Avoidance of hypoxic areas can make organisms more vulnerable to predation. Fish have been observed to change their feeding habits to prey upon hypoxia-stressed benthic invertebrates (Diaz *et al.* 1992). Hypoxia may also affect foraging of predators, reducing prey capture rates, (Sandberg *et al.* 1996, Abrahams *et al.* 2007, Altieri 2008, Johnson *et al.* 1984). Other important behaviors are also dependent upon oxygen concentrations. Fish schooling behavior responds to varying oxycline depth (Bertrand *et al.* 2008). Many benthic organisms such as sea anemones and polychaetes will leave their burrows, and bivalves will extend their siphons upward into the water column above the sediment–water interface, to gain access to more oxygenated water (Pihl *et al.* 1992, Nilsson and Rosenberg 1994, Hervant *et al.* 1996, Sandberg 1997).

Few studies have examined the effects of hypoxia on reef-building corals, even though oxygen concentrations can fluctuate widely on a diurnal cycle and be very low at night (Haas *et al.* 2010; Wild *et al.* 2010). Under low oxygen (2–4 mg/L) conditions, the Indo-Pacific coral *Acropora yongei* bleached, lost major portions of its tissue, and suffered mortality within three days. Its decline in health was accompanied by a significant decrease in photosynthetic performance (Haas *et al.* 2014). In Hawaiʻi, a spill of 233,000 gallons of molasses in Honolulu Harbor resulted in hypoxia-related mortality in coral and fish (Basu 2013), although the extent of the kill is still unresolved. A wide range of Indo-Pacific reef fish have been shown to be more tolerant to hypoxia than expected; 31 species across seven families could tolerate oxygen concentrations as low as 1 mg/L (Nilsson and Ostlund-Nilsson 2004). However, their ability to tolerate hypoxic conditions decreased as water temperature increased (Nilsson *et al.* 2010).

Seagrasses tend to grow in hypoxic sediment and transport oxygen produced by photosynthesis to below-ground tissues (Sand-Jensen *et al.* 1982, Smith *et al.* 1984; Caffrey and Kemp 1991).

However, this photosynthetic oxygen pool can be depleted during the night, and insufficient oxygen supplied to the roots results in sulfide intrusions (Pedersen *et al.* 2004, Holmer *et al.* 2009), which has severe adverse effects growth and survival (Holmer and Bondgaard 2001, Koch *et al.* 2007, Mascaró *et al.* 2009, Borum *et al.* 2005, Frederiksen *et al.* 2007). Anoxia also impairs root growth, and nutrient uptake (Smith *et al.* 1988, Zimmerman and Alberte 1996). The depletion of oxygen reserves during night time respiration is exacerbated when water column oxygen concentration is lower (Holmer *et al.* 2009). Likewise, mangrove trees have special physiological adaptations to oxygenate roots and avoid sulphide intrusion, which have been demonstrated to depress normal growth and metabolism in *Rhizophora mangle* (Lin and Sternberg 1992).

At a population and ecosystem scale, sensitive species may be eliminated in hypoxic areas, thereby causing changes in species composition of benthic, fish, and phytoplankton assemblages. Decreases in species diversity and species richness are well documented in hypoxic areas, and changes to food web structure and functional groups have also been reported in areas with low oxygen availability (Wu 1982, Dauer 1993, Pihl 1994, Diaz and Rosenberg 1995, Altieri 2008). Under hypoxic conditions, there is a general tendency for suspension feeders to be replaced by deposit feeders (Levin 2000); demersal fish by pelagic fish; and macrobenthos by meiobenthos. Microflagellates and nanoplankton also tend to dominate phytoplankton assemblages in hypoxic environments (Josefson and Widbom 1988, Diaz and Rosenberg 1995, Qu *et al.* 2015, Rakocinski and Menke 2016, Briggs *et al.* 2017). A reduction in the biomass of fishes has been generally observed in hypoxic areas (Dyer *et al.* 1983, Rosenberg and Loo 1988, Pihl *et al.* 1992, Baden *et al.* 1990a, Baden *et al.* 1990b, Breitburg 1992, Petersen and Pihl 1995, Lekve *et al.* 1999), accompanied by shifts in species dominance, with less biomass of deep-dwelling species, but more biomass of opportunistic ones (Dauer 1993).

While data are limited, it appears recovery of benthic communities in temperate regions that have suffered hypoxic conditions can take several years (Diaz and Rosenberg 1995), but recovery may occur more quickly in subtropical environments (Wu 1982). Small-scale hypoxia associated with a point source discharge may recover more quickly because organisms can easily migrate from the surrounding, non-affected areas (Rosenberg 1976).

4.2 Biological Stresses

4.2.1 Invasive Species

Introduced species are organisms that have been moved, intentionally or unintentionally, into areas where they do not naturally occur. Many of them fail to establish persistent populations in their new environment; still others may establish breeding populations but do not experience rapid population growth or appear to cause adverse effects on the ecosystem (*e.g.*, they appear to "naturalize"). Other species, free of the ecological processes and interactions that controlled their population growth in their native range, rapidly increase in abundance to the point that they come to dominate their new environment, creating adverse ecological effects to other species of the ecosystem and the functions and services it may provide. These species are considered invasive (Goldberg and Wilkenson 2004).

Summary Stressor Table: Potential effects of hypoxia
Color reflects the relative severity of an adverse effect compared across all stressors: green=none to mild, yellow=moderate, red=severe.

Ecosystem	Potential Effects
Intertidal	● Hypoxia not a significant issue
Mangrove Forests	● Reduced mangrove tree growth and metabolism, contributing to lower productivity, altered nutrient cycling, reduced ability to filter contaminants ● Changed organism behavior, likely exposing organisms to increased predation risk ● Displacement of mobile species to less hypoxic areas, potentially increasing predation-related mortality ● Increased mortality, especially if oxygen concentrations drop below ~2-4 mg/L ● Altered species composition of benthic, fish, and phytoplankton assemblages, including decreased diversity and altered food web structure
Seagrass Beds	● Reduced seagrass growth and metabolism, contributing to lower productivity and altered nutrient cycling ● Increased dominance of macroalgae, which are more tolerant to hypoxia; potential for a phase-shift toward algal-dominated assemblage under low light regimes ● Changed organism behavior, likely exposing organisms to increased predation risk ● Displacement of mobile species to less hypoxic areas, potentially increasing predation-related mortality ● Increased mortality, especially if oxygen concentrations drop below ~2-4 mg/L ● Altered species composition of benthic, fish, and phytoplankton assemblages, including decreased diversity and altered food web structure
Coral Reefs	● Increase coral mortality at oxygen concentrations between 2-4 mg/L, resulting in loss of topographic structure ● Increased dominance of macroalgae, which are more tolerant to hypoxia; potential for a phase-shift toward algal-dominated assemblage under low light regimes ● Changed organism behavior, likely exposing organisms to increased predation risk ● Displacement of mobile species to less hypoxic areas, potentially increasing predation-related mortality

Ecosystem	Potential Effects
	● Increased mortality, especially if oxygen concentrations drop below ~2-4 mg/L ● Altered species composition of benthic, fish, and phytoplankton assemblages, including decreased diversity and altered food web structure
Deep Reef Slopes	See *Coral Reefs*
Banks and Seamounts	See *Coral Reefs*
Deep Ocean Floor	● Potential for severe hypoxia to result from increase transport of POM into the deep water ● Changed organism behavior, likely exposing organisms to increased predation risk ● Displacement of mobile species to less hypoxic areas, potentially increasing predation-related mortality ● Increased mortality, especially if oxygen concentrations is low ● Altered species composition of benthic, fish, and phytoplankton assemblages, including decreased diversity and altered food web structure ● Disruption of ocean-wide nutrient cycling
Pelagic Environment	● Likely not a significant problem near the surface due to mixing ● Displacement of mobile species to less hypoxic areas, potentially increasing predation- and fishing-related mortality ● Increased mortality, especially among larval forms which appear less tolerant to hypoxia than adults ● Altered species composition of benthic, fish, and phytoplankton assemblages, including decreased diversity and altered food web structure ● Mortality could increase export of particulate organic matter to deep ocean.

While most often invasive species are non-native, native species can also display invasive behaviors following a perturbation that disrupts the “normal” operation of their environment. For example, the native algae *Dictyosphaeria cavernosa*, became invasive in Kāne‘ohe Bay, Hawai‘i following decades of nutrient enrichment and decreased herbivory (Stimson *et al.* 2001) and was the dominant benthic organism in many areas of the bay until a dieback appeared to enable natural ecological process to reassert controls on its population (Stimson and Conklin 2008).

In a review of available data on invasive species, Molnar *et al.* (2008) found nearly three-quarters of marine invasive species were unintentionally introduced via shipping (*i.e.*, ballast water and/or hull fouling). Other significant pathways include agricultural imports, the aquarium trade, and the live fish trade.

While marine invasive species have received relatively little attention globally compared to their terrestrial counterparts, numerous species have become problematic in tropical marine ecosystems, especially on coral reefs. These invasive species have displaced native species, caused the loss of native genotypes, modified the physical environment, changed assemblage structures, affected food web dynamics and ecosystem processes, functions and service, impacted human health, and caused substantial economic losses (Grosholz 2002, Perrings 2002, Wallentinus and Nyberg 2007, Molnar *et al.* 2008, Vilà *et al.* 2010, Lapointe and Bedford 2010, Smith *et al.* 2002, Fernandez and Cortes 2005, Stimson *et al.* 2001, Conklin and Smith 2005 Andrefouet *et al.* 2004, Smith *et al.* 2004, Albins and Hixon 2008, Green *et al.* 2012). The growth and success of invasive species are often enhanced by other anthropogenic stressors, such as nutrient runoff (*e.g.*, promotes growth of algae) and overharvest of key herbivore species, although natural stressors, such as disease, can also contribute to their success.

Nearly 500 introduced species have been identified in Hawaiʻi, but only a small number of them are invasive, including three species of algae, 19 invertebrates, and three fishes (Coles and Eldredge 2002, Carlton and Eldredge 2009, Randall 1987, Smith *et al.* 2002). Several of these invasive species are increasing in both abundance and spatial distribution, and threaten ecosystem function by outcompeting native species, especially native structure-forming organisms such as coral. This will contribute to decreased species diversity, changes in trophic structure, and loss of physical structure, but it is not clear exactly how this will affect individual species; effects will likely vary depending upon whether the species-specific interaction affected by invasive species is of a facultative or obligate nature, with the latter relationship likely more sensitive to effects.

On reefs subjected to nutrient enrichment or the removal of herbivores, invasive algae have overgrown corals and other benthic invertebrates; cover of invasive algae on some reefs in Hawaiʻi has exceeded 50% (Smith *et al.* 2002, Concepcion *et al.* 2010). The snowflake coral *Carijoa riisei* has been observed overgrowing deep water black corals, causing the mortality of large, sexually mature colonies (Kahng and Grigg 2005). These same individuals provide important ecological functions to deep reef ecosystems. Invasive snappers have altered behavior and habitat use by some goatfish, potentially exposing them to higher mortality from fishing and possibly predation (Schumacher and Parrish 2005).

Fewer invasive species have been documented in other jurisdictions in the Western Pacific Region, but this is likely a result of inadequate survey effort. Given the correlation between shipping and harmful invasions (Seebens *et al.* 2014), regions with high port traffic but few reported invasions (*e.g.*, Guam and Saipan) probably contain more marine invaders than have been documented (Molnar *et al.* 2008), and may benefit from surveys targeted at identifying the presence of invasive species. A recent assessment of invasive species in the PRIA (Franklin and Mancini 2015) identified 15 non-native and potentially invasive species, including five species of bryozoan, two species of polychaete worms, three tunicate species, two sponge species, and one species each of macroalgae, fish, and hydroid. These species were identified from Palmyra Atoll and Johnston Island, both of which have a prior history of human and military activity, and have been the subject of comprehensive biological surveys over the past two decades. Other areas within the PRIA lack sufficient baseline biological information to make determinations (Franklin and Mancini 2015).

4.2.2 Disease

Diseases are a natural part of all ecosystems and play an important selective role in population dynamics. However, when disease outbreaks occur, mortalities can affect not only the host population, but have the potential to cascade through the ecosystem, leading to altered assemblage structure (Lessios 1988), including changes to benthic diversity, composition, and topographic structure, all of which have wide reaching implications on ecosystem function. However, despite decades of research, the ecological effect of diseases in the ocean remains relatively unknown, even when these diseases affect economically and ecologically important species (Ward and Lafferty 2004, Harvell *et al.* 2002). The lack of baseline data on historical disease levels in marine ecosystems is an impediment to determining diseases demographics, etiology, infectiousness, virulence, and spatial distribution.

Many marine organisms serve as potential hosts for a diversity of parasites and pathogens. Lafferty *et al.* (2015) identified 67 diseases with specific economic impacts. Most occurred in temperate waters, and while present in the wild, appeared to be problematic only under high-density aquaculture conditions. Marine disease outbreaks appear to be increasing over the past half century (Ward and Lafferty 2004), but not for all marine taxa. Turtles, corals, mammals, urchins, and mollusks have all shown significant increases in the rate of disease outbreaks, which cannot be attributed simply to increased vigilance or other reporting bias.

Over the past decade and a half, links between changing ocean temperatures and pathogens have been made (Porter *et al.* 2001, Harvell *et al.* 2002, Ward *et al.* 2007, Miller and Richardson 2014). Growth rates of marine bacteria (Shiah *et al.* 1994) and fungi (Holmquist *et al.* 1983) are positively correlated with temperature, and the optimum temperatures for fungal growth coincides with thresholds that trigger thermal stress and bleaching for many coral species (Holmquist *et al.* 1983, Coles *et al.* 1976), leading to the likely co-occurrence of bleaching and fungal infection. The 1998 mass bleaching of coral caused pronounced mortality worldwide, but the demise of some corals was accelerated by opportunistic infections (Harvell *et al.* 2001). Three coral pathogens grow well at temperatures close to or exceeding probable host optima, which suggests that they would increase in warmer seas (Harvell *et al.* 2002). Among marine invertebrates and seagrass, many disease outbreaks are also linked to temperature increases (Harvell *et al.* 2002), and increased ocean temperature has been linked to the northward expansion of oyster diseases in the mid-1980s (Ford 1996, Cook *et al.* 1998).

Additionally, stressors such as increasing water temperature and pollution, make hosts more susceptible to infection (Holmes 1996, Bruno *et al.* 2003, Trevathan-Tackett *et al.* 2013), although some stressors may affect parasites more than their hosts (Lafferty 1997). For example, stressors that decrease host population density may reduce density-dependent transmission of host-specific diseases by reducing contact rates between infected and uninfected individuals

Summary Stressor Table: Potential effects of invasive species
Color reflects the relative severity of an adverse effect compared across all stressors: green=none to mild, yellow=moderate, red=severe.

Ecosystem	Potential Effects
Intertidal	● Decreased species diversity, altered trophic structure ● Disrupted behavior and interactions among and between species
Mangrove Forests	● Decreased species diversity, altered trophic structure, ● Disrupted behavior and interactions among and between species ● Decreased value as nursery habitat ● Altered ecosystem functions to filter sediment, nutrients, and other pollutants
Seagrass Beds	● Decreased species diversity, altered trophic structure, and the potential for a phase-shift to an algal-dominated assemblage ● Potential disruption of nutrient cycling and transport among nearshore marine ecosystems ● Disrupted behavior and interactions among and between species
Coral Reefs	● Decreased species diversity, altered trophic structure and ecosystem function and services ● Disrupted behavior and interactions among and between species ● Increased potential for a phase-shift toward an algal-dominated assemblage
Deep Reef Slopes	See *Coral Reefs*
Banks and Seamounts	See *Coral Reefs*
Deep Ocean Floor	● Effects unclear due to a lack of research, but likely include decreased species diversity and altered trophic structure, and a potential disruption of nutrient cycling
Pelagic Environment	● Effects unclear due to a lack of research, but likely include decreased species diversity, altered trophic structure, and a potential decrease in productivity, alteration of food web dynamics, change in rate of POM export to deep ocean

(Lafferty and Holt 2003). However, any stressor that increases physiological stress in the host has the potential to increase the host's susceptibility to infection. For example, the bioaccumulation of toxins in marine mammals has been demonstrated to affect their immune system and increase susceptibility to disease (Lafferty and Gerber 2002).

Like many invertebrates, corals possess an innate immune system that is characterized by a series of mechanisms that defend the host from infection (Toledo-Hernández and Ruiz-Diaz 2014). In reef-building corals, mucus forms a physical barrier and acts as a first line of defense. Coral mucus is a viscous fluid made of a complex mixture of compounds secreted by the polyps, and which contains a variety of anti-bacterial compounds (Kvennefors *et al.* 2012, Krediet *et al.* 2013), including a variety of symbiotic microbes that prevent the settlement of potentially noxious bacteria (Brown and Bythell, 2005), and a range of viruses that also may play an important role in coral immunology (Nguyen-Kim *et al.* 2015). Factors that affect the mucus layer may have directly lower a coral's immunity to disease. While coral immune systems are generally considered rudimentary and simplistic (Pollock *et al.* 2011, Toledo-Hernández *et al.* 2013), recent research suggests they are surprisingly complex, with some components similar to those found in vertebrates (Reed *et al.* 2010, Palmer and Traylor-Knowles 2012).

The incidence of coral disease has been found to be positively correlated with increasing algal cover (Hayes and Goreau 1998, Harvell *et al.* 1999, Harvell *et al.* 2004), and a link between direct algal contact and coral disease has been established (Nugues *et al.* 2004, Bender *et al.* 2012). Macroalgae populations, including species of common Western Pacific Region genera *Halimeda*, *Hypnea* and *Chlorodesmia*, have been shown to harbor pathogens that have been directly linked to coral disease, although the specific mechanism of transfer between algae and coral is poorly understood (Sweet *et al.* 2013).

In general, Pacific reefs have been considered in good condition, with little concern given to coral and other diseases, but this may only reflect inadequate information for many geographic areas. As more studies are conducted on Pacific reefs, it is becoming clear that diseases exist and may be more widespread than originally believed (Ruiz-Moreno *et al.* 2012, Maynard *et al.* 2015), causing some experts to warn that Pacific coral reefs are on a trajectory of degradation similar to that experienced in the Caribbean where coral reefs have been decimated by disease (Galloway *et al.* 2009, Maynard *et al.* 2015).

Approximately 30 coral diseases are known from the Indo-Pacific region, affecting 97 species of coral (approximately 15% of all species) from 34 genera, and the identification of new diseases appears to be accelerating. Coral disease in the Western Pacific region is widespread with prevalence varying from a low of 0.14% in American Samoa to 0.5% in the Northwestern Hawaiian Islands, and up to ocean-wide highs of 10% along the Great Barrier Reef and 14% in the Philippines (Willis *et al.* 2009, Aeby 2009, Work *et al.* 2009). Disease progression can be variable, advancing across a few millimeters of tissue to >1 centimeter (cm) per day, and depending on the severity and length of the infection can cause partial or total colony mortality (Southerland *et al.* 2004).

Other coral reef organisms affected by identified diseases include coralline algae (Littler and Littler 1995, Aeby *et al.* 2005) and sea urchins, for which a massive die-off contributed to a regional phase-shift on Caribbean reefs (Mumby *et al.* 2006). Researchers believe an urchin disease outbreak may have responsible for a recent mass mortality of *Tripnuestes gratilla* (collector urchin) in Hawaiʻi (T. Work, pers. comm.).

No reports of seagrass disease have been located for the Western Pacific Region, but likely, seagrass diseases are present and their prevalence may increase in the Pacific in the future under warming seas. The limited information on seagrass disease comes from seagrass wasting diseases which has been reported in at least two Atlantic species: *Zostera marina* (eel grass) and *Thalassia testudinum* (turtle grass) (Loucks 2013). This disease was responsible for decimating *Z. marina* meadows in the 1930s with over 90% loss (Muehlstein 1989). The same micro-organism has been identified as the causative agent for both species, suggesting this disease has potential to affect numerous species in different genera. When not lethal, wasting disease has been shown to affect photosynthesis, growth, and leaf litter production (Ralph and Short 2002), which can affect nutrient transport and cycling.

Similarly, relatively few diseases of mangrove trees have been identified, and those that have been identified primarily affect *R. mangle* (Weir *et al.* 2000). Most are linked to a fungal causative agent, at least one of which has been identified in Hawaiian *R. mangle* populations (Kohlmeyer 1969), and which was responsible for rotting of woody tissue below the waterline.

4.2.3 Fish Aggregating Device (FAD) Effect

Nearly any floating object (anchored or unanchored) in the ocean will attract and aggregate organisms, mostly fish, underneath it. This behavioral response has led to the development of FADs as a fishery tool, but this report reviews the FAD effect from non-fishing activities including marine debris, anchored ships, navigational buoys, fixed structures, and floating platforms.

Unlike many of the other stressors discussed in this report, the FAD effect does not directly alter the condition of the physical or biological habitat. The only direct effect to the EFH is the deployment of the object into the environment, which then alters the behavior, and potentially the distribution and fitness of some species. Removal of the object would be expected to restore behavior to its pre-deployment condition. As such, the presence of the object itself is the primary effect on environment.

Fish aggregation has been best studied in relation to fishing FADs, which have been shown to have the potential to adversely affect fishery species and ecosystems (Wang *et al.* 2014), although considerable debate about their potential adverse effects exists (Dagorn *et al.* 2012). FADs have been shown to cause pelagic fishes to move away from their usual migration routes, which can lead them into regions with lower productivity (Fléchet 2008) and result in lower individual fitness and altered spatio-temporal dynamics of the population (Wang *et al.* 2014), but the converse has also been demonstrated (Dagorn *et al.* 2007, Dagorn *et al.* 2012). Compared to free-swimming tuna, tuna associated with FADs show significant differences in feeding patterns (Williams and Terawasi 2014, Fonteneau 2014, Wang *et al.* 2014), fish condition (Hallier and Gaertner 2008, Harley *et al.* 2014, Williams and Terawasi 2014), growth rates (Harley *et al.* 2014, Williams and Terawasi 2014), aggregation patterns (Fléchet 2008), and migratory

Summary Stressor Table: Potential effects of disease
Color reflects the relative severity of an adverse effect compared across all stressors: green=none to mild, yellow=moderate, red=severe.

Ecosystem	Potential Effects
Intertidal	● Species-specific disease may affect populations but not likely to significantly alter tropical intertidal assemblage ● Depending on the species, could result in reduced species diversity, changes in trophic dynamics, and reduced resilience
Mangrove Forests	● Few diseases of mangrove trees have been identified and trees appear to be relatively resistant to disease. ● For non-mangrove tree species, disease could result in reduced species diversity, and changes in trophic dynamics
Seagrass Beds	● Seagrass wasting disease has potential to eradicate seagrass beds, removing important nursery habitat ● Reduced photosynthesis, growth, and leaf litter production ● Altered nutrient transport processes ● For non-coral species, disease could result in reduced species diversity, and changes in trophic dynamics
Coral Reefs	● Increased mortality in coral and important herbivores can lead to significant changes in assemblage diversity and composition, including the potential for a phase-shift toward an algal-dominated assemblage ● “Flattening” of reef structure leading to loss of diversity, abundance and biomass, including important fishery species ● Decreased coral recruitment if significant loss of CCA algae occurs ● Sub-lethal effects reduce growth, reproduction and likely impair organism fitness
Deep Reef Slopes	See *Coral Reefs*
Banks and Seamounts	See *Coral Reefs*
Deep Ocean Floor	● Unknown, no research available
Pelagic Environment	● Depending on the species, could result in reduced species diversity, changes in trophic dynamics, and reduced resilience

direction and displacement rates (Hallier and Gaertner 2008, Williams and Terawasi 2014), although research conducted as part of the Hawaiʻi FAD program suggest these effects are not universal (Grubbs *et al.* 2002, Holland *et al.* 2003, Dagorn *et al.* 2007).

FADs have also been implicated in increased bycatch and mortality of high-level, or apex, predators. An estimated 480,000 to 960,000 sharks per year are killed in the Indian Ocean when caught in drifting FADs (Filmater *et al.* 2013), although the design of these units may be directly responsible. "Smooth-bodies" FAD designs, such as those deployed in the Western Pacific Region have resulted in few adverse interactions with sharks, turtles and other protected species (Holland 2012). Juvenile bigeye tuna often gathers under FADs and are caught before they have a chance to reproduce. In 2013 more than 85% of bigeye tuna landed in the Western Pacific Region were small, and most of these were caught in association with purse seiners around FADs (Harley *et al.* 2014). Nevertheless, the potential to catch small FAD-associated individuals using other methods exists. While mortality from FADs is most likely associated with fishing (which is beyond the scope of this report), other potential ecological effects of fish aggregation should not be discounted. Fish will aggregate under and around any floating object in any shallow water marine ecosystem, not just the open ocean where traditional fishery-related FADs are generally deployed. Shifts in abundance of high-level predators from their natural habitat, can have significant ecosystem effects on the individuals and the population. Changes in the spatial distribution and density through the depletion or concentration of apex predators could induce ecological changes in marine assemblages (Stevens *et al.* 2000, Bascompte *et al.* 2005; Mumby *et al.* 2006), both near the aggregating structure and away from the structure. While potential ecosystem-level effects on the pelagic ecosystem are unclear, reef areas dominated by high-level predators often support greater biomass of herbivores (Stevenson *et al.* 2007), likely because of an indirect effect of predators preying upon intermediate consumers, thereby releasing herbivores from predatory control (Bascompte *et al.* 2005). The presence of herbivores has far reaching ramifications on ecosystem health, particularly on coral reefs, and particularly in combination with other stressors (*e.g.*, nutrients). However, to achieve a substantial adverse effect, structures that promote fish aggregation would need to be numerous and densely deployed in order exert sufficient attraction on many apex predators. Even so, the attractive capacity of a FAD array would be limited because FADs appear to have a limited range of attraction, approximately 10 km (Girard *et al.* 2004). Therefore, provided fishery related mortality is managed at any fish aggregating structure (*e.g.*, Cabral *et al.* 2014), ecosystem-level effects would likely be localized and small in magnitude.

4.3 Physical Stress

4.3.1 Physical Damage

Physical damage to an ecosystem can occur when sufficient mechanical force is generated either naturally through the movement of water (*e.g.*, by a storm, tsunami, etc.) or anthropogenically through contact with an object (*e.g.*, dredge, anchor, feet, groundings, etc.). Shallow water benthic organisms are most at risk to physical damage because they are unable to leave the area of impact or otherwise avoid being impacted. In Hawaiʻi, reef fish have been observed to move into deeper water prior to large storm events (Walsh 1983), likely to escape the physical effects of the storm. Likewise, deep water ecosystems tended to be less affected by physical stress

Summary Stressor Table: Potential effects of fish aggregating
Color reflects the relative severity of an adverse effect compared across all stressors: green=none to mild, yellow=moderate, red=severe.

Ecosystem	Potential Effects
Intertidal	🟩 ● Fish aggregating not a significant stressor
Mangrove Forests	🟩 ● Fish aggregating likely not a significant stressor
Seagrass Beds	🟩 ● Fish aggregating likely not a significant stressor
Coral Reefs	🟩 ● Altered distribution of apex predators ● Altered trophic dynamics, for example, change in fish herbivore abundance could alter herbivory rates
Deep Reef Slopes	🟩 ● Fish aggregating like not a significant stressor
Banks and Seamounts	🟩 ● Fish aggregating likely not a significant stressor
Deep Ocean Floor	🟩 ● Fish aggregating not a significant stressor
Pelagic Environment	🟨 ● Altered distribution apex predators ● Altered fitness for aggregated species ● Altered trophic dynamics

because storm-generated surge seldom extends deeper than ~50 m in the ocean (but see Smith *et al.* 2016), and human activity is generally restricted to shallow, coastal areas. Although storm damage has been observed as deep as 100 m (Harmelin-Vivien and Laboute 1986), activities such as deep-ocean mining (Sharma 2015) have the potential to cause substantial but localized physical damage to deep water ecosystems.

In tropical oceans, physical damage has been best studied in coral reef and seagrass ecosystems. Seagrasses are primarily affected through physical removal of plants, leaving bare patches (sometimes called "blowouts") that are subject to further erosion. Blowouts may lead to a decrease in topographical structure, and an increase in the abundance of early colonizing species, such as fast growing native and/or invasive algae (Short and Neckles 1999). Recolonization for many seagrass species occurs primarily through vegetative branching, and populations may take many years to recover (Williams 1990; van Tussenbroek 1994, Creed and Amado Filho 1999). However, deep water seagrass beds (30 m or more), such as those composed of *Halophila decipiens*, a common species seagrass in Hawai‘i and elsewhere in the Western Pacific Region, show higher recovery rates due to the prolific sexual reproduction and high rhizome growth rates

(Williams 1988). This species (and similar ones) would be less likely to suffer long-term adverse effects from physical damage.

Physical damage on coral reefs is often associated with the breakage or dislodging of coral colonies, but can also manifest itself less severely (*e.g.*, tissue abrasion). Scleractinian corals, which are responsible for the structural complexity of coral reefs, are particularly vulnerable to physical damage because their slow-growing carbonate skeleton is relatively brittle and their polyps are easily damaged. A number of studies have reported coral damage from coastal development (Hawkins and Roberts 1994), boating and anchoring (Tilmant 1987, Rogers 1993), especially in large anchorages such as the Garapan Anchorage off Saipan (Rooney *et al.* 2005), derelict fishing gear and other marine debris (Edward 1999), as well as snorkeling (Rogers *et al.* 1988, Allison 1996), reef walking (Neil 1990, Hawkins and Roberts 1993, Rodgers and Cox 2003, Rodgers *et al.* 2003), and scuba diving (Tratalosa and Austin 2001, Zakai and Chadwick-Furman 2002, Hasler and Ott 2008). While nearly always very minor relative to the other activities mentioned above, scientific investigations have the potential, especially in pristine areas, to result in physical damage to coral colonies and other organisms.

The severity of the damage caused by physical stress to a coral colony is dependent on many factors, including the magnitude of the physical force and the skeletal strength of the organism, which for coral is dependent on skeletal density and colony morphology (Storlazzi *et al.* 2005, Shimabukuro 2014). In general, lobate, encrusting, and other massive colony morphologies tend to withstand breakage better than foliose, table, plating, and branching morphologies. However, these more fragile forms tend to have higher growth rates (Minton 2013), which would facilitate more rapid recovery following damage, provided the colony did not experience total mortality.

Recovery from physical damage can be slow, often on the order of years to decades (Rogers and Garrison 2001). Recovery can be hampered by loose rubble (Dollar 1982, Raymundo *et al.* 2007), which is often generated by the pulverizing of fragile coral morphologies, such as branching or foliose forms. The loose rubble rolls around on the bottom, causing secondary damage to small corals and other organisms, and impairs recruitment (Brown and Dunne 1988, Lindahl 1998, Fox and Caldwell 2006). Often, no recovery is observed until the rubble is washed from the area or solidified to the bottom (Fox and Caldwell 2006, Raymundo *et al.* 2007), usually by coralline algae (natural recovery) or human intervention. While rubble fields may inhibit coral settlement and regrowth, for some coral species fragmentation is a viable form of dispersal (Highsmith 1982), and if environmental conditions are suitable, coral fragments of these species can reattach to the bottom and continue to grow.

The abundances of fish and other coral-associated organisms depend on a reef’s topographic complexity, and the flattening of reefs can lead to declines in biodiversity (Alvarez-Filip *et al.* 2009), including among fisheries species. When combined with other stressors, such as nutrient enrichment, large-scale physical damage can increase the probability of a shift in dominance from coral to algae, known as “phase-shifts.” For example, Jameson *et al.* (2007) found that sites suffering from anchor and scuba diver damage, had a lower frequency of hard coral (especially *Acropora* coral), and higher percentage of algae, suggesting physical damage can contribute to a shift from coral- to algal-dominated assemblages.

The deep ocean floor is unlikely to experience a significant amount of physical damage from non-fishing effects. However, deep ocean mining has the potential to cause significant localized effects. While most studies that have examined the potential adverse effects of deep ocean mining have focused on adverse faunal effects without attempting to link the observed changes to a specific stressor (Ozturgut *et al.* 1980, Foell *et al.* 1990, Schriever *et al.* 1997, Tkachenko *et al.* 1996, Radziejewska 1997, Sharma *et al.* 2001), physical damage to the substratum is expected to be the primary mechanism causing damage. Most mining appears to be conducted in unconsolidated sediment, so breakage of structure-forming organisms is unlikely (Sharma 2015), and many effects are likely associated with sedimentation and smothering. Unfortunately, it's unknown how these changes may cascade through the deep sea food web.

4.4 Pollution Stress

4.4.1 Sediment

A large body of information exists examining the effects of sedimentation, nutrient enrichment and turbidity on marine ecosystems, especially coral reefs (see Rogers 1990, Fabricius 2005, Cabaçoa *et al.* 2008, Erftemeijer and Lewis 2006). Given the often confounding relationship between sediment, nutrients, turbidity, heavy metals, and other pollutants, it has often been difficult to assess the direct causal relationships between increasing sedimentation and ecosystem degradation (Fabricius 2005). Therefore, this section will focus primarily on the direct effects (*e.g.*, smothering, scouring, and burial) that can be attributed to sedimentation. Potential adverse effects associated with nutrients (4.4.3 Chemicals), metals, and other chemicals (4.4.2 Nutrient Enrichment), and turbidity (4.1.5 Irradiance) are covered elsewhere in this report.

Suspended sediment can elicit short- and long-term responses from aquatic organisms depending on the quantity, quality, and duration of suspended sediment exposure (Kjelland *et al.* 2015). In general, high rates of sediment deposition contribute to reduced fitness or death in filter-feeding organisms such as mussels, oysters and other bivalves by clogging their feeding mechanisms (*i.e.*, cilia and siphons) and through direct smothering (Wilber and Clarke 2001, Nicholls *et al.* 2003). Fish are more likely to undergo sublethal stress from suspended sediment rather than mortality because of their ability to move out of an area with high suspended sediment load, although specific responses are not well-studied in coral reef fish or other tropical fish. Displacement can disrupt social interactions, increase intraspecific aggression, reduce reproductive success, increase predator–prey interactions, and alter food web dynamics, larvae disbursement, and settlement (Kjelland *et al.* 2015).

The transport of sediment from land into coastal marine ecosystems is a natural process that is important to mangrove forests and some seagrass ecosystems, but can be detrimental when its rate is changed and/or the physical or chemical composition of the sediment is altered by human activity. Coral reef assemblages change naturally along sediment gradients (McClannahan and Obura 1997, West and vanWoesik 2001, Fabricius 2005), and can flourish at relatively high levels of particulate matter and siltation (Anthony 1999). Sediment transport in the marine environment depends on two factors: the size of the particles, and the strength of water flow (either prevailing currents and/or tidal flux). Sediment composition and grain size are also important parameters when assessing the potential adverse effects on marine ecosystems. Fine

Summary Stressor Table: Potential effects of physical damage
Color reflects the relative severity of an adverse effect compared across all stressors: green=none to mild, yellow=moderate, red=severe.

Ecosystem	Potential Effects
Intertidal	● Organism tend to be resistant to physical damage
Mangrove Forests	● Organism tend to be resistant to physical damage ● Increase mangrove tree mortality if significant damage occurs
Seagrass Beds	● Increased bed erosion in areas where seagrass is removed ● Altered topographic structure could change assemblage structure ● Decreased nursery habitat quality for coral reef fish species
Coral Reefs	● Increased partial or total coral colony mortality ● Damage unlikely to affect all coral colonies, reducing overall threat to the ecosystem ● If widespread damage occurs, shift in coral species composition to more breakage resistant colony morphologies could happen, with likely loss in topographic complexity; may contribute to a “flattening” of the reef and associated loss of biodiversity, abundance, and biomass of reef associated fish and invertebrates
Deep Reef Slopes	See *Coral Reefs*
Banks and Seamounts	See *Coral Reefs*
Deep Ocean Floor	● Physical damage likely not a significant stressor
Pelagic Environment	● Physical damage not a significant stressor

sediment has more potential for greater adverse effects due to their slow settlement rate, ability to re-suspend into the water column, thus prolonging periods of reduced water clarity, and the tendency to form microbial-rich organic flocs (Fabricius and Wolanski 2000). Finally, the composition of the sediment (*e.g.*, terrestrial vs. marine) affects the chemical properties of the particles, which can affect interactions with other pollutants and the availability and quality of light (Te 1997).

Mangrove trees require ~0.5 and 1 cm/yr of natural sediment input from which they extract nutrients. Rates above this threshold can lead to burial of mangrove roots, which is likely to result in tree mortality (Ellison 1998) due to reduced oxygenation of the roots resulting in hypoxia stress. Moreover, the accumulation of sediment can change bathymetry, altering current velocities and impeding the tidal system on which mangroves depend for vital nutrients (Armstrong *et al.* 2010), and reducing the flushing rate of excess sediment (Ellison 2000). Even if burial does not result in mangrove tree death, it can lead to reduced reproductive rates and increased mortality of seedlings (Terrados *et al.* 1997). Effects on mangrove-associated species are not as clear, but burial of soft sediment infauna is likely, and could result in a reduction of light reaching phototrophs and affecting primary productivity, especially in benthic bacteria and algae species.

Sedimentation in seagrass beds can result in burial and decreased photosynthesis due to higher turbidity (see 4.1.5 Irradiance). Sedimentation can also alter bathymetry by changing current velocities and wave conditions (Jensen and Mogensen 2000), which affect the natural deposition rates and cause erosion that can undercut seagrass beds (MacInnis-Ng 2003). The effect of burial by sediment on seagrass depends on several factors including the depth of burial and life history of the species involved (Duarte *et al.* 1997); for example, seagrass species with vertical shoots (*e.g.*, Western Pacific Region genera *Cymodocea*, *Thalassia*, *Thalassodendron*) can modify their vertical growth to keep their leaf-producing meristems close to the new sediment level provided sedimentation is not excessive (Marba and Duarte 1994). Response to burial is highly variable among species, although burial under ~5 cm of sediment often leads to substanital mortality in most species (Manzanera *et al.* 1995, Mills and Fonseca 2003, Erftemeijer and Lewis 2006). The adverse effects of sedimentation are often increased when blade epiphytes are abundant because leaf blades with high cover of epiphytes tend to collect a greater amount of sediment than those with fewer epiphytes, resulting in interference with photosynthesis (Shepherd *et al.* 1989) and causing the blades to sink to the bottom, thus increasing the probability of complete burial (Short *et al.* 1989). Sediment composition can be an important factor limiting seagrass distribution (Koch 2001), and incoming sediment can alter the silt and clay content and the amount of organic matter, leading to changes in species diversity, and/or shoot density and leaf biomass (Terrados *et al.* 1998).

Like seagrasses, potential sedimentation effects on coral reef ecosystems include burial and decreased water clarity from increased turbidity. Unlike seagrass beds, most coral reefs do not experience naturally high sedimentation rates, making them more susceptible to increased sediment loads. Coral reef benthic organisms are easily smothered by sediment (Golbuu *et al.* 2003), and rates >100 $mg/cm^2/day$ can kill exposed coral tissue within a few days (Riegl and Branch 1995), although corals show considerable interspecific variability. Sedimentation rates below a species mortality threshold can reduce photosynthesis rates (Philipp and Fabricius 2003), disrupt polyp gas exchange, inhibit nutrient acquisition (Rogers 1990, Richmond 1993), and increase metabolic costs (Telesnicki and Goldberg 1995) because a coral must increase mucus production to remove sediment from its surface. Sedimentation stress in corals increases linearly with the amount of sediment and the duration of exposure (Philipp and Fabricius 2003), and tissue damage is associated not only with amount and duration, but also with sediment type. Tissue damage is higher when exposed to sediment containing higher organic content and microbial activity, and small grain size (Hodgson 1990, Weber *et al.* 2004); mortality can occur quickly under these conditions, especially for newly settled coral recruits (Fabricius *et al.* 2003). High organic content in sediment promotes microbially induced anoxia and reduced pH, which can cause coral death within less than a day, depending on the concentration of organic matter in the sediments (Weber *et al.* 2004). Coral settlement can be inhibited by a layer of sediment covering otherwise suitable hardbottom (Hodgson 1990), and can disrupt larval attachment and metamorphosis (Gilmour 1999), leading to recruitment failure. Removing cohorts of young corals will impair reef recovery after a disturbance, leading to long-term, ecosystem-level effects.

Sedimentation has been shown to reduce biodiversity, alter coral colony size-frequencies of an assemblage, decrease mean colony sizes, alter growth forms, and reduce growth and survival (see Rogers 1990 for an extensive review). Large colonies, or species with branching growth forms and/or thick tissues tend to be more tolerant of sedimentation; whereas small colonies or species with thin tissues and flat surfaces are often more sensitive (Rogers 1990). Some species with thick tissues can remove particles from their surfaces by tissue extension, mucus production, or ciliary movement (Stafford-Smith and Ormond 1992).

Decreased light reduces photosynthesis (both through partial burial and increased turbidity), lowers calcification rates, and contributes to tissue thinning (Telesnicki and Goldberg 1995; Anthony and Hoegh-Guldberg 2003), but many corals can photo-acclimate to reduced light levels, provided the reduction is not too severe. In areas with chronic sediment issues, reduced irradiance can lead to compressed depth distributions, resulting in lower biodiversity at deeper depths, and will also result in a shallower lower depth limit for overall reef growth, leading to a decrease in the suitable substratum available across the entire coral reef ecosystem.

Natural sedimentation can affect MCE (Sherman *et al.* 2010), but overall, natural sedimentation rates are generally low (Smith *et al.* 2008) and lacks a significant terrestrial component (Weinstein 2014). Sediment effects in MCE tend to be associated with scour, especially in conjunction with intense storm events (Smith *et al.* 2016). The low exposure to natural sedimentation suggest deep reef slopes, particularly those with deep water corals may be sensitive to elevated inputs of terrestrial sediment. Appeldoorn *et al.* (2015), in an assessment of the effects on a MCE within a deep-water dredge disposal site, noted a heavy sediment coating on the substratum, and reduced fish abundance. They attributed the decrease fish abundance to an absence of herbivores, such as surgeonfishes and parrotfishes, and hypothesized this was the result of a decrease in algal cover from reduced light intensity attributable to high turbidity.

In most situations, non-fishing activities are unlikely to introduce significant sediment into pelagic and deep ocean ecosystems, but deep sea mining has the potential to introduce substantial sediment loads over a wide area of the pelagic and the deep ocean floor ecosystems via the dumping of sediment-rich effluent from surface processing vessels. Nutrient-rich bottom water filled with fine particulates has the potential to alter surface water column primary productivity and could result in bacterial flocculation (Wolanski and Fabricius 2000), which will quickly be exported to the deep ocean. Upon sinking, this POM will undergo microbial decomposition, which could increase the probability of hypoxic conditions. Additionally, nodule harvesters suspend fine sediment that settles back on the ocean bottom, burying infauna. This has been shown to alter the structure of benthic macro- and meiofaunal assemblages (Foell *et al.* 1990), and these disturbances can persist for a decade or more (Schriever *et al.* 1997, Sharma 2015).

Summary Stressor Table: Potential effects of sedimentation
Color reflects the relative severity of an adverse effect compared across all stressors: green=none to mild, yellow=moderate, red=severe.

Ecosystem	Potential Effects
Intertidal	● Sedimentation not a significant issue on most exposed shores ● Reduce tide pool depth and area could affect nursery habitat
Mangrove Forests	● Increased mortality through root burial ● Reduced mangrove reproduction success and increased seedling mortality ● Altered oceanographic processes could affect nutrient cycling and transport to offshore ecosystems ● Increased burial of benthic organisms, including photosynthetic algae ● Reduced fitness/increased mortality of filter-feeding organisms (*e.g.*, mussels, oysters and other bivalves) through clogging of their feeding apparatus or smothering
Seagrass Beds	● Increased seagrass mortality from burial (>5 cm of sediment) ● Altered silt and clay content and the amount of organic matter can result in long-term changes in species diversity, and/or shoot density and leaf biomass ● Reduced fitness/increased mortality of filter-feeding organisms (*e.g.*, mussels, oysters and other bivalves) through clogging of their feeding apparatus or smothering ● Altered behavior in fish, potentially causing decrease in fitness
Coral Reefs	● Increased coral mortality at rates >100 mg/cm^2/d, potentially significant assemblage-level effects at >50 mg/cm^2/d ● Decreased photosynthesis, calcification, and growth ● Coral recruitment failure ● Shift in coral species composition, with likely loss in topographic complexity; may contribute to a "flattening" of the reef and associated loss of biodiversity, abundance, and biomass ● Altered assemblage composition, including loss of diversity of reef associated fish and invertebrates ● Reduced fitness/increased mortality of filter-feeding organisms (*e.g.*, mussels, oysters and other bivalves) through clogging of their feeding apparatus or smothering ● Altered behavior in fish, potentially causing decrease in fitness
Deep Reef Slopes	See *Coral Reefs*

Ecosystem	Potential Effects
Banks and Seamounts	● Banks and Seamounts tend to be isolated from sediment sources, so effects are expected to be minimal.
Deep Ocean Floor	● Increased risk of burial ● Change in species composition, abundance of benthic macro- and meiofauna ● Potential effects through food chain
Pelagic Environment	● Increased flocculation and export of particulate to the deep ocean

4.4.2 Nutrient Enrichment

Rapid population growth on small islands, the development of tourism-based economies, poorly developed and maintained infrastructure, poorly designed or insufficient sewage treatment systems (*e.g.*, coastal zone septic systems and cesspools), and generally poor land management have resulted in significant nutrient enrichment of nearshore marine ecosystems in the tropical Pacific (Adams 1996, Verhoeven *et al.* 2006, Honey *et al.* 2010, Spaulding *et al.* 2011). Coastal development, often immediately adjacent to the ocean, has occurred at a considerable pace and often without regard to its potential effects on the marine environment, although this appears to be changing. Residential and commercial landscaping and agricultural practices have contributed to nutrient-rich, non-point source runoff. In addition to often being a significant human health issue, nutrient enrichment adversely affects nearshore marine ecosystems (Bell 1992, Dubinsky and Stambler 1996, Lapointe 1997, Downing *et al.* 1999, Cloern 2001, Lovelock *et al.* 2009). The section will focus on nearshore nutrient enrichment; for information on changes to open ocean productivity see 4.1.2 Shifts in Productivity.

While mangroves are highly productive ecosystems and fix and store large amounts of carbon (Duarte and Cebrian 1996), they are often nutrient poor (Lovelock *et al.* 2005). Mangroves sustain high levels of productivity despite nutrient limitation through efficient nutrient cycling and nutrient conservation strategies (Reef *et al.* 2010). Nutrient additions can stimulate mangrove growth, and studies have found that small inputs over short time periods often result in no detectable effect on mangrove leaves, soils, or the assemblage structure (Wong *et al.* 1995, Trott and Alongi 2000), although prolonged eutrophication has been shown to have negative consequences on mangrove growth (Lovelock 2009). Under chronic nutrient enrichment, growth tends to favor shoots and canopy production over root structures (Lovelock 2009), resulting in stunted growth forms and a lack of pneumatophores, which eventually lead to plant mortality (Mandura 1997). Less root growth can also increase sensitivity to drought and hypersalinity, leading to increased mortality from water deficits. Nutrient enrichment has also been associated with increased densities of marine wood-borers (Kohlmeyer *et al.* 1995) and herbivory in some bark-mining moths (Feller and Chamberlain 2007). The rate of release of N_2O, a potent greenhouse gas, to the atmosphere can increase exponentially with external nitrogen inputs

(Corredor *et al.* 1999, Allen *et al.* 2007, Krithika *et al.* 2008). Nutrient enrichment favors growth of algae over other benthic organisms, resulting in an algal-dominated benthic assemblage (Lapointe *et al.* 1993).

Nutrient enrichment is considered a major threat to seagrasses worldwide (Short and Wyllie-Echeverria 1996, Ralph *et al.* 2006, Ralph *et al.* 2007, Waycott *et al.* 2009). Short-term additions of nutrients to seagrass beds generally stimulate plant growth resulting in increased biomass and shoot density (Hughes *et al.* 2004). However, if nutrient enrichment is sufficiently large or chronic, it can alter plant architecture, decrease shoot density, reduce biomass, and if persistent, result in seagrass death (Short 1983, van Katwijk *et al.* 1997, Brun *et al.* 2002, Hughes *et al.* 2004, Romero *et al.* 2006, Burkholder *et al.* 2007, Fertig *et al.* 2013). Elevated nutrients can contribute to the excessive growth of epiphytes, macroalgae and phytoplankton, all of which could decrease seagrass growth and survival (McGlathery 1995, Ralph *et al.* 2006, Lee *et al.* 2007, Schmidt *et al.* 2012). Extremely high nutrient regimes can also result in a build-up of organic matter in the sediment, increasing anoxia and creating unfavorable and sometimes toxic sediment conditions for seagrasses (Koch 2001, Koch *et al.* 2006, Ralph *et al.* 2006) and associated organisms. Nutrient enrichment promotes algal growth over seagrasses, potentially contributing to a phase shift from a seagrass- to an algal-dominated assemblage (Lapointe *et al.* 1993).

Coral reefs generally grow in oligotrophic, or nutrient-poor, waters (D'Elia and Wiebe 1990), and nutrient enrichment has been shown to negatively affect coral reef ecosystems (Pastorok and Bilyard 1985, Stambler *et al.* 1991; Dubinsky and Stambler 1996, Loya 2004). Reefs that have been exposed to chronic nutrient enrichment often show an increase in primary productivity, but this is mainly associated with algal growth (Smith *et al.* 1981, Hatcher *et al.* 1989, Bell 1992, Done 1992, Hughes 1994, Lapointe 1997, Schaffelke *et al.* 1998, Fabricius *et al.* 2010), which can quickly occupy hard substratum and potentially overgrow corals, smothering or otherwise outcompeting them (Smith *et al.* 1981, Nairn 1993, Genin *et al.* 1995). This could contribute to a shift to an assemblage dominated by algae (McManus and Polsenburg 2004, Dudgeon *et al.* 2010, Edinger *et al.* 2000, Lapointe 1997), although it is unlikely that nutrient enrichment alone is sufficient to cause such a change, and instead must occur in combination with other stresses (Szmant 2002).

The growth rates of reef algae are believed to be constrained by nutrient limitation and herbivore grazing, thereby preventing algae from overgrowing and killing corals under normal conditions (Carpenter 1986, Lewis 1986, Birkeland 1988, Hay 1991, Littler *et al.* 1991; Lapointe 1997). In the absence of grazing, a nutrient increase could shift the competitive balance in favor of algae. Nutrient enrichment also has the potential to increase water column productivity, resulting in plankton blooms that can reduce water clarity and light for benthic producers, and trigger an increase in the abundance of deposit and filter feeders (Grigg 1995). This shift away from coral dominance would likely result in a "flattening" of the reef (Alvarez-Filip 2009).

While research suggests the effects of nutrient enrichment vary by coral species, type of nutrient input, and the history of the exposed individuals or population, nutrient enrichment generally has an adverse effect on coral. Eutrophication has been reported to cause subtle physiological changes in parameters such as coral growth, skeletal tensile strength, reproduction (Stambler *et* *al.* 1991, Ferrier-Pages *et al.* 2000; Bucher and Harrison 2002; Cox and Ward 2003, Dunn *et al.* 2012), and suppressed calcification rates (Kinsey and Davies 1979; Marubini and Davies 1996; Ferrier-Pages *et al.* 2000). Corals exposed to elevated nutrients often show lower larvae and planula production, impaired planula settlement, decreased gonadal index and fertilization rates, and higher rates of irregular embryos and hermaphroditism (Tomascik and Sander 1987, Richmond 1997, Harrison and Ward 2001, Cox and Ward 2003, Bongiorni *et al.* 2003, Koop *et al.* 2001, Loya *et al.* 2004). Nutrient enrichment has been implicated in reduced ability to withstand disease (Bruno *et al.* 2003, Voss and Richardson 2006, Harvell *et al.* 2007) and may increase susceptibility to temperature stress, thereby increasing the chances of bleaching (Wiedenmann *et al.* 2013). However, responses vary considerably within and among species (Tomascik and Sander 1987; Ward and Harrison 2000; Harrison and Ward 2001; Bongiorni *et al.* 2003), making it difficult to identify generalize trends.

Nutrient additions to the open ocean are unlikely to occur at a large spatial scale, but small scale inputs from activities such as deep ocean mining or OTEC could create localized nutrient inputs. The effects of nutrient additions on primary productivity in the open ocean would be mediated by the availability of limiting elements, primarily iron, which enters the tropical Pacific via wind-blown, terrestrially-derived dust (Falkowski *et al.* 1998). The tropical Pacific, however, is predominately nutrient poor (except in upwelling areas) due to oceanic stratification (Sigman and Hain 2012), and thus may not be severely iron-limited. It could respond to additions of nitrogen, through rapid uptake by phytoplankton and cyanobacteria, potentially leading to phytoplankton blooms. These would then contribute to a zooplankton bloom that could be exploited up through the pelagic foodchain. Ultimately, the production of organic matter, especially POM, would sink and be exported out of the surface layer, into the deep ocean for nutrient recycling. Excess POM in the deep ocean could result in an increased of hypoxia because of microbial decomposition (see
). While localized nutrient enrichment might be possible, humans appear incapable of fertilizing a large enough area of the ocean on a continuous basis to create significant basin-wide effects.

Coastal areas may be subjected to sufficient, chronic nutrient inputs derived from land-based activities to promote conditions that result in seasonal or even persistent phytoplankton blooms. This increased productivity can have numerous potentially adverse effects on nearshore waters, including increased turbidity which can reduce irradiance, altered trophic dynamics in which planktivores and filter feeding organisms are favored over other trophic groups, and an increased likelihood of seasonal dead zones resulting from microbial decomposition of POM, especially in areas were currents and flushing are low (*e.g.,* harbors, enclosed lagoons, etc.).

4.4.3 Chemicals

All marine ecosystems are under threat of contamination from toxic substances, including oil and oil dispersants, industrial chemicals from discharges, household and personal-use chemicals, pharmaceuticals, pesticides from run-off, and antifouling compounds (Spaulding *et al.* 2001). These chemical pollutants can have a variety of lethal and sub-lethal effects on marine organisms, including alteration of growth, interference with reproduction, disruption of metabolic processes, and changes in behavior. These adverse effects can cascade through

Summary Stressor Table: Potential effects of nutrient enrichment
Color reflects the relative severity of an adverse effect compared across all stressors: green=none to mild, yellow=moderate, red=severe.

Ecosystem	Potential Effects
Intertidal	🟩 ● Increased algal growth in lower intertidal, with the potential to alter species composition ● Likely little or no effect on upper intertidal
Mangrove Forests	🟨 ● Chronic nutrient enrichment favors canopy growth over root growth, resulting in a lack of pneumatophores and increased tree mortality ● Increased release of N_2O, a potent greenhouse gas ● Short-term nutrient enrichment unlikely to have noticeable effect
Seagrass Beds	🟥 ● Under high or chronic nutrient enrichment, altered plant architecture, decreased shoot density and biomass, increased hypoxia in sediment, contributing to increased mortality ● Increased abundance of benthic deposit- and filter-feeders ● Increased growth of seagrass epiphytes, macroalgae and phytoplankton, which compete with seagrasses for space and light ● Potential for a phase-shift toward an algal-dominated assemblage
Coral Reefs	🟥 ● Altered coral growth rates, decreased calcification and skeletal tensile strength (could increase physical damage) ● Decrease coral reproductive output, increased rates of irregular embryos, decreased recruitment ● Decreased coral disease resistance ● Increase sensitivity to temperature stress in coral, increasing the risk of bleaching ● Increased abundance of benthic deposit and filter feeders ● Increased growth of macroalgae and phytoplankton, which compete for space and light ● Potential for a phase-shift toward an algal-dominated assemblage
Deep Reef Slopes	🟥 See *Coral Reefs*
Banks and Seamounts	🟩 ● Banks and Seamounts tend to be isolated from nutrient sources, so effects are expected to be minimal.

Ecosystem	Potential Effects
Deep Ocean Floor	Unknown; no research available.
Pelagic Environment	● Increased primary productivity until iron becomes limiting ● Increased abundance of phytoplankton and cyanobacteria, leading to phytoplankton bloom ● Formation of POM that eventually sinks into the deep ocean.

ecosystems, altering species composition, and ecosystem functions and services. Some pollutants are environmentally persistent and can take years or even decades to biodegrade, and others can bio-accumulate and biomagnify through the food chain, eventually posing a direct threat to human health.

Chemicals enter the marine environment through a variety point and non-point pathways (Figure 3), and may be transported great distances from their origin. In the marine environment, the transport, dispersion, and the biological effects of pollutants depend upon the environmental persistence of these chemicals under tropical conditions (*e.g.*, their biodegradation rates), and their propensity to bioaccumulate (van Dam *et al.* 2011). Many contaminants readily attach to sediment particles and are transported into the ocean where they become entrained in the bottom sediment of estuaries, reefs, and potentially deeper ocean ecosystems. Once trapped in sediment porewater, they can continue to flux into the overlying water column (Figure 3), creating a persistent source of contamination long after the initial input has ended. Contaminated organisms carrying accumulated loads of persistent chemicals in their tissues can transport pollutants between marine ecosystems and far from their application or deposition sites (*e.g.*, heavy metals in pelagic fish).

Hydrocarbons

The jurisdictions in the Western Pacific Region have no significant fossil fuel deposits or ongoing extraction activity, so the threat of oil and hydrocarbon pollution is likely low. Hydrocarbons will enter the ocean primarily through run-off from urban areas, and through activities associated with shipping (*e.g.*, spills, fueling, groundings, etc.).

Often, hydrocarbons entering the marine environment do not contact organisms because they stay near the surface where much of it evaporates within a few days (Neff *et al.* 2000), before the remaining non-volatile and semivolite components sink and become entrained in the benthic sediment, where they can potentially persist for years to decades (Owens *et al.* 2008, Bagby *et al.* 2016). However, organisms that use the surface (*e.g.*, marine mammals, some jellyfish, sea birds, etc.) or life history stages that are positively buoyant (*e.g.*, many benthic gametes, including coral spawn) are particularly susceptible to adverse effects from direct contact with hydrocarbons (Haapkylä *et al.* 2007). Rough sea surface conditions can mix hydrocarbons into the water column, and over time some types of crude oils will weather, sink, and adsorb to particulate material (before eventually becoming entrained in the bottom sediment (Fitzpatrick *et*

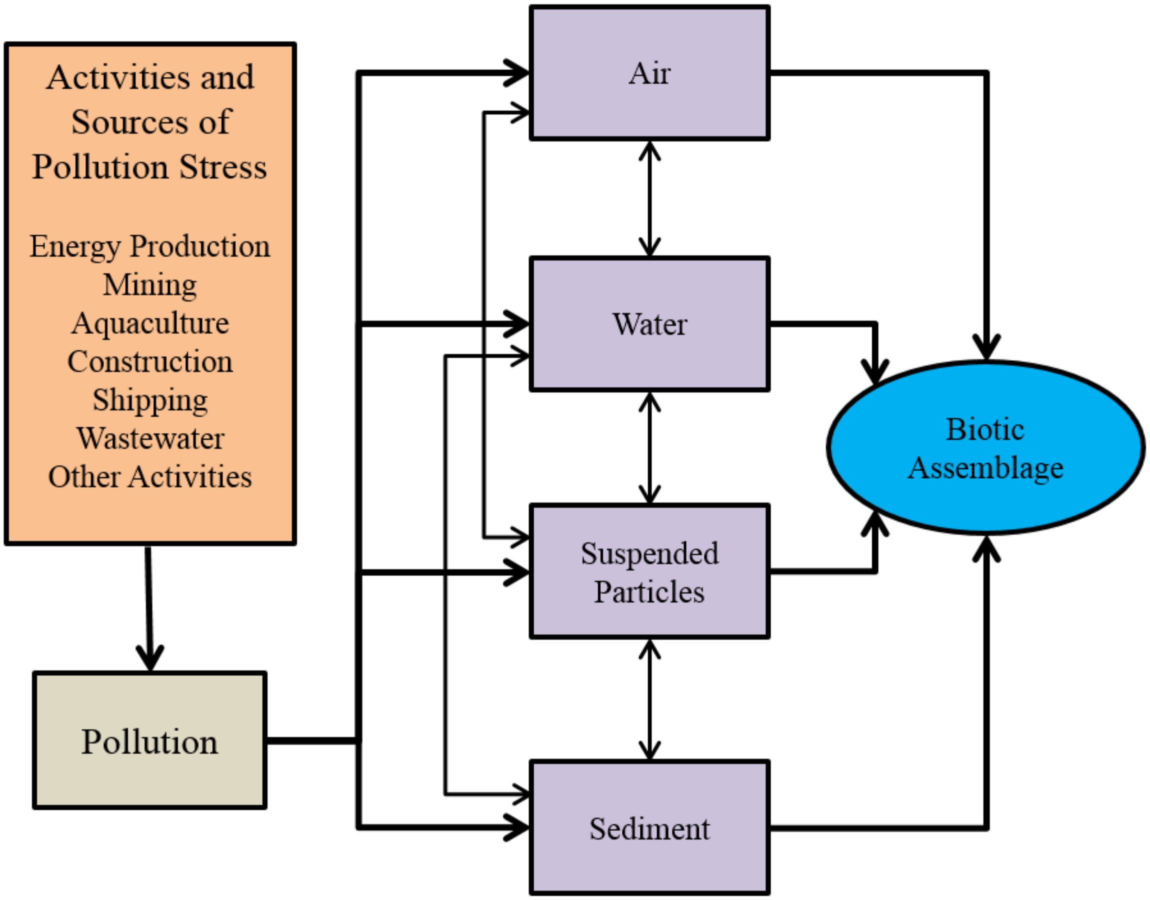

Figure 3. Conceptual model for pollutant pathways in marine ecosystems (modified from van Dam *et al.* 2011).

al. 2015, Gong *et al.* 2014). The sinking of the non-volatile component of the crude increases the chance for adverse effects on "sub-surface" organisms. Direct contact with hydrocarbon itself is not required for an adverse effect to occur because most oil products have a "water-accommodating fraction" that will dissolve into seawater and disperse throughout the water column (Neff *et al.* 2000, Beyer *et al.* 2016). Unfortunately, dispersing agents used to clean up oil spills are often more toxic than the oil itself, and have been demonstrated to cause larval deformities, loss of normal larval swimming behavior, and tissue damage in corals (Epstein *et al.* 2000, Lane and Harrison 2000, Shafir *et al.* 2007, DeLeo *et al.* 2015, Beyer *et al.* 2016).

Mangrove forest ecosystems are particularly sensitive to hydrocarbon pollution because they span the air/water interface and tend to have calm water conditions, which makes it difficult to flush contaminants (Moore 1972, Getter *et al.* 1981). Mangroves are especially sensitive to smothering when pneumatophores, which are responsible for aerating roots, become clogged with oil, causing roots to die from the lack of oxygen (Teas *et al.* 1987, Boer 1993). Both light and heavy crudes have been shown to be difficult to remove from clogged pneumatophores (Reilinger 1991), and recovery can take more than a year (Wardrop *et al.* 1987, Lugo *et al.* 1981, Snedecker *et al.* 1981). Oil can disrupt normal root growth, resulting in deformed aerial roots (Boer 1993, Snedecker *et al.* 1981, Lewis *et al.* 1979, Getter *et al.* 1980, Lewis 1980, Getter *et al.* 1982). The anaerobic soil conditions found in most mangals are not conducive to the biodegradation of oil, and hydrocarbons can persist in mangal soils for years (Page *et al.* 1979). Oiled mangrove trees show reduced productivity, lower rates of litter production and lower seedling survival (Saenger *et al.* 1983). While direct, immediate mortality of mangroves and associated organisms can be high (Nadeau and Berquist 1977, Ray 1981, Getter *et al.* 1981, Saenger *et al.* 1983, Jernelov and Linden 1983, Lewis 1983, Hoi-Chow 1984, Hoi-Chow *et al.* 1984, Teas *et al.* 1987, Garrity and Levins 1993). The added long-term stress on mangrove trees can lead to mortality that extends years into the future (Dodge *et al.* 1995). Recovery of severely damaged mangrove forests can take decades, and depending on the characteristics of the forest, a century or more may be required to replace the lost features, functions and services (Klekowski *et al.* 1994, Davis 1940, Noakes 1955, Tschirley 1969, Westing 1971, Lugo *et al.* 1975). Infaunal populations might recover rapidly, but shrimp, polychaetes, mollusks, and sipunculids may be affected for years (Krebs and Burns 1977, Gilfillian *et al.* 1981, Garrity and Levins 1993), and could experience increased mutations (Klekowski *et al.* 1994).

Damage to seagrass ecosystems includes direct mortality from smothering, fouling, asphyxiation, and chemical toxicity, as well as indirect effects associated with decreased irradiance, trophic disruption, habitat destruction, and loss of sensitive juvenile fish and invertebrates (Zieman *et al.* 1984). Oil in direct contact with seagrasses decreases growth rates, smothers or otherwise damages leaves, and decrease spatial coverage (Jacob 1988). Photosynthetic rates are often depressed, but the magnitude of the reduction varies considerably among species and exposure parameters (Thorhaug *et al.* 1986, Baca and Getter 1984, Thorhaug and Marcus 1985); for example, following spills in the Persian Gulf, seagrasses appeared to be unaffected (Kenworthy 1993). The level of exposure is particularly important for seagrasses because under light oiling, some seagrass species may actually experience enhanced growth for up to decade afterwards (Ballou *et al.* 1989, Dodge *et al.* 1995), a phenomenon in toxicology known as hormesis. Seagrass-associated organisms may or may not recolonize previously oiled beds, resulting in a potential loss of biodiversity (Marshall *et al.* 1993).

Coral reefs may be more susceptible to small, frequent spills than to large single-spill events (Bak 1987, Keller *et al.* 1993, Loya and Rinkevich 1980, Craik 1991). While the chemical composition of the oil can affect its dispersion, emulsification, and weathering, oil released over a reef will generally float above it and not come into direct contact with the corals or other benthic organisms (although reef flats are at risk to direct contact). Oil globules can adhere to the coral tissue (Jackson *et al.* 1989, Marumo and Kamada 1973, Knap *et al.* 1982), and soluble oil components can be adsorbed from the water column by polyps (Knap *et al.* 1982, Burns and Knap 1989, Peters *et al.* 1981), likely a result of the high lipid content of most corals. Effects on coral colonies include mortality, tissue death, reduced growth, impaired reproduction, bleaching, reduced photosynthetic rates, and decreased cellular lipid content, which is correlated with coral fitness (Fucik *et al.* 1984, Cook and Knap 1983, Neff and Anderson 1981, Burns and Knap 1989, Ballou *et al.* 1989, Guzman *et al.* 1993). Coral cover tends to decrease in oiled areas, with potential cascading effects throughout the coral reef ecosystem. Both brooding and broadcasting coral species that are oiled often experience impaired gonadal development (Peters *et al.* 1981, Guzman and Holst 1993). Oil-caused reductions in colony size can result in decreased egg size and fecundity that can persist for years after exposure (Guzman and Holst 1993). Spills occurring near or at peak reproductive season (*e.g.*, summer spawning months for most jurisdictions in the Western Pacific Region) could adversely affect an entire year of reproductive effort because coral gametes and eggs are buoyant, potentially bringing them into direct contact with floating oil. Finally, settlement and recruitment survival can be severely compromised by oil exposure (Loya and Rinkevich 1980, Guzman *et al.* 1993, Messiha-Hanna and Ormand 1982).

Few studies have been conducted on the adverse effects of oil on tropical fish, but decreased growth, altered behavioral responses, and changes in metabolic rate have been observed (Johnson *et al.* 1979, Kloth and Wohlschlag 1972). For several pelagic fish species, including yellowfin tuna, amberjack tuna, and mahi-mahi, exposure resulted in impaired larval swimming and cardiotoxicity (Icardona *et al.* 2014, Mager *et al.* 2014). The water-accommodating fraction can disrupt tropical invertebrate reproduction (Neff *et al.* 2000).

The Deepwater Horizon spill in 2010 produced an extensive hydrocarbon plume that affected deepwater corals up to 22 km away and at a depth of 1,950 m (Fisher *et al.* 2014), resulting in varying degrees of coral tissue loss, sclerite enlargement, excess mucous production, bleached commensal ophiuroids, and a covering of the benthos by brown flocculent material that contained traces of oil (potentially lengthening the exposure period). At sites closer to the wellheads, corals still exhibited significant colony damage at four months after the spill (White *et al.* 2012). Additionally, oil in combination with dispersants used in the clean-up effort proved markedly more toxic than the water-accommodating fraction of the oil alone (Goodbody-Gringley *et al.* 2013, DeLeo *et al.* 2015).

Pesticides/Herbicides

While run-off from Pacific Islands likely contains a range of pesticides and/or herbicides at low concentrations (Orazio *et al.* 2007, Burdick *et al.* 2008, Knee *et al.* 2010, Royer *et al.* 2014), levels below those that impact human health have been shown to adversely affect marine organisms (Richmond 1997, Peters *et al.* 1997, Downs *et al.* 2012). In general, pesticides can cause mortality, reduce growth and fecundity, inhibit fertilization and metamorphosis, alter behavior, and affect photosynthesis. While studies are limited, residual herbicides and breakdown products may not persist at high concentration in aquatic or marine sediment (Edwards 1970).

Unlike many other pollutants, the effects of herbicides on mangals and mangrove trees have received little attention in the scientific literature. Not surprisingly, the few studies available suggest mangals are particularly sensitive to herbicide exposure. Mangrove trees exposed to herbicides experience reduced photosynthesis, plant growth, and biomass production, often leading to mortality (Duke *et al.* 2005, Lovelock *et al.* 2009, Maiti and Chowdhury 2013). Declines in seedling health have been noted (Duke *et al.* 2005). Following extensive aerial herbicide spraying during the Vietnam War, over 40% of the total mangrove forest area of Vietnam experienced substantial mortality (Snedaker 1984, Westing 1984), a level greater than that observed in other vegetative ecosystems that received similar herbicide treatment (NAS 1974, Snedaker 1984, Westing 1984). The heightened sensitivity of mangroves relative to other types of vegetation, however, is poorly understood, but may be associated with its saline environment (Westing 1971), or an increased susceptibility to endocrine disrupting compounds (Snedaker 1984, Westing 1984), which interfere with meristematic tissue (Lugo and Snedaker 1974). In Australia in the 1990s, the herbicide Diuron was implicated in a massive dieback of mangal (Duke *et al.* 2005).

Larger ecosystem effects have also been observed, but direct causal links to herbicides have been difficult to clearly establish. In Vietnam, mangals affected by herbicides showed lower abundance and species richness of planktonic organisms and large fish, but more fish eggs and larvae (NAS 1974), possibly because of an absence of predators. After herbicide spraying marine fishery stocks declined, likely from loss of critical nursery habitat, and the local extirpation of some species occurred (DeSylva and Michel 1975). Not surprisingly, enormous reductions in the abundance of birds were noted in mangals that had been sprayed (Orians and Pfeiffer 1970), which can reduce important nutrient inputs via guano (Adame *et al.* 2015). Recovery of mangrove forest following herbicide exposure is uncertain; estimates vary from 20 years to more than 100 years (Tschirley 1969, NAS 1974, Snedaker 1984). Natural regeneration of mangroves has been minimal in coastal South Vietnam, even after half a century (Westing 1984, Hiep 1984, Marchand 2008). The restoration that has occurred, was the result of extensive human efforts and took over a quarter of a century to return small areas to pre-herbicide condition (Marchand 2008). Recovery in Vietnam has been impeded by the loss of mature seed- or propagule-bearing trees (NAS 1974, Snedaker 1984, Ross 1975), the susceptibility of seedlings to herbicide residuals (Walsh *et al.* 1973), a lack of vegetative cover (NAS 1974) and debris (Ross 1975), and increased erosion (Westing 1984, Ross 1975).

Pesticide applications have adverse effects on mangal species as well. At normal application rates, a mosquito larvicide reached concentrations that were toxic to mysids (Pierce *et al.* 1989), caused sub-lethal effects in fish (Sanders *et al.* 1985, Gehrke 1988), and had significant adverse effects on fiddler crabs (Ward and Howes 1974, Ward and Bush 1976, Ward *et al.* 1976).

Seagrasses appear to show considerable interspecific variability in sensitivity to herbicides, although studies are limited. Diuron has been identified as a significant threat to seagrasses (Haynes *et al.* 2000), and like other herbicides appears to primarily affect seagrasses by disrupting photosynthesis (Ralph 2000, Macinnis-ng and Ralph 2003, Schäfer *et al.* 2007). Diuron is heavily used in U.S. agriculture, including in Hawaiʻi (Royer *et al.* 2014), and has been detected in runoff from sugarcane fields on Maui. Other potential effects of herbicide exposure include mortality, decreases respiration, and decreased production of new shoots and above-sediment biomass (Walsh *et al.* 1982, Mitchell 1987, Grady 1981, Ramachandran *et al.* 1984, Johnson *et al.* 1995).

Pesticides may be more prevalent on coral reefs than suspected, and might merit more attention. For example, in Florida, pesticide residues have been found in samples of lobsters, sponges, crustaceans and fishes from numerous coral reef locations (Glynn *et al.* 1995), suggesting pesticides may be a widespread problem. While no obvious effects on organisms or reef ecosystem were observed in Glynn *et al.*'s study, low concentrations of pesticides, herbicide, and fungicides can inhibit fertilization and metamorphosis and to reduce photosynthesis in numerous species crossing multiple genera that occur in the Western Pacific Region (Markey *et al.* 2007, Jones *et al.* 2003). Pesticides associated with sugarcane production have been shown to reduce photosynthetic efficiency in *Pocillopora damicornis* recruits at low concentrations and short exposure times (Negri *et al.* 2005), cause bleaching in several coral species (Jones *et al.* 2003), and reduce fecundity or entirely inhibit planulae release under longer exposure times (Cantin and Negri 2007). Diuron has been detected at levels above those found to be lethal to corals in runoff adjacent to Maui sugarcane fields, but it is unclear if the runoff entered the nearshore marine waters from the drainage areas in which is was detected (Royer *et al.* 2014).

Metals

Metals can enter the marine environment via numerous pathways, including runoff from urban landscapes, spills, and lubricating muds used in drilling (including directional drilling) (Guzmán and Jiménez 1992, Marx and McGowan 2010, Denton *et al.* 2014, Denton *et al.* 2016). Atmospheric deposition is also a significant source, and is likely the primary source of iron, mercury and other metals to the open ocean (Mason and Sheu 2002, Jickells *et al.* 2005, Sunderland *et al.* 2009). Until the ban on the use of tri-butyl tin (TBT) in 2003, antifouling paints contained the compound as a biocidal component, and were a significant source of tin, copper and zinc. TBT is a persistent compound and is still present in the sediment of many harbors and waterways and around shipwrecks (Smith *et al.* 2003), where it is an important source of toxic substances, especially if the entraining sediment is disturbed.

Mangrove sediment is composed of fine particles with a high organic content and low pH, and are effective at sequestering potentially toxic metals as sulfides (Rand 1995, Harbison 1986, Riedel and Sanders 1988, Lacerda and Rezende 1987, Klerks and Bartholomew 1991). Thus, adverse effects from metal exposures on mangrove trees tend to be minor or nonexistent (Harbison 1986, Defew *et al.* 2005), but at sufficiently high concentrations can result in reduced leaf numbers and stem diameter (Yim and Tam 1999). While metal effects on mangrove trees are generally low, metals can be reintroduced to nearshore waters when they are taken up and concentrated in exported leaf detritus. Metal concentrations can be higher in leaves than in the underlying water or sediment (Peterson *et al.* 1979, Snedaker and Brown 1981, Lacerda *et al.* 1986), although this is not a universal pattern. Tam *et al.* (1995) did not detect lead, chromium, or cadmium in leaf samples from the mangroves in China, but found them in high concentrations in the sediment. Additionally, storms and human activities such as dredging or clearing of mangrove forests can remobilize metals and facilitate transport into coastal waters. Leaf litter is an important food source for many invertebrates (Heald and Odum 1970, Boto and Bunt 1981), and could serve as a pathway through which metals could be transported from mangrove forests to surrounding marine ecosystems. Mercury, a bioaccumulative metal, has been detected in mangrove leaf litter, as well as in a variety of invertebrates and fish trophically linked to the leaf debris (Reimold 1975). Metals have been shown to increase in concentration in mangrove leaf detritus as it ages (Rice and Windom 1982), possibly because of the loss of organic material. Zinc, cadmium, lead, manganese, and copper have all been detected in high concentrations in mangrove leaf debris (DeLaune *et al.* 1981, Nye 1990, Mackey and Hodgkinson 1995, Defew *et al.* 2005).

Many seagrasses directly incorporate metals from the water column into leaf tissue (Brinkhuis *et al.* 1980, Nienhuis 1986), making them a major transport pathway for copper, iron, manganese, and zinc (Drifmeyer *et al.* 1980) to easily pass into the food chain (Ward 1987), and bioaccumulate through higher trophic levels. Several seagrass species are capable of bioaccumulating a range of metals (Pulich 1980, Nienhuis 1986, Wolfe *et al.* 1976, Wahlbeh 1984), including nickel, copper, lead, and zinc (Nienhuis 1986). Seagrass ecosystems have been shown to rapidly uptake TBT, increasing the potential exposure to associated fauna (Levine *et al.* 1990), and potentially leading to decreased invertebrate abundance (Kelly *et al.* 1990). A range of drilling muds have been shown to adversely affect seagrass ecosystems, reducing invertebrate abundance and species richness (Morton *et al.* 1986, Kelly *et al.* 1987), and reducing photosynthetic rates and growth in both seagrasses and their epiphytes (Morton *et al.* 1986, Kelly *et al.* 1987).

Elevated concentrations of metals have been found in the tissues of reef invertebrates. Corals near populated areas have been found to have significantly higher concentrations of metals than those near less populated areas (Howard and Brown 1987, Harland and Brown 1989, Howard and Brown 1984, Howard and Brown 1986, Reichelt and Jones 1994**,** Reichelt-Brushett 2012, Tanaka *et al.* 2013). Metals can enter coral tissues or skeleton via numerous pathways, and evidence exists whereby corals might be able to regulate the concentrations of metals in their tissues (Leatherland and Burton 1974, Riley and Segar 1970, Klumpp and Peterson 1979, Bryan and Gibbs, Brown and Howard 1985, Harland *et al.* 1990). Coral tissue tends to retract in response to environmental stress, exposing skeletal spines, which can directly take up metals from the surrounding seawater (Brown *et al.* 1991). Coral mucus, which is produced in copious quantities in response to metal and chemical exposure (Thompson 1980, Thompson and Bright 1980, Thompson *et al.* 1980, Krone and Biggs 1980, Szmant-Froelich *et al.* 1981, Dodge and Szmant-Froelich 1985, Esquivel 1986), can effectively bind heavy metals (Howell 1982, Harland and Nganro 1990) and may be involved in metal regulation (Harland and Nganro 1990).

Coral branchlets exposed to sediment with a high concentration of anti-fouling compounds suffered significant mortality (Smith *et al.* 2003). Elevated levels of tin can affect the growth rates of coral, especially branching corals (Howard and Brown 1987), by lowering linear extension rates and carbonate accretion, and can affect key biological processes such as respiration (Howard *et al.* 1986), fertilization, metamorphosis (Reichelt-Brushett and Michalek-Wagner 2005; Reichelt-Brushett and Harrison 1999; Negri and Heyward 2001) and larval settlement (Goh 1991, Reichelt-Brushett and Harrison 2000). Even at low concentrations, TBT and copper inhibited fertilization and larval metamorphosis (Negri and Heyward 2001). Heyward (1988) detected the complete inhibition of fertilization in the Western Pacific Region corals *Goniastrea aspera*, *Favites chinensis* and *Platygyra ryukyuensis* gametes when exposed to copper sulphate solutions, and fertilization in the Hawaiian species *Montipora capitata* was adversely affected at low copper concentrations (Hedouin and Gates 2013). Copper has also been shown to impair larval motility (Reichelt-Brushett and Harrison 2004). At the coral assemblage level, metal pollution has been linked to decreased coral species abundance, diversity (Ramos *et al.* 2004), and cover, and more broadly can lead to a shift in the assemblage from one dominated by primary producers to one dominated by filter- and detritus-feeders (Scott 1990).

Zooxanthellae have been shown to accumulate higher concentrations of metals than do host tissues in corals (Buddemeier *et al.* 1981, Harland and Nganro 1990) and clams (Benson and Summons 1981). It has been suggested that sequestering metals in zooxanthellae might diminish possible toxic effects to the host (Harland and Nganro 1990), and that expulsion of algae, which has been reported as a stress response to heavy metals (Harland and Brown 1989, Esquivel 1986, Howard *et al.* 1986), may be a mechanism for metal excretion (Harland *et al.* 1990, Harland and Nganro 1990). Two common Pacific corals, *Porites lutea* and *Pocillopora damicornis*, expelled their symbiotic algae when exposed to elevated metal concentrations (Esquivel 1986; Harland and Brown 1989), a response that was more noticeable in corals obtained from pristine areas. This suggests that corals may be able to develop a tolerance to metal contamination (Harland and Brown 1989).

Like corals, giant clams collected from a populated atoll had significantly higher concentrations of iron, manganese, copper, zinc, and lead than clams from an unpopulated atoll (Khristoforova and Bogdanova 1981). Their symbiotic algae can also influence the uptake of metals by substituting potentially toxic metals for essential elements such as manganese (Hannan and Patouillet 1972, Pilson 1974, Harland and Nganro 1990). This may serve to concentrate metals in zooxanthellae, which can then be expelled to remove the toxic materials.

Metals, including zinc, copper, cadmium, chromium, lead, and mercury, have been detected in the tissue of 50 Indo-Pacific reef fish species from Australia (Denton and Burdon-Jones 1986a), in reef fish from the Gulf of Aqaba (Ismail and Abu-Hi1al 2008), and in a wide range of invertebrates and fish from Apra Harbor, Guam (Denton *et al.* 2006a), with mercury showing evidence of bioaccumulation. Changes in behavior, including erratic swimming, increased gill ventilation, and disrupted schooling ability have been noted in tropical fish exposed to heavy metals (Denton and Burdon-Jones 1986b), as has increased mucus production, fin erosion, and changes in color. While exposure to drilling muds in the Western Pacific Region is expected to be low compared to areas where active oil exploration and extraction are occurring, use of drilling muds in the region is increasing with the increased use of directional drilling technology. The effects of short-term, localized exposure to drilling muds are expected to be low, but considerable uncertainty about the environmental effects of many drilling muds exists due to lack of information on their specific composition. Short-term exposure to drilling muds can decrease coral calcification and growth rates (Hudson and Robin 1980, Kendall *et al.* 1983, Dodge and Szmant-Froelich 1985), including lowering calical relief which could impair sediment-shedding capabilities (Dodge and Szmant-Froelich 1985). Corals were not able to remove drilling muds from their surface under laboratory conditions (Thompson and Bright 1980), but may be successful with assistance from currents (Dodge and Szmant-Froelich 1985). Exposure can reduce photosynthesis, cause bleaching (Kendall *et al.* 1983), increase the likelihood of disease (Parker *et al.* 1984), and result in mortality for some species (Thompson *et al.* 1980). Long-term monitoring of reefs near drilling sites (within ~100 m) have documented large reductions in foliose, branching, and plating corals, although massive corals appeared relatively unaffected (Hudson *et al.* 1982).

Most studies examining the effects of deep ocean mining have focused on adverse faunal effects without attempting to link observed changes to a specific stressor (Ozturgut *et al.* 1980, Foell *et al.* 1990, Schriever *et al.* 1997, Tkachenko *et al.* 1996, Radziejewska 1997, Sharma *et al.* 2001). Deep ocean mining will result in increased sedimentation, physical damage, nutrient enrichment, and the release of trace metals, including nickel, cobalt, copper, manganese, and iron, into both the pelagic and deep ocean environment (Sharma 2015). While the effect of many of these

metals on pelagic and deep ocean organisms is currently unclear, iron has the potential to increase primary productivity in surface waters, and in combination with high-nutrient deep ocean water could increase productivity in areas where mining effluent is discharged. Increased productivity could result in more export of POM from surface waters into the deep ocean, increasing the risk of hypoxia, and potentially alter nutrient cycling (see
), depending on the size of the mining operation.

Polychlorinated biphenyls

Polychlorinated biphenyls (PCBs) are a class of persistent, synthetic chlorinated hydrocarbons manufactured and used in the U.S. beginning in 1929 with production peaking in the 1960s (Parnell *et al.* 2008). Although the U.S. banned their production in 1977 (Breivik *et al.* 2007), PCBs persist as legacy pollutants whose chronic toxicity represents a serious environmental risk (Pivnenko *et al.* 2016). The main bulk of PCBs produced were used in closed applications, especially electrical transformers, where they served as coolants and insulating fluids, and in old fluorescent light ballasts. Open application included uses in carbonless copy paper, plasticizers, flexible coatings for electrical cables, pesticides, flame retardants, caulking, adhesives, etc. Thus, many legacy landfills can have high levels of PCB contamination, both from civilian and military waste (Pivnenko *et al.* 2016). Two particularly relevant avenues for PCBs to enter the marine environment are via marine debris, especially through macro- and micro-plastics (UNEP 2016), and atmospheric deposition, although they can also enter through wastewater treatment facilities (Wang *et al.* 2007, Yao *et al.* 2014). PCBs have been identified from several areas in Mariana Islands (EPA 2000, Denton *et al.* 2006b, Haddock *et al.* 2011), including in marine sediment and organisms from several Guam harbors (Denton *et al.* 2006b), as well as American Samoa (EPA 2015), Hawai‘i (HDOH 2011), and the PRIA (Kerr *et al.* 1997, APSNet 2005, Hathaway *et al.* 2011).

Given their extreme physical and chemical inertness (*e.g.*, thermal stability, low water solubility, etc.) and tendency to adhere to sediment particles, PCBs often accumulate and persist in the marine environment, especially in the sediment of many industrialized bays and watersheds. Offshore sewage discharge and disposal or suspension and transport by ocean currents of sediment dredged from harbors are also potential avenues for contamination of coastal areas with PCBs. PCBs have entered marine food chains through benthic feeding organisms and the ingestion of plastics by higher trophic-level organisms (Ryan *et al.* 1988; Bjorndal *et al.* 1994). Additionally, plankton near the surface can take up PCBs, allowing them to enter pelagic food chains and bioaccumulate in shellfish, and tuna (Soedergren *et al.* 1990).

While considerable research has focused on the human health effects associated with PCB ingestion (especially PCBs bioaccumulated in fish), little research has examined the effect of PCBs on marine organisms. Adverse effects from PCB exposure in adult fish and macroinvertebrates appear to be minor, although some evidence exists suggesting adverse effects may occur to the livers of fish (Rochman *et al.* 2013). Overall, considerably more research is needed. Evidence exists that phyto- and zooplankton are adversely affected through reduced photosynthesis and growth rates, and cell damage (Keil *et al.* 1971, Harding *et al.* 1978, Harding and Phillips 1978). Zooplankton were particularly sensitive to PCB exposure, entirely disappearing in some studies (Iseki *et al.* 1981), but overall, the effects of PCB exposure were variable among species. Widespread PCB contamination could lead to the alteration of the species composition of the plankton assemblage (Iseki *et al.* 1981, Zhao *et al.* 2013). Early larval stages of cod were also found to be sensitive (Foekema *et al.* 2008). Exposure of eggs to low concentrations of PCBs caused developmental abnormalities in subsequent life stages, leading Foekema *et al.* (2008) to postulate that accumulation of PCBs in adult females could have reproductive consequences that are difficult to detect, but may have long-term effects on the population. Fortunately, many PCBs can be metabolized, and rendered inert, although this can often be a slow process, especially for PCBs that are stored in fatty tissue.

Ordnance

Disposal of military munitions in the oceans has been practiced since World War II (Darrach *et al.* 1998, Denton *et al.* 2014), especially in and near historic battle fields in the Western Pacific Region (Minton *et al.* 2006). Additionally, multiple locations within the Western Pacific Region, including numerous small islands, have been employed as military training ranges (*e.g.*, Kaho‘olawe, Ka‘ula Rock, Farallon de Medinilla) resulting in considerable unexploded ordnance (UXO) on the islands and in nearshore marine ecosystems.

The biological effects of UXO on marine organisms and ecosystems, including contamination levels and biological accumulation rates, are not well studied and therefore, poorly understood (Clausen *et al.* 2004, Rosen and Lotufo 2007, Lotufo *et al.* 2009). Two potential threats exist with UXO: detonation and leakage of toxic materials. Detonation risk for UXO in the marine environment appears relatively low. Concussive damage from an exploding ordnance could cause extensive physical damage (see Fox and Caldwell [2006] for a discussion of damage associated with dynamite fishing), but it would be spatially limited, and therefore do not pose a large threat to marine ecosystems.

Munitions are comprised of many potentially toxic compounds that over time will leak into the marine environment. However, their bioaccumulative potential is low because they are weakly hydrophobic (Lotufo and Lydy 2005, Lotufo *et al.* 2009). This has been demonstrated for some of the known UXO compounds in a variety of model test animals, including minnows, carp, goldfish, and marine worms (Lotufo and Lydy 2005, Lang *et al.* 1997, Wang *et al.* 1999, Condor *et al.* 2004). Dietary uptake has also been shown to be minimal relative to aqueous uptake through the gills in fish (Belden *et al.* 2005, Huston and Lotufo 2005), suggesting these compounds will have minor effects through food webs. However, even with low uptake, the transfer and bioaccumulation of many of these compounds in marine organisms have been not been adequately investigated. While no significant effects were found on a mussel or flounder species, low concentrations of chemicals from munitions have been linked to increased mortality in marine copepods, an important component of the zooplankton (Ek *et al.* 2006). Likewise, marine polychaetes and amphipods showed decreased growth, survival, and reproduction (Lotufo *et al.* 2001), and mortality in bivalve larvae (Pascoe *et al.* 2010) at low levels of exposure. Marine algae are also efficient at uptaking toxic compounds leaked from UXO, and can efficiently biotransform the compounds, rendering them inert, although exposure can reduce photosynthesis (Cruz-Uribe and Rorrer 2006).

Even in areas with high concentrations of UXO, most organisms are likely to receive only limited exposure to low chemical concentrations because the munition casings are slow to corrode and break, generally resulting in a slow release of the constituent compounds. Many of the compounds are also efficiently biotransformed and eliminated from organisms once the organisms are removed from the exposure, suggesting mobile organisms are unlikely to bioaccumulate toxic UXO compounds. The potential risk for deleterious biological effects is thus spatially-limited and minor compared to many other potential stressors.

Endocrine Disruptors

In addition to the pollutants described above, many other chemical compounds enter the marine environment because of human activity. While the effects of most chemicals on marine ecosystems are poorly known, endocrine disruptors are a group that has received considerable attention due to their potentially harmful effects. Endocrine disruptors are a diverse group of compounds that adversely affect organisms through deleterious interactions with the endocrine system (Colborn *et al.* 1993). A wide range of substances are thought to cause endocrine disruption, including pharmaceuticals, dioxin and dioxin-like compounds, PCBs, various organochlorine pesticides, plasticizers, and surfactants. These compounds can be found in many common products, including plastic bottles, metal food cans, detergents, flame retardants, food, toys, cosmetics, and pesticides (Porte *et al.* 2006). Many known endocrine disruptors are estrogenic (also known as estrogen mimics), and disrupt reproductive functions. Because of their persistent nature in organisms, many endocrine disruptors bioaccumulate and biomagnify in marine organisms (Colborn 1998, Arukwe *et al.* 1996, Matthiessen 2003, Langston *et al.* 2005, Lye 2000), including in corals (Tarrant *et al.* 2001, Stocker 2016). Similar to exposure to some metals (*e.g.*, TBT in gastropods), endocrine disruptors have been shown to affect hormone systems (Scott and Sloman 2004, Tierney *et al.* 2010).

The effects of endocrine disruptors have largely been studied in marine vertebrates. Fish are particularly vulnerable to exposure because uptake occurs through multiple routes including directly from the water via the gills, skin and gut, through the diet, and through contact with contaminated sediment (Weber and Goerke 2003, Kwong *et al.* 2008). Some endocrine disruptors have been shown to bioaccumulate and bioconcentrate in fish (Ferreira-Leach and Hill 2001, Barber *et al.* 2006, Smith and Hill 2004, Sharma *et al.* 2009).

Endocrine disruptors most commonly affect fish growth, development, reproduction (Hutchinson *et al.* 2006), and behavior (Jones and Reynolds 1997, Scott and Sloman 2004, Sloman and Wilson 2006), potentially affecting the fitness of individuals and adversely affecting the larger populations. Endocrine disruptors disrupt sex steroid activity, thereby affecting sexual development and reproduction. Sex steroid hormones play vital roles in almost all aspects of reproduction, including sexual differentiation, gonadal growth, and reproductive behaviors (Jobling *et al.* 1996, Kiparissis *et al.* 2003 van der Ven *et al.* 2003, Jensen *et al.* 2004, van den Belt *et al.* 2002, Weber *et al.* 2003, Örn *et al.* 2006). Their disruption can lead to high incidence of intersex, abnormal spawning behavior, skewed population sex ratios, and lessened reproductive success (Nimrod and Benson 1998, Parrott and Blunt 2005, Seki *et al.* 2005, Kang *et al.* 2006, Larsen *et al.* 2008, Örn *et al.* 2003, Hahlbeck *et al.* 2004, Örn *et al.* 2006, Iwanowicz and Blazer 2014).

Compared to vertebrates, relatively little is known about the effect of endocrine disruptors on marine invertebrates, mostly due to a poor understanding of invertebrate endocrine systems (Porte *et al.* 2006). In some mollusks and sponges, endocrine disruptors have been shown to interfere with key enzymatic pathways, leading to cellular damage (Wiens *et al.* 1999, Viarengo *et al.* 2000) and reproductive abnormalities (Sarojini *et al.* 1986, Wasson *et al.* 2000), including high incidence of imposex and blocked embryonic development. Diverse effects of estrogen mimics on invertebrates have been reported, including stimulated ovarian and/or oocyte development (Shoenmakers *et al.* 1981, Sarojini *et al.* 1986, Wasson *et al.* 2000), blocked embryonic development (Hathaway and Black 1969), altered enzymatic activities (Ghosh and Ray 1993a, 1993b), accumulation of proteins (Ghosh and Ray 1992, Wiens *et al.* 1999, Billinghurst *et al.* 2000), and cellular damage or cell death (Wiens *et al.* 1999, Viarengo *et al.* 2000). On the other hand, some studies have failed to detect effects of estrogen mimics on invertebrates (Hutchinson *et al.* 1999, Breitholtz and Bengtsson 2001, Pascoe *et al.* 2002). In one of the few studies on corals, two common Hawaiian coral species showed adverse effects from exposure to endocrine disruptors; *Montipora capitata* coral colonies showed reduced fecundity and *Porites compressa* displayed decreased skeletal growth rates (Tarrant *et al.* 2004).

4.5 Sea level Rise

Sea level rise is a unique marine stressor with important implications for the jurisdictions in the Western Pacific Region. Sea level rise alone might appear to be relatively unimportant to many marine ecosystems, but it has the potential to affect nearly all marine ecosystems through indirect effects and interactions with other stressors discussed in this report. Under current climate change predictions, sea level rise is expected to exacerbate many of the stressors described in this report.

Indirectly, sea level rise will displace large numbers of people and decrease food availability and security. Coastal inundation will destroy homes and other infrastructure, forcing many people to undertake coastal modifications or to relocate to higher ground or higher islands (for those living on atolls). These changing patterns in human density will alter patterns of marine resource use. Inundation and groundwater intrusions with salt water will degrade drinking water supplies and render low-lying agricultural lands unproductive (Rahman *et al.* 2009, Nicholls 2010, Chen *et al.* 2012), potentially increasing reliance and harvest pressure on fisheries (IPCC 2014).

Shallow water marine ecosystems will be directly affected through inundation with ocean water, altering salinities, depth, temperature, sedimentation, and nutrients. Sea level rise is expected to not only increase coastal erosion rates, but also nutrient loading (IPCC 2014), especially in areas where septic and cesspool systems are in use. In addition, municipal sewer systems that have aging infrastructure will become vulnerable to leaking.

Mangrove and seagrass ecosystems are expected to experience "coastal squeeze" (IPCC 2014) especially along urbanized coastlines. With little opportunity to migrate inland, mangrove trees will be inundated by rising seas and experience high mortality. Increased wave energy will result in less suitable habitat for seedling germination or sediment accretion, which is necessary to produce and/or maintain the substratum at the appropriate depth. Seagrass ecosystems are

Summary Stressor Table: Potential effects of chemical pollutants
Color reflects the relative severity of an adverse effect: green=mild, yellow=moderate, red=severe.

Ecosystem	Potential Effects
Intertidal	● Adverse effects vary by contaminant and by organism ● Intertidal areas particularly sensitive to hydrocarbons ● Potential to significantly alter species composition, abundance, and biomass of the assemblage
Mangrove Forests	● Adverse effects vary by contaminant and by organism ● Mangrove trees particularly sensitivity to hydrocarbons and herbicides, and less sensitivity to heavy metal ● Potential to significantly alter species composition, abundance, and biomass of the assemblage
Seagrass Beds	● Adverse effects vary by contaminant and by organism ● Potential to significantly alter species composition, abundance, and biomass of the assemblage ● Light oiling from hydrocarbons has potential “beneficial” effects on seagrass growth
Coral Reefs	● Adverse effects vary by contaminant and by organism ● Potential to significantly alter species composition, abundance, and biomass of the assemblage
Deep Reef Slopes	See *Coral Reefs*
Banks and Seamounts	See *Coral Reefs*
Deep Ocean Floor	● Effects poorly studied in deep ocean floor ecosystems, but likely vary by contaminant and by organism ● Increase atmospheric deposition associated with climate change and deep ocean mining are likely to be the primary source of future pollutants in the Western Pacific Region
Pelagic Environment	● Effects poorly studied in pelagic ecosystem, but likely vary by contaminant and by organism

expected to experience higher salinity and lower irradiance levels due to increase in turbidity because of coastal erosion (Scavia *et al.* 2002). For both mangroves and coastal seagrass beds, the rate of sea level rise, coupled with erosion, could outpace the ability of primary producers to maintain optimal depth for survival.

The direct effects of sea level rise on deeper marine ecosystems are expected to be smaller, although concern has been expressed about the ability of some coral and other slow growing organisms to maintain an optimal depth for photosynthesis. This concern is heightened when considering the effects of ocean acidification and temperature on calcification rates for many marine organisms, although most coral reefs seem to have kept pace with the recent sea level rise (Buddemeier and Smith 1988, Brown *et al.* 2011). Sea level rise is expected to exacerbate sedimentation rates, nutrient enrichment and pollution on coastal coral reefs.

Summary Stressor Table: Potential effects of sea level rise
Color reflects the relative severity of an adverse effect: green=mild, yellow=moderate, red=severe.

Ecosystem	Potential Effects
Intertidal	● Inundation and entire loss on low islands ● Increased coastal fortification in inhabited areas leading to changes in shoreline process ● Increase erosion, nutrient enrichment, influx of pollutants, etc., especially in urbanized areas
Mangrove Forests	● Inundation and entire loss on low islands and along urban/developed coastline on high island, where it is not possible for the mangrove to “retreat” ● Increased salinity altering mangrove species composition, with cascading effects through the ecosystem ● Increase erosion, nutrient enrichment, influx of pollutants, etc., especially in urbanized areas
Seagrass Beds	● Increased salinity and within bed erosion via increase water flow ● Lower irradiance because of increased turbidity, leading to lower photosynthetic rates and growth in seagrasses ● Altered water quality from coastal inundation ● Potential for a phase-shift toward an algal-dominated assemblage
Coral Reefs	● Altered water flow could affect the distribution of species ● Altered water quality from coastal inundation ● Potential for a phase-shift to an algal-dominated assemblage

Ecosystem	Potential Effects
Deep Reef Slopes	● Affects likely to be small ● Altered water quality from coastal inundation ● Potential for change in distribution of species and shift in lower depth limit
Banks and Seamounts	See *Coral Reefs* (shallow) and *Deep Reef Slopes* (deep)
Deep Ocean Floor	● Likely little or no effect
Pelagic Environment	● Likely little or no effect

5.0 Cumulative Effects

Under the MSA implementing regulations, each FMP must contain an evaluation of the potential adverse effects, both individually and cumulatively, of non-fishing activities on the function of EFH at an ecosystem or watershed scale. Cumulative effects are impacts on the environment that result from the incremental effect of an action when added to other past, present, and reasonably foreseeable future actions, regardless of who undertakes such actions (Council on Environmental Quality 1997). Cumulative effects can result from individually minor, but collectively significant actions taking place over a period of time, or from the cumulative and interactive effects of multiple actions (Figure 4).

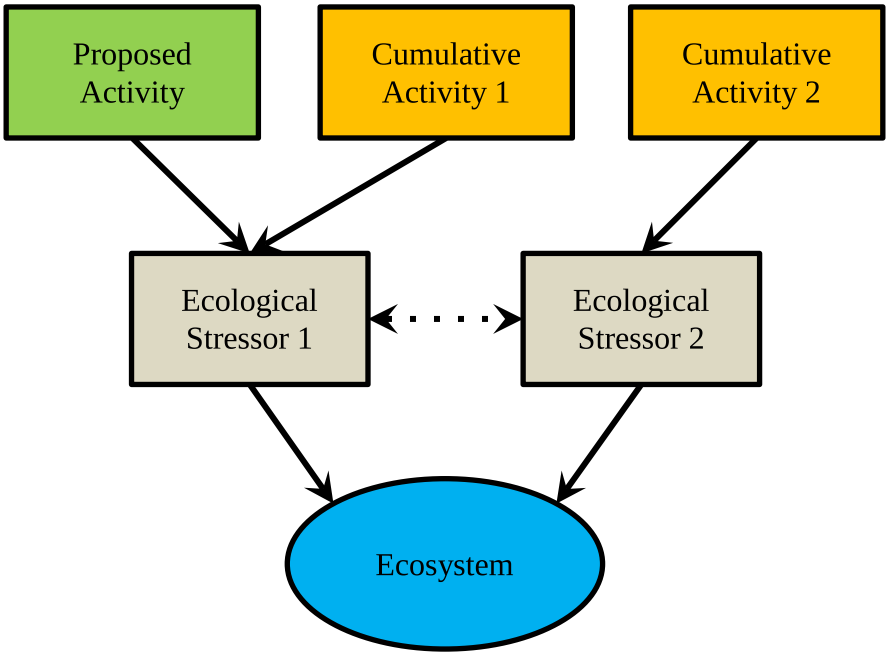

Figure 4. When assessing cumulative effects, the incremental effects of other past, present, and reasonably foreseeable future actions must be considered. In the flow diagram above, two types of cumulative effects are illustrated. In the first type, the Proposed Activity and Cumulative Activity 1 both act on Ecological Stressor 1, producing an additive effect on the ecosystem. While neither activity alone may have resulted in an adverse effect on the ecosystem, the two activities occurring together could. In the second type, Cumulative Activity 2 affects Ecological Stressor 2, which is known to interact with Ecological Stressor 1 (dotted arrow). This interaction, if synergistic in nature, would increase the total effect on the ecosystem beyond the additive effect of the two stressors, and thus heighten the adverse effects of the Proposed Activity beyond what would be expected if the Proposed Activity were implemented alone. However, if the interaction is antagonistic, it would produce a total effect on the ecosystem less than additive effect of the two stressors.

Evidence is increasing that the greatest environmental effects may result not from the direct effects of a particular activity, but from the combination of individually "minor" effects of multiple actions[2] concentrated in space ("space crowded") and/or time ("time crowded"). Assessing the cumulative environmental effects of an activity requires identifying from the complex networks of possible interactions those that substantially affect species and/or ecosystems, and then describing the response of the species and/or ecosystem to this environmental change. Predicting the effects of a stressor on an ecosystem is particularly difficult when many stressors of different types act in concert (NRC 1986).

Conceptually, cumulative effects involving multiple stresses can encompass three broad categories of interaction types (Crain *et al.* 2008). For the most common case involving two stressors, the resulting cumulative effect (CE_F) can be additive ($CE_F = E_A + E_B$), antagonistic ($CE_F < E_A + E_B$), or synergistic ($CE_F > E_A + E_B$). If two stressors show no interaction, their cumulative effects would be additive; that is, the effect of each stressor would act on the ecosystem in the same manner, as if the other stressor were not present. However, if two stressors interact, two scenarios are possible:

1) The stressors when co-occurring may produce a synergistic effect, whereby the presence of one stressor increases the effect of the other. This could result if a stressor acted on an organism to increase its susceptibility to the second stressor, thus producing a cumulative effect that is larger than what would be expected with no interaction.

2) The stressors when co-occurring produce an antagonistic effect, whereby the presence of one stressor reduces the effect of the other. For example, if a stressor acted on an organism to reduce susceptibility to the second stressor, thus producing a cumulative effect that is smaller than what would be expected with no interaction. An antagonistic interaction could be considered "beneficial" if the net effect of the two stressors together was smaller than the effect of the single stressor ($E_A + E_B < E_A$).

Given the complex interconnections among marine ecosystems, cumulative effects associated with human activities are expected to occur and to be potentially substantial and far-reaching. Thus, an assessment of cumulative effects must consider actions that may affect the ecosystem, regardless of where the action occurs and for a long enough period both into the past and into the future[3]. For example, actions potentially affecting a coastal coral reef should consider actions occurring in nearby seagrass, intertidal and mangal ecosystems that may also directly or indirectly affect the coral reef ecosystem when assessing the cumulative effects of an activity on the coastal reef. Selecting an appropriate time frame can be more challenging, but at minimum should attempt to include any projects previously conducted that have not recovered to their pre-activity condition and any future projects that would occur before the ecosystem has recovered

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[2] This is sometimes referred to as "nibbling" in the literature.

[3] For practical guidance, Hegmann *et al.* (1999) is good source for using "Scoping" to set appropriate spatial and temporal boundaries. The practical guide is available online: https://www.canada.ca/en/environmental-assessment-agency/services/policy-guidance/cumulative-effects-assessment-practitioners-guide.html

from the effects of the proposed activity (Hegmann *et al.* 1999). Failure to do so could result in an incorrect assessment of all the potential effects of an action and could result in an adverse effect on EFH.

Climate change is a reality, and the ocean is rapidly changing. A cumulative effects analysis must consider the changes to the marine environment that are expected to occur under our current climate trajectory. This is especially critical for any activity that will result in long-term effects on any marine ecosystem (*e.g.*, a sewage outfall, coastal road, waterbased energy production facility). Activities that produce long-term effects that are at present not detrimental to EFH, may become detrimental in the coming decades. Considering that many effects in marine ecosystems have long durations due to slow ecosystem recovery (*e.g.*, coral reefs), many activities proposed today, could result in significant and irreversible damage to EFH in coming decades. Without immediate action at the global level, marine ecosystems will continue to decline over the next half century (Hoegh-Guldberg *et al.* 2007, Cheung *et al.* 2009) and maintaining fishery sustainability will require tough decisions be made about human activities today (Cheung *et al.* 2009, Sumaila *et al.* 2011).

Many of the stresses identified in this report have the potential to interact, and often in ways that increase adverse effects on one or more ecosystems (Brown 1997, Negri and Hoogenboom 2011). For example, elevated seawater temperatures can cause coral bleaching, but the temperature threshold at which coral bleaching occurs is lowered under elevated nutrient conditions (Wooldridge 2009, Wolldridge *et al.* 2012), leading to a higher probability of bleaching in the presence of both thermal and nutrient stressors compared to a temperature increase alone. A cumulative effects analysis should account for such potential interactive effects.

Unfortunately, predicting the cumulative effect of multiple stressors is challenging (NRC 1986, Cooper and Shaete 2002, Bérubé 2007). In addition to the stressors themselves interacting, a species may respond similarly or differently to sets of stressors due to evolutionarily- or ecologically-derived tolerances (*e.g.*, coral colonies that have been bleached often show increased tolerance to later potential bleaching events), such that the interaction also depends upon which species are present, and their relevant history. Additionally, the response of an assemblage can differ due to changing functional roles and interactions among species (Crain *et al.* 2008, Breitburg *et al.* 1999), its species composition (and associated issues of redundancy and resilience), its connectivity to other ecosystems, and its environmental stochasticity (Breitburg *et al.* 1999). Temporal patterns of stressor occurrence (simultaneous vs. consecutive, frequency of stressor occurrence, etc.) and the intensity of the stressor (Relyea and Hoverman 2006) also influence the strength of the cumulative effects.

Fortunately, interactions among stressors have received more attention over the past 15 years, and enough information on potential interactions between and among multiple stressors now exist to allow for some understanding of when and where interactions can be expected to occur. Crain *et al.* (2008) reviewed over 200 studies examining cumulative effects for multiple stressors in intertidal and nearshore marine ecosystems to elucidate general patterns in cumulative stressor effects. The cumulative effects of any two stressors were distributed among all interaction types with 26% being additive, *i.e.*, no interaction, 36% synergistic and 38% antagonistic, and with all interaction types found to some degree for all stressors pairs with >5 studies (Figure 5 and Figure 6). In 62% of all cases, interactions between stressors resulted in an adverse effect on the species or ecosystem that was at least additive (Crain *et al.* 2008). In cases where a third stressor was considered, over two-thirds of the interaction became more negative, and the number of synergistic interactions increased to 66% of the cases. Thus, any activity or set of activities that significantly increases the negative effects of three or more stressors should be closely examined for adverse effects on EFH.

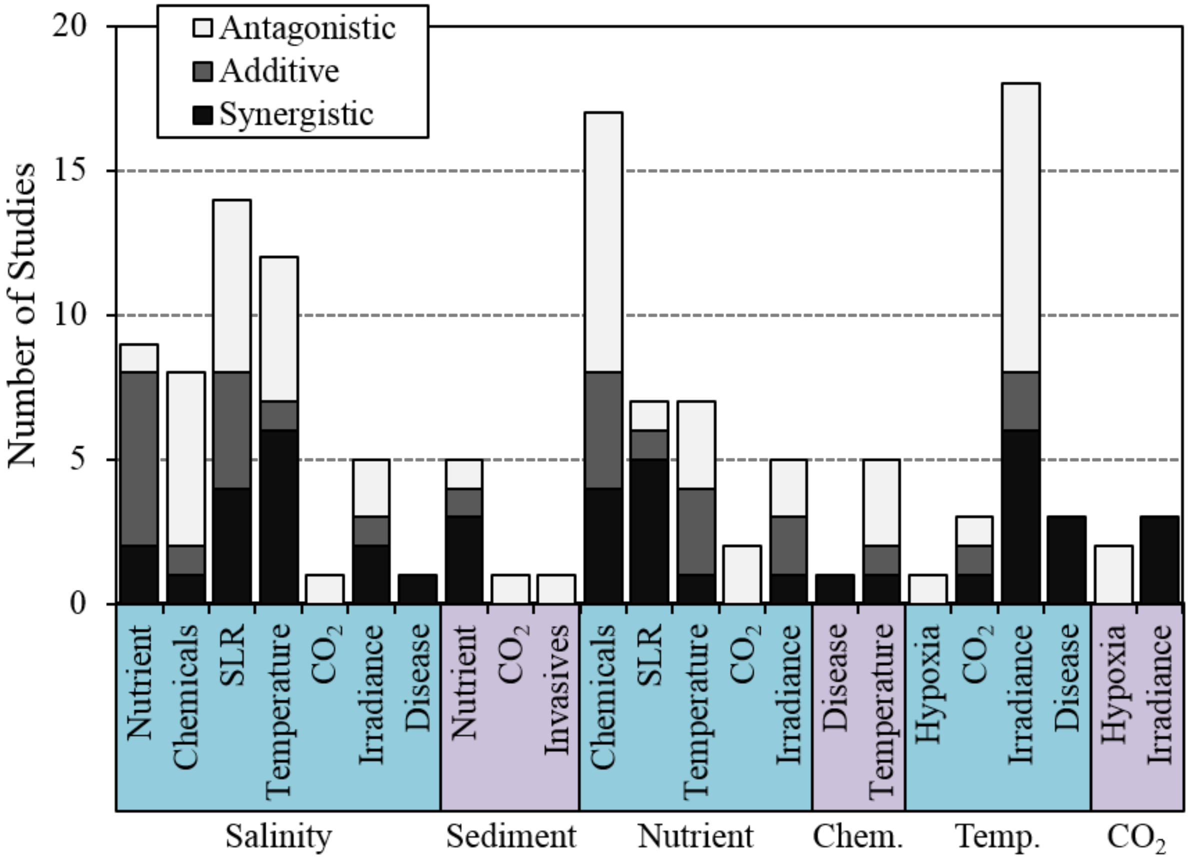
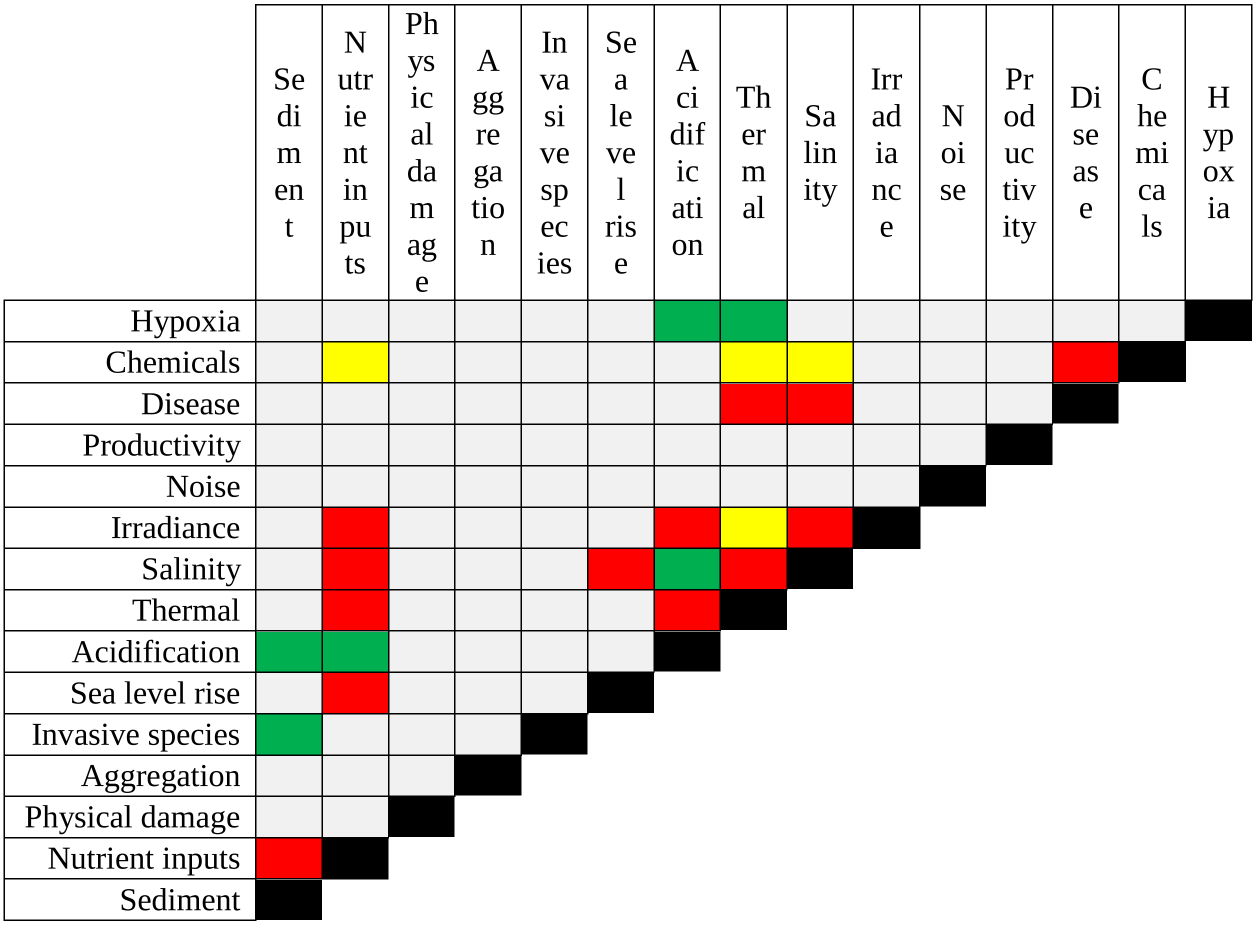


Figure 5. Frequency distribution of interaction types (additive, synergistic, and antagonistic) across stressor pairs. Stressor pairs are indicated within blocks on the x-axis that list one stressor horizontally (*e.g.*, salinity) with all stressor combinations listed vertically (*e.g.*, nutrient). See text for discussion of additive, synergistic, and antagonistic interactions. CO2=acidification, SLR=Sea Level Rise. Figure adapted from Crain *et al.* (2008).



	Sediment	Nutrient inputs	Physical damage	Aggregation	Invasive species	Sea level rise	Acidification	Thermal	Salinity	Irradiance	Noise	Productivity	Disease	Chemicals	Hypoxia
Hypoxia	⬜	⬜	⬜	⬜	⬜	⬜	🟩	🟩	⬜	⬜	⬜	⬜	⬜	⬜	⬛
Chemicals	⬜	🟨	⬜	⬜	⬜	⬜	⬜	🟨	🟨	⬜	⬜	⬜	🟥	⬛	
Disease	⬜	⬜	⬜	⬜	⬜	⬜	⬜	🟥	🟥	⬜	⬜	⬜	⬛		
Productivity	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬛			
Noise	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬛				
Irradiance	⬜	🟥	⬜	⬜	⬜	⬜	🟥	🟨	🟥	⬛					
Salinity	⬜	🟥	⬜	⬜	⬜	🟥	🟩	🟥	⬛						
Thermal	⬜	🟥	⬜	⬜	⬜	⬜	🟥	⬛							
Acidification	🟩	🟩	⬜	⬜	⬜	⬜	⬛								
Sea level rise	⬜	🟥	⬜	⬜	⬜	⬛									
Invasive species	🟩	⬜	⬜	⬜	⬛										
Aggregation	⬜	⬜	⬜	⬛											
Physical damage	⬜	⬜	⬛												
Nutrient inputs	🟥	⬛													
Sediment	⬛														

Figure 6. Interaction matrix for pairs of stressors acting on the marine ecosystems of the Western Pacific Region. Red = >50% of the studies show additive or synergistic interactions; yellow = <50% of the studies showed additive or synergistic interactions, green = studies showed only antagonistic interactions; gray = no data available; solid color = determination based on >5 studies; hatched color = determination based on <5 studies. Data from Crain *et al.* (2008).

6.0 Conservation and Enhancement Recommendation

The WPRFMC is tasked with describing ways to avoid, minimize, or compensate for the adverse effects to EFH and for promoting the conservation and enhancement of EFH. Activities that may result in significant adverse effects on EFH should be avoided when less environmentally harmful alternatives are available. If there are no alternatives, the adverse effects of these activities should be minimized to the extent practicable by employing conservation and enhancement recommendations.

For this report, a conservation and enhancement recommendation is a single practice or combination of practices that has been determined to be an effective and practicable means of preventing or reducing the effect of an activity on a stressor, or in reducing the magnitude of a stressor acting on an organism or the ecosystem. A best management practice (BMP) is a type of conservation and enhancement recommendation that includes generalized practices that can be employed across a range of activities with little modification. In contrast, some conservation and enhancement recommendations are specific to a project or location, and are not applicable across a range of activities. Due to the broad applicability of BMPs, they will be the focus of this report.

Non-fishing activities and other sources of stress act on organisms and ecosystems through stressors (see 3.0 Non-fishing Activities and Other Sources of Stress). BMPs can be applied at two different locations in the event chain (Figure 7):

- A BMP can reduce the effect of an activity on a stressor. For example, a road construction project may choose to narrow a road or re-route it around a hill, thus reducing the amount of earth moving that is required. A sewage treatment plant may choose to route grey water to agricultural fields instead of discharging it into the marine environment.

- Alternatively, a BMP can reduce the effect of the stressor on the organism or ecosystem. For example, a road construction project may erect sediment fencing along a stream bank to reduce the amount of sediment washing into the ocean. A sewage treatment plant may install a long diffuser system to promote dilution of nutrients over a wider area of the discharge site.

Ideally, BMPs that act at either position in the event chain can be recommended to avoid and minimize adverse effects to EFH. However, BMPs that act to reduce the effect of an activity on a stressor are preferable to those that reduce the effect of the stressor on an organism or ecosystem because the former addresses the root cause of the potential adverse effect. To be effective, a BMP must:

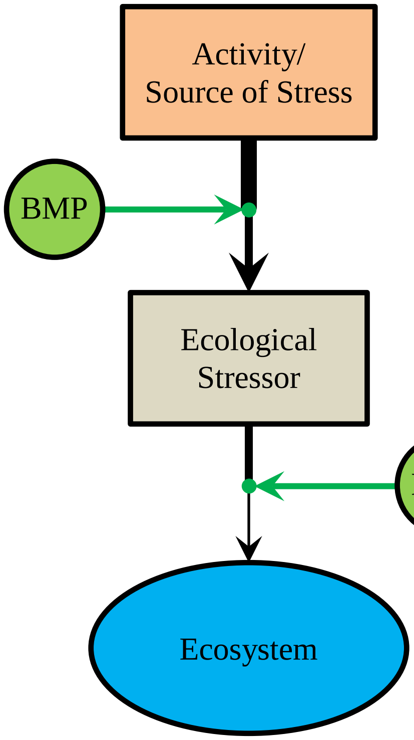


Figure 7. Conservation and enhancement recommendations, of which BMPs are common type, are practices intended to reduce the adverse effects of an activity on an ecosystem. BMPs can reduce the effect of an activity on the particular stressor (top) or reduce the effect of a stressor on an organism or ecosystem (bottom).

1) *Provide meaningful and measureable minimization of potential adverse effects.* BMPs are specifically developed to combat specific problems and often display a range of effectiveness associated with activity-specific factors. BMPs that have been demonstrated to be ineffective in providing meaning minimization of an adverse effects should not be recommended or implemented.

2) *Be properly selected and implemented.* BMPs are specifically developed to combat specific problems under certain conditions, and it is important that the correct BMP is selected for any given activity or stressor. Proper BMP selection and implementation is required or the BMP will be ineffective (Figure 8).

3) *Regularly inspected to insure its integrity.* Regular inspection of a BMP insures it is in proper working condition provides the opportunity to repair or adjust a BMP that has fallen into disrepair or is not working as effectively as it should. How frequently a BMP should be inspected depends on the specific conditions of the project and the BMP, but all BMPs should have a regular inspection schedule that is determined prior to implementation.

4) *Monitored to assess its effectiveness.* Few if any BMPs are 100% effective, but their effectiveness can vary considerably depending on the specifics of the project and the BMP. Monitoring the effectiveness of a BMP enables adaptive management to occur, and ineffective BMPs can either be reinstalled to improve performance or replaced with another BMP that may be better suited to the conditions and/or project.

The following BMPs can reduce the potential adverse effects of non-fishing activities on EFH. These BMPs have been identified from the scientific literature, recommendations made by federal and state/territorial/commonwealth agencies, and regulatory documents such as environmental impact statements. This list is not exhaustive, but represents commonly-employed, proven approaches as well as some common-sense recommendations to reduce adverse environmental effects. To facilitate selection, the BMPs have been organized into two tables: BMPs by activity category and BMPs by stressor. When recommending BMPs, BMPs from both tables should be considered, as appropriate.

The BMPs recommended by activity category generally contain recommendations on the design, placement and execution of activities with the intention of avoiding and minimizing potential adverse effects on EFH at the development stage of an activity.

The BMPs recommended by stressor type contain recommendations intended to reduce the effect of a specific stressor on EFH, either through reduction of the activities' effect on the stressor or

Figure 8. An inappropriately-selected BMP or one that is improperly-implemented is ineffective at reducing the adverse effect of a non-fishing activity on EFH: a) an inappropriately-selected oil control boom for the ocean conditions; b) an improperly-installed silt fence.

by reducing the effect of the stressor on the ecosystem. These BMPs are not necessarily specific recommendations for a single category of non-fishing activity, but could be broadly applied across a range of activities. These BMPs tend to address temporary issues (*e.g.*, construction-related runoff).

Summary BMP Table: BMPs by activity category

Activity Category	**BMPs**
General Considerations	● Areas of high diversity, abundance, and productivity or which serve as habitat for sensitive or important fishery species should be avoided to the maximum extent possible. ● Environmental surveying/sampling/monitoring should be developed with input from federal and state/territorial/commonwealth resource agencies. ● Biological surveys to determine species composition, abundance/biomass and productivity of an assemblage should be conducted using scientifically-rigorous survey designs and methods, and be completed prior to approval of any activity. ● All activities should reference latitude–longitude coordinates of the site so that information can be incorporated into Geographic Information Systems (GIS). ● All plans should have an adaptive management component, and a schedule for review and update.
Energy Production	● See BMPS for *Development/Construction (Land-based)* and *Development/Construction (Water-based)*
Mining	● Quarries should be placed outside the coastal zone where practicable and not adjacent to rivers. ● Measures to reduce/avoid runoff should be implemented, including; minimizing hard surfaces, minimize runoff through installing/preserving existing natural (and native) vegetation and/or building of a retention pond, and attempting to restore disturbed lands to as close to natural conditions, as possible, after no longer being mined. (HDOT 2008) ● Mining (coral and sand) should be avoided in coral reefs and other shallow water ecosystems (*i.e.*, those within the euphotic

Activity Category	**BMPs**
	zone). ● Deep ocean mining in areas of high biological diversity, abundance, and productivity (including the overlying surface waters) should be avoided. This is especially true if mining waste will be discharged into these waters due to the potential to expand the area of effect. ● For deep ocean mining, interaction of the collected with the seafloor should be kept to a minimum. Separation of the minerals from the sediment (and other debris) should occur as close as possible to the bottom to reduce water column discharge. (Sharma 2015) ● Deep ocean mining should be conducted in a "strip-wise" fashion, leaving alternate strips of undisturbed seafloor to promote recovery. (Sharma 2015) ● Surface discharge from deep ocean mining should be kept to a minimum and be dispersed across a wide area to dilute. Sufficient light should be allowed to penetrate the watercolumn for photosynthetic activity. Discharge of sediment at different levels in the water column should be encouraged. (Sharma 2015)
Land-based Aquaculture	● Facilities should be in upland areas and not in the coastal zone where practicable. (Howerton 2001) ● Tidally-influenced wetlands[4] should not be converted for aquaculture use. Wetland conversion reduces the functional value of the ecosystem, and potentially lacks a mechanism to control nutrient/waste exchange between the ponds and the coastal marine waters. (Howerton 2001) ● The siting of any aquaculture facility (regardless of type) should consider the size of the operation, the presence or

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[4] In Hawai'i, fishponds have been constructed in many estuarine and coastal areas, and are important native Hawaiian cultural and historical features. Where appropriate, existing fishponds should be restored, maintained, and managed for both their cultural and ecological value. This BMP is intended for non-historical/cultural activities or for activities that would represent a "new" structure/fishpond. In general, tidal wetlands should not be converted into ponds for aquaculture production when other viable alternatives exist.

Activity Category	BMPs
	absence of submerged vegetation and coral reef ecosystems, proximity of wild fish stocks, migratory patterns, competing uses, and hydrographic conditions. ● Operational plans should contain measures to prevent nutrient and waste disposal from reaching the marine environment without appropriate treatment. Where possible, water systems should recycle back into the pond or be used as grey water. (Ozbay *et al.* 2014, FDACS 2016) ● A plan to optimize feeding protocols to minimize nutrient accumulation at the site should be in place before operations start. Water quality thresholds should be established prior to the start of operations. (Ozbay *et al.* 2014) ● Chemical anti-foulants should not be used, instead, mechanical cleaning methods and air drying should be employed when practicable. (FDACS 2016) ● To the extent practicable, water intakes should be designed to avoid entrainment of flora and fauna. ● Non-native species that *could* adversely affect the ecological balance of an area (*i.e.*, have a reasonable probability of becoming invasive), should not be imported for aquaculture. A thorough scientific review and risk assessment should be undertaken by invasive species experts prior to any non-native species introduction. (FDACS 2016)
Development/Construction (Water-based)	● Dredging projects should be allowed only when water-dependent and when no other feasible and practicable alternative is available. ● Dredging activities should be sited in deep-water areas or designed in such a way as to minimize the amount of dredging and reduce the need for maintenance dredging. ● To the extent practicable, fill materials from dredging operations should be placed in an upland site. Unless unavoidable, fill should not be allowed in areas with mangal, subaquatic vegetation, coral reefs, or other areas of high productivity. (Johnson 2011) ● For clamshell dredges, a closed (environmental) bucket should be considered for use to reduce suspended sediment. Likewise, slower cycle times, single “bites” with the bucket, and no

Activity Category	**BMPs**
	bottom stockpiling should be implemented when practical. (Johnson 2011) ● If a hydraulic dredge (*e.g.*, cutterhead, suction, etc.) is to be used, selecting the appropriate type will minimize sediment loss. (Johnson 2011) ● The disposal of contaminated dredge material should not be allowed in EFH. ● Ocean disposal should be restricted to an approved, deep ocean disposal site. Currently, Hawai‘i and Guam have EPA approved ocean disposal sites. ((Johnson 2011, EPA 2016a, EPA 2016b) ● If the need for dredging (especially maintenance dredging) has been caused by excessive sedimentation from a land-based source, the source should be identified, and appropriate management actions to remediate the source should be proposed as part of the pre-dredging planning activities. Where legal and practicable, actions to remediate the upland sediment source should be part of the dredging project. ● Where practicable, pipelines (*e.g.*, wastewater, cooling discharge, etc.) should be elevated off the bottom using pedestals. (PBS&J 2008) ● Where possible, use horizontal directional drilling technology to install pipes, conduits, etc. instead of trenching or surface installation. (PBS&J 2008)
Development/Construction (Land-based/Coastal roads)	● Coastal hardening should only occur after all other alternatives have been determined not to be feasible or practicable. Alternative should include re-alignment of any road/activity to a different, upland location. ● Where practicable, bioengineering approaches should be used to protect altered shorelines. The alteration of natural, stable shorelines should be avoided as much as is practicable. ● For roads, parking lots, and other applicable structures, considering using oil/water or oil/grit separators, swales, constructed wetlands, etc., as part of the stormwater management to remove pollutants such as oils, grease, sand,

Activity Category	**BMPs**
	and grit from runoff. (HDOT 2007) ● Avoid upland and coastal earth-moving during the local rainy season. (USCRTF 2016) ● For coastal directional drilling activities, the volume of drill mud and the drill pressure should be monitored constantly to detect potential leaks ("frac-outs"). For the last 15-20 m of bore, seawater should be used in place of drill mud to prevent drill mud from entering the water. Any free-flowing slurry at the upland site during pull back and drilling should be properly contained and disposed of so that it does not enter marine waters. (PBS&J 2008, CALTRANS 2015)
Shipping/Boating	● The siting of any anchorage should consider the size and number of the vessels, the presence or absence of submerged aquatic vegetation and coral reef ecosystems, proximity of wild fish stocks, migratory patterns, competing uses, and hydrographic conditions. ● Where possible and practicable, permanent mooring facilities that reduce the activity's contact footprint with the bottom should be used. Contact footprint includes any anchors, chains, and/or lines that have the potential to adversely affect EFH. Potential adverse indirect effects associated with mooring buoys need to be considered. (Taratalos and Austin 2001, PADI 2005, USCRTF 2016)
Marine Debris	● No trash or other debris should be disposed of or otherwise allowed to enter the ocean. Ensure adequate trash receptacles with lids are available onsite or onboard vessels. ● All debris that enters the water because of the activity should be removed using means that do not cause additional damage to organisms such as coral (*e.g.*, dip net, snorkel, SCUBA, etc.). ● All loose articles (*e.g.*, clothing, towels on the deck, etc.) should be secured to prevent them blowing off or accidentally falling overboard.
Non-fishing, human activities (Military)	● A clear protocol to decrease sonar power when sensitive organisms are detected near a vessel should be in place. (USN 2008)

Activity Category	BMPs
	- No underwater detonations (training) should occur except within pre-approved areas designated for such activity. Detonations should be conducted using approved protocols, which should include protection measures for coral and other sensitive or important fishery species. (USN 2008)
"Waste" water discharge	- Where practicable, outfall structures should be placed sufficiently far offshore in areas of good mixing and use diffusers to promote dilution and reduce risk of discharged effluent from adversely affecting EFH. (Tate *et al.* 2016) - Where practicable, pipelines (*e.g.*, wastewater, cooling discharge, etc.) should be elevated off the bottom using pedestals. (PBS&J 2008) - Where possible, use horizontal directional drilling technology to install pipes, conduits, etc. instead of trenching or surface installation. (PBS&J 2008) - When practicable, wastewater effluent should be treated using the best available and practicable technology, including implementation of up-to-date methods to reduce discharges of biocides (*e.g.*, chlorine), endocrine disruptors, other toxic substances, and potential disease agents.

Summary BMP Table: BMPs by stressor type

Stressor	BMPs
Thermal	● Where practicable, discharges with different thermal or salinity characteristics than the receiving waters should be "treated" (*e.g.*, cooling or warming towers) prior to discharging, or should be discharged through means that will dilute the effluent to reduce the differential between it and the receiving body. (North Shore Consultants 2012, Tate *et al.* 2016) ● An effort should be made to ensure discharge temperatures (both heated and cooled effluent) do not exceed the thermal tolerance of the most sensitive organism[5] in the receiving waters.
Salinity	● Where practicable, discharges with different thermal or salinity characteristics than the receiving waters should be discharged through means that will dilute the effluent, reducing the differential between it and the receiving body. (Tate *et al.* 2016)
Irradiance	● Irradiance levels (PAR) should be monitored beneath any temporary structure that shades benthic, photosynthetic organisms. Prolonged exposure to levels below 35% of surface irradiance is likely to cause adverse effects on coral (see Erftemeijer *et al.* 2012 for more information). ● Temporary platforms or other structures that shade benthic photosynthetic organisms should be removed immediately upon completion of the activities that required them. ● Organisms, especially corals, beneath a temporary, shading structure should be monitored for condition, and if the organisms show signs of stress (*e.g.*, color change [especially paling], increased mucus production etc.), the temporary structure should be removed, if practicable and would not result in additional adverse effects. The structure can be returned once the organisms have sufficiently recovered.

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[5] This will be site-specific, but in most shallow water ecosystems this will likely be coral, which have been shown can bleach when temperatures exceed the summer maximum temperature by only a few degrees for a prolonged period (Baker *et al.* 2009). Deep slope ecosystems, especially deep sea corals, might be more sensitive given the lower natural variability in temperature.

Stressor	**BMPs**
Noise	● High amplitude noise should not exceed 150 decibel (dB) in a single strike. Noise more than 150 dB has been found to cause adverse behavioral effects in fish. High amplitude noise exceeding 180 dB has been shown to cause injury in fish. (Hastings 2002, WSDOT 2015) ● Where appropriate and practicable, bubble screens should be used to attenuate single strike noise. Curtains have been shown to reduce noise by 10-30 dB. (MacGillivray *et al.* 2007, WSDOT 2015)
Invasive species	● All vessels should undergo routine inspections for presence of non-native species growing on the hull of the vessel prior conducting work in a different area of operation. ● Any equipment that has been previously used in an area known to contain invasive species should be sanitized prior to its use elsewhere[6]. ● Any effluent from a facility containing non-native species (*e.g.*, aquaculture, aquarium, etc.) should be treated prior to discharge to ensure gametes/larvae[7] are not released into the marine environment. ● All facilities that contain live non-native species should have a thorough biosecurity plan. Staff should be trained in the execution of the plan to decrease the potential for release of non-native species or propagules into the environment.
Disease	● Where practicable, discharges that have the potential to contain biological pathogens (*e.g.*, sewage, aquaculture waste, etc.) should treated to neutralize disease-causing agents.

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[6] For more information on cleaning equipment, see NOAA's Preventing Invasive Species: Cleaning Watercraft and Equipment fact sheet available at: http://www.habitat.noaa.gov/pdf/best_management_practices/Cleaning%20of%20Watercraft%20and%20Equipment.pdf

[7] For example, see Tucker *et al.* (2012) for a discussion of using UV on non-native fish larvae to control invasive species.

Stressor	**BMPs**
FAD Effect	● Any structure using netting (*e.g.*, silt curtains, etc.) should have small enough webbing, and be installed to prevent entanglement by sensitive and fishery species. ● No marine life should be fed.
Physical damage	● No anchors, tools, or other equipment should be placed on any organism, especially coral. Preference should be to place anchors and spuds in soft-sediment only. ● No tools or materials should be dropped on the bottom during demolition and/or construction activities. ● Floating tow and anchoring lines should be used to prevent lines and cables from dragging in the water or on the bottom. All lines should be kept taut to reduce chance of entanglement of sensitive or fishery species. (Harnois *et al.* 2015) ● Where practicable, corals and other sensitive species that are likely to experience adverse effects, especially mortality, should be translocated/transplanted to a nearby, suitable location that is not likely to be impacted by the proposed or future projects. The condition of the relocated organisms should be monitored for at least two years[8]. (USCRTF 2016) ● All vessels should operate at "no wake/idle" speeds at all times while in water depths where the draft of the vessel provides less than a 2 m (6 ft.) clearance. All vessels should preferentially follow deep-water routes (*e.g.*, marked channels) whenever possible. If operating in shallow water, all vessels should employ a dedicated "lookout" to assist the pilot with avoiding large coral colonies and other benthic organisms that might extend up from the bottom.

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[8] Effective evaluation of translocation/transplantation success for coral has been a problematic because few efforts have monitored the relocated coral colonies sufficiently to determine long-term success. Given limited data, 18-24 months appears to be a critical threshold point (see figure 2 in Okuba and Omori 2001, USCRTF 2016), but most monotirng efforts only continue for about 12 months. While interspefic variability exists, survival after one year is often high, but after 18 months, colonies appear to expierence more mortality. Success appears to be correlated with the quality of the habitat to which the corals are moved (USCRTF 2016).

Stressor	BMPs
Sediment	- Runoff control measures, including silt screens, retention basins, swales, etc., should be installed prior to any activity that could result in sediment entering any waterbody[9]. The best land management practices should be used to control soil erosion. (HDOT 2008) - As appropriate and practicable, apply water and/or dust control measures to minimize wind transport of dust. (HDOT 2008) - Avoid upland and coastal earth-moving during the local rainy season. (USCRTF 2016) - All dredge/fill activities should be avoided to the extent possible during the coral broadcast spawning season (May-September in the northern hemisphere; Richmond and Hunter 1990). If dredge/fill window cannot be avoided, no activity should occur the 7 days before and 14 days after the full moon to avoid coral spawning[10]. This dredge/fill window may be narrowed based on site-specific spawning information. (PBS&J 2008) - Dredging activities should be conducted only under calm sea state conditions and with a slack tide. Depending on project-specific conditions, an incoming or outgoing tide might also be suitable for dredging. (PBS&J 2008) - Based on project-specific conditions, an appropriate turbidity

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[9] A thorough assessment of the effectiveness of BMPs is beyond the scope of this review, but such an assessment is a critical need to assist NMFS in making conservation and enhancement recommendations that will have positive benefits on EFH. For example, while silt fences are nearly universally employed for erosion control during earth moving activities and are often an effective BMP, they have been shown to exacerbate sediment erosion in some situations (Wear *et al.* 2013).

[10] Little is known about larval competency for most coral species. *Pocillopora damicornis* (lace coral) can be competent within one day of spawning, and *Seriatopora caliendrum* (birdsnest coral) in as little as five hours (Cumbo *et al.* 2013, Edmunds *et al.* 2013). Both are brooding species that produce larger propagules than broadcast spawing species. Even broadcast spawners appear to have relatively short minimum compentency periods. Broadcaster *Favites chinensis* (larger star coral) and brooder *Coelastrea (=Goniastrea) aspera* (lesser star coral) are competent within one to three days after spawning, and possess a relatively long maximum settlement-competency period of nearly 70 days (Nozawa and Harrison 2002). Corals, while likely competent to settle quickly, can remain competent for as much as 2-3 months (Harrison 2011). Given this relatively sparse data, 7-14 days following the full moon appears to be a reasonably cautious period because spawning occurs for several days after the full moon, providing 7-10 days for coral larvae to move from the site. This window can be revised as more information becomes available.

Stressor	**BMPs**
	barrier (*e.g.*, turbidity curtains, turbidity screens, gunderbooms, pneumatic screens, etc.) should be considered as a potential approach to reduce the adverse effects of suspended sediment resulting from dredge/fill operations. However, due to highly variable, and often overstated effectiveness, this method should not be the sole approach to sediment management. (PBS&J 2008, Johnson 2011, Cutroneo *et al.* 2014, Radermacher *et al.* 2015) ● Where practicable, corals and other sensitive species that are likely to experience adverse effects, especially mortality, should be translocated/transplanted to a nearby, suitable location that is not likely to be impacted by the proposed or future projects. The condition of the relocated organisms should be monitored for at least two years[11]. (USCRTF 2016)
Nutrients	● For construction projects near or in marine waters, nutrient and water quality "stop work" thresholds should be established prior to implementing any activity. If the thresholds are exceeding, work should be suspended immediately until conditions improve. The water quality monitoring should be conducted to determine if the threshold criteria have been exceeded. (PBS&J 2008)
Chemicals	● A spill contingency plan should exist for both the construction and operation (as appropriate) of a facility, and all employees should be familiar with its contents and be trained in how to respond to a spill. (HDOT 2013) ● Containment equipment and sufficient supplies to combat spills should be on-site at all facilities that handle hydrocarbons, chemicals and/or other hazardous substances. (HDOT 2013) ● To the maximum extent practicable, storage of hydrocarbons,

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[11] Effective evaluation of translocation/transplantation success for coral has been problematic because few efforts have monitored the relocated coral colonies sufficiently to determine long-term success. Given limited data, 18-24 months appears to be a critical threshold point (see figure 2 in Okuba and Omori 2001, USCRTF 2016), but most monitoring efforts only continue for about 12 months. While interspecific variability exists, survival after one year is often high, but after 18 months, colonies appear to experience more mortality. Success appears to be correlated with the quality of the habitat to which the corals are moved (USCRTF 2016).

Stressor	BMPs
	chemicals and/or hazardous substances should be in an area that would prevent spills from reaching marine environments. (HDOT 2013) ● All equipment should be properly maintained to prevent discharge of contaminants into marine waters. All equipment should be free of contaminants prior to use in or near the marine environment. ● Fueling of any equipment should be conducted in a dedicated area on land with control mechanisms to stop and spill from reaching the ocean. Seagoing vessels should be fueled at an approved location. (HDOT 2013) ● To the extent practicable, no heavy equipment should be driven or operated on reefs or tidal flats regardless of the tidal stage or exposure. ● Where practicable, an oil containment boom should be placed around mechanical equipment such as a dredge to contain any spilled oil or fuel. ● In the event of a spill, caution should be used when deploying and anchoring containment booms near reefs to prevent physical damage to corals and to prevent entangling marine species. ● The use of oil dispersants directly over shallow coral reefs and seagrass beds or near mangal and intertidal ecosystems should be avoided. ● The use of pesticides, herbicides, and fungicides in areas that would allow for their entry into marine environments should be avoided. ● Enzyme-based cleaners should be used instead of detergents, degreasers or chemicals.

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