



FEDERAL REGISTER

Vol. 79

Tuesday,

No. 241

December 16, 2014

Part IV

Department of Commerce

National Oceanic and Atmospheric Administration

50 CFR Parts 223 and 224

Endangered and Threatened Wildlife and Plants; 12-Month Finding for the Eastern Taiwan Strait Indo-Pacific Humpback Dolphin, Dusky Sea Snake, Banggai Cardinalfish, Harrison's Dogfish, and Three Corals Under the Endangered Species Act; Proposed Rule

DEPARTMENT OF COMMERCE**National Oceanic and Atmospheric Administration****50 CFR Parts 223 and 224**

[Docket No. 140707555-4999-01]

RIN 0648-XD370

Endangered and Threatened Wildlife and Plants; 12-Month Finding for the Eastern Taiwan Strait Indo-Pacific Humpback Dolphin, Dusky Sea Snake, Banggai Cardinalfish, Harrisson's Dogfish, and Three Corals Under the Endangered Species Act

AGENCY: National Marine Fisheries Service (NMFS), National Oceanic and Atmospheric Administration (NOAA), Commerce.

ACTION: Proposed rule; 12-month petition finding; request for comments.

SUMMARY: We, NMFS, have completed comprehensive status reviews under the Endangered Species Act (ESA) for seven foreign marine species in response to a petition to list those species. These seven species are the Eastern Taiwan Strait population of Indo-Pacific humpback dolphin (*Sousa chinensis*), dusky sea snake (*Aipysurus fuscus*), Banggai cardinalfish (*Pterapogon kauderni*), Harrisson's dogfish (*Centrophorus harrissoni*), and the corals *Cantharellus noumeae*, *Siderastrea glynni*, and *Tubastraea floreana*. We have determined that the Eastern Taiwan Strait Indo-Pacific humpback dolphin is not a distinct population segment and therefore does not warrant listing. We have determined that, based on the best scientific and commercial data available, and after taking into account efforts being made to protect the species, *Pterapogon kauderni*, and *Centrophorus harrissoni* meet the definition of a threatened species; and *Aipysurus fuscus*, *Cantharellus noumeae*, *Siderastrea glynni*, and *Tubastraea floreana* meet the definition of an endangered species. Therefore, we propose to list these six species under the ESA. We are not proposing to designate critical habitat for any of the species proposed for listing, because the geographical areas occupied by these species are entirely outside U.S. jurisdiction, and we have not identified any unoccupied areas that are currently essential to the conservation of any of these species. We are soliciting comments on our proposals to list the six species. We are also proposing related administrative changes to our lists of threatened and endangered species.

DATES: Comments on our proposed rule to list eight species must be received by February 17, 2015. Public hearing requests must be made by January 30, 2015.

ADDRESSES: You may submit comments on this document, identified by NOAA-NMFS-2014-0083, by any of the following methods:

- Electronic Submissions: Submit all electronic public comments via the Federal eRulemaking Portal. Go to www.regulations.gov/#!docketDetail;D=NOAA-NMFS-2014-0083. Click the "Comment Now" icon, complete the required fields, and enter or attach your comments.
- Mail: Submit written comments to, Lisa Manning, NMFS Office of Protected Resources (F/PR3), 1315 East West Highway, Silver Spring, MD 20910, USA.

Instructions: You must submit comments by one of the above methods to ensure that we receive, document, and consider them. Comments sent by any other method, to any other address or individual, or received after the end of the comment period, may not be considered. All comments received are a part of the public record and will generally be posted for public viewing on <http://www.regulations.gov> without change. All personal identifying information (e.g., name, address, etc.), confidential business information, or otherwise sensitive information submitted voluntarily by the sender will be publicly accessible. We will accept anonymous comments (enter "N/A" in the required fields if you wish to remain anonymous). Attachments to electronic comments will be accepted in Microsoft Word, Excel, or Adobe PDF file formats only.

You can obtain the petition, status review reports, the proposed rule, and the list of references electronically on our NMFS Web site at <http://www.nmfs.noaa.gov/pr/species/petition81.htm>.

FOR FURTHER INFORMATION CONTACT: Lisa Manning, NMFS, Office of Protected Resources (OPR), (301) 427-8403.

SUPPLEMENTARY INFORMATION:**Background**

On July 15, 2013, we received a petition from WildEarth Guardians to list 81 marine species as threatened or endangered under the Endangered Species Act (ESA). This petition included species from many different taxonomic groups, and we prepared our 90-day findings in batches by taxonomic group. We found that the petitioned actions may be warranted for 27 of the 81 species and announced the initiation

of status reviews for each of the 27 species (78 FR 63941, October 25, 2013; 78 FR 66675, November 6, 2013; 78 FR 69376, November 19, 2013; 79 FR 9880, February 21, 2014; and 79 FR 10104, February 24, 2014). This document addresses the findings for 7 of those 27 species: the Eastern Taiwan Strait population of Indo-Pacific humpback dolphin (*Sousa chinensis*), dusky sea snake (*Aipysurus fuscus*), Banggai cardinalfish (*Pterapogon kauderni*), Harrisson's dogfish (*Centrophorus harrissoni*), and the corals *Cantharellus noumeae*, *Siderastrea glynni*, and *Tubastraea floreana*. The remaining 20 species will be addressed in subsequent findings.

We are responsible for determining whether species are threatened or endangered under the ESA (16 U.S.C. 1531 *et seq.*). To make this determination, we consider first whether a group of organisms constitutes a "species" under the ESA, then whether the status of the species qualifies it for listing as either threatened or endangered. Section 3 of the ESA defines a "species" to include "any subspecies of fish or wildlife or plants, and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature." On February 7, 1996, NMFS and the U.S. Fish and Wildlife Service (USFWS; together, the Services) adopted a policy describing what constitutes a distinct population segment (DPS) of a taxonomic species (the DPS Policy; 61 FR 4722). The DPS Policy identified two elements that must be considered when identifying a DPS: (1) The discreteness of the population segment in relation to the remainder of the species (or subspecies) to which it belongs; and (2) the significance of the population segment to the remainder of the species (or subspecies) to which it belongs. As stated in the DPS Policy, Congress expressed its expectation that the Services would exercise authority with regard to DPSs sparingly and only when the biological evidence indicates such action is warranted.

Section 3 of the ESA defines an endangered species as "any species which is in danger of extinction throughout all or a significant portion of its range" and a threatened species as one "which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range." We interpret an "endangered species" to be one that is presently in danger of extinction. A "threatened species," on the other hand, is not presently in danger of extinction, but is likely to become so in the foreseeable future (that

is, at a later time). In other words, the primary statutory difference between a threatened and endangered species is the timing of when a species may be in danger of extinction, either presently (endangered) or in the foreseeable future (threatened).

When we consider whether species might qualify as threatened under the ESA, we must consider the meaning of the term “foreseeable future.” It is appropriate to interpret “foreseeable future” as the horizon over which predictions about the conservation status of the species can be reasonably relied upon. The foreseeable future considers the life history of the species, habitat characteristics, availability of data, particular threats, ability to predict threats, and the reliability to forecast the effects of these threats and future events on the status of the species under consideration. Because a species may be susceptible to a variety of threats for which different data are available, or which operate across different time scales, the foreseeable future is not necessarily reducible to a particular number of years. Discussions of the considerations for each relevant species are in the species-specific sections below.

Section 4(a)(1) of the ESA requires us to determine whether any species is endangered or threatened due to any one or a combination of the following five threat factors: The present or threatened destruction, modification, or curtailment of its habitat or range; overutilization for commercial, recreational, scientific, or educational purposes; disease or predation; the inadequacy of existing regulatory mechanisms; or other natural or manmade factors affecting its continued existence. We are also required to make listing determinations based solely on the best scientific and commercial data available, after conducting a review of the species’ status and after taking into account efforts being made by any state or foreign nation to protect the species.

In making a listing determination, we first determine whether a petitioned species meets the ESA definition of a “species.” Next, using the best available information gathered during the status review for the species, we complete a status and extinction risk assessment. In assessing extinction risk, we consider the demographic viability factors developed by McElhany *et al.* (2000) and the risk matrix approach developed by Wainwright and Kope (1999) to organize and summarize extinction risk considerations. The approach of considering demographic risk factors to help frame the consideration of extinction risk has been used in many

of our status reviews, including for Pacific salmonids, Pacific hake, walleye pollock, Pacific cod, Puget Sound rockfishes, Pacific herring, scalloped hammerhead sharks, and black abalone (see <http://www.nmfs.noaa.gov/pr/species/> for links to these reviews). In this approach, the collective condition of individual populations is considered at the species level according to four demographic viability factors: Abundance, growth rate/productivity, spatial structure/connectivity, and diversity. These viability factors reflect concepts that are well-founded in conservation biology and that individually and collectively provide strong indicators of extinction risk.

We then assess efforts being made to protect the species, to determine if these conservation efforts are adequate to mitigate the existing threats. Section 4(b)(1)(A) of the ESA requires the Secretary, when making a listing determination for a species, to take into consideration those efforts, if any, being made by any State or foreign nation to protect the species. We also evaluate conservation efforts that have not yet been fully implemented or shown to be effective using the criteria outlined in the joint NMFS/USFWS Policy for Evaluating Conservation Efforts (PECE; 68 FR 15100, March 28, 2003), to determine their certainty of implementation and effectiveness. The PECE is designed to ensure consistent and adequate evaluation of whether any conservation efforts that have been recently adopted or implemented, but not yet demonstrated to be effective, will result in recovering the species to the point at which listing is not warranted or contribute to forming the basis for listing a species as threatened rather than endangered. The two basic criteria established by the PECE are: (1) The certainty that the conservation efforts will be implemented; and (2) the certainty that the efforts will be effective. We consider these criteria in each species-specific section, as applicable, below. Finally, we re-assess the extinction risk of the species in light of the existing conservation efforts.

Status Reviews

Status reviews for the petitioned species addressed in this finding were conducted by NMFS OPR staff. Separate status reviews were done for the Eastern Taiwan Strait Indo-Pacific humpback dolphin (Whittaker, 2014), dusky sea snake (Manning, 2014), Banggai cardinalfish (Conant, 2014), Harrison’s dogfish (Miller, 2014), and the three corals (Meadows, 2014). In order to complete the status reviews, we compiled information on the species’

biology, ecology, life history, threats, and conservation status from information contained in the petition, our files, a comprehensive literature search, and consultation with experts. We also considered information submitted by the public in response to our petition findings. Draft status review reports were also submitted to independent peer reviewers; comments and information received from peer reviewers were addressed and incorporated as appropriate before finalizing the draft reports.

Each status review report provides a thorough discussion of demographic risks and threats to the particular species. We considered all identified threats, both individually and cumulatively, to determine whether the species responds in a way that causes actual impacts at the species level. The collective condition of individual populations was also considered at the species level, according to the four demographic viability factors discussed above.

The status review reports are available on our Web site (see **ADDRESSES** section). Below we summarize information from those reports and the status of each species.

Eastern Taiwan Strait Population of the Indo-Pacific Humpback Dolphin

The following section describes our analysis of the status of the Eastern Taiwan Strait (ETS) population of the Indo-Pacific Humpback dolphin, *Sousa chinensis*.

Species Description

The Indo-Pacific humpback dolphin, *Sousa chinensis* (Osbeck, 1765), within the genus *Sousa*, family Delphinidae, and order Cetacea, is broadly distributed. The taxonomy of the genus is unresolved and has historically been based on morphology, but genetic analyses have recently been used. Current taxonomic hypotheses identify *Sousa chinensis* as one of two (Jefferson *et al.*, 2001), three (Rice, 1998), or four (Mendez *et al.*, 2013) species within the genus. Each species is associated with a unique geographic range, though the species’ defined ranges vary depending on how many species are recognized. Rice (1998) recognizes *Sousa teuzii* in the eastern Atlantic, *Sousa plumbea* in the western Indo-Pacific, and *Sousa chinensis* in the eastern Indo-Pacific. Mendez *et al.* (2013) recently identified an as-yet unnamed potential new species in waters off of northern Australia. Currently, the International Union for Conservation of Nature (IUCN) and International Whaling Commission (IWC) Scientific Committee

recognize only two species, *Sousa chinensis* in the Indo-Pacific, and *Sousa teuzii* in the eastern Atlantic. Here, we follow a similar two-species taxonomy in our consideration of the genus and identification of the species *Sousa chinensis*. Under that taxonomy, *Sousa chinensis*' range includes nearshore tropical and subtropical habitats in southern Africa, the Indian Ocean, North Australia, southern mainland China, Hong Kong, and Taiwan (Jefferson *et al.*, 2001; Mendez *et al.*, 2013). We chose to follow a two-species taxonomy as it provides the clearest genetic, morphological, and geographic delineation of the species and is well supported by the current data available. While growing genetic and phylogeographic evidence suggests that *Sousa chinensis* is associated with further genetic subdivisions, more data are needed to clarify the taxonomy and delineate the geographic boundaries and ranges of these additional genetic units (Cockroft *et al.*, 1997; Jefferson *et al.*, 2004b; Frère *et al.*, 2008; Frère *et al.*, 2011; Lin *et al.*, 2012; Mendez *et al.*, 2013).

The Indo-Pacific humpback dolphin is easy to distinguish from other dolphin species in its range, as it is characterized by a robust body, a long, distinct beak, a short dorsal fin atop a wide dorsal hump, and round-tipped, broad flippers and flukes (Jefferson *et al.*, 2001). The Indo-Pacific humpback dolphin is medium-sized, up to 2.8 m in length, weighing 250–280 kg (Ross *et al.*, 1994). Morphological plasticity exists among populations of the species and is correlated with their geographic distributions (Ross *et al.*, 1994). For example, the Eastern Taiwan Strait population, which occurs at the eastern portion of the species' range, has a short dorsal fin with a wide base; the base of the fin measures 5–10 percent of the body length and slopes gradually into the surface of the body. This differs from individuals in the western portion of the range, which have a larger hump that comprises about 30 percent of body width, and forms the base of an even smaller dorsal fin (Ross *et al.*, 1994). Males and females from the Pearl River Estuary population, and in other populations of Southeast Asia, do not exhibit sexual dimorphism in size, growth patterns, or morphology (Jefferson *et al.*, 2001; Jefferson *et al.*, 2012). In contrast, individuals from South Africa exhibit sexual dimorphism in terms of size and dorsal hump morphology (Ross *et al.*, 1994; Karczmarski *et al.*, 1997).

The species occurs in a range of nearshore habitats, including estuaries, mangroves, seagrass meadows, coastal

lagoons, and sandy beaches (Ross *et al.*, 1994). In Thailand, Malaysia, and Indonesia, nearshore ecosystems are associated with tropical seagrass, coral, and mangrove lagoons (Beasley *et al.*, 1997; Smith *et al.*, 2003; Adulyanukosol *et al.*, 2006; Jaroensutasinee *et al.*, 2011; Cherd sukai *et al.*, 2013). In India, the species is associated with nearshore habitat consisting of mangroves, corals, and tidal mudflat, heavily influenced by monsoons that regulate the influx of freshwater to the system (Sutaria *et al.*, 2004). The coast of mainland China is thought to host at least eight populations of the species, primarily occurring in estuarine systems at the mouths of large rivers (Jefferson *et al.*, 2001; Jefferson *et al.*, 2004a). Two coastal Chinese populations, in close proximity to the population in the Eastern Taiwan Strait, are relatively well-studied. These are the Pearl River Estuary/Hong Kong population and the Jiulong River Estuary/Xaimen population, both of which depend upon ecosystem productivity associated with the nutrient output supplied by large rivers (Chen *et al.*, 2008; Chen *et al.*, 2010).

The Eastern Taiwan Strait population of *Sousa chinensis* (henceforth referred to as the ETS humpback dolphin), for which we were petitioned, was first described in 2002 during an exploratory survey of coastal waters off of western Taiwan (Wang *et al.*, 2004). Prior to these coastal surveys, there are few records mentioning the species in this region, save two strandings, a few photographs, and anecdotal reports (Wang, 2004), so their history in the region is unclear. Since the first survey in 2002, researchers have confirmed their year-round presence in the Eastern Taiwan Strait (Wang *et al.*, 2011), inhabiting estuarine and coastal waters of central-western Taiwan.

The ETS humpback dolphin habitat is most similar to that of the populations located off the coast of mainland China. Individuals of the ETS humpback dolphin population are thought to be restricted to water less than 30 meters deep, and most observed sightings have occurred in estuarine habitat with significant freshwater input (Wang *et al.*, 2007b). Across the ETS humpback dolphin habitat, bottom substrate consists of soft-sloping muddy sediment with elevated nutrient inputs, primarily influenced by river deposition (Sheehy, 2010). These nutrient inputs support high primary production, which fuels upper trophic levels, contributing to the dolphin's source of food (Jefferson, 2000).

The Indo-Pacific humpback dolphin is considered a generalist and

opportunistic piscivore (Barros *et al.*, 2004). As is common to the species as a whole, the ETS population uses echolocation and passive listening to find its prey. While little is known about the specific diet and feeding of the ETS population, diet can be inferred from that of other humpback dolphin populations (Barros *et al.*, 2004; Chen *et al.*, 2009). In Chinese waters off Hong Kong, the species consumes both bottom-dwelling and pelagic fish species, including croakers (Sciaenidae), mullets (Mugilidae), threadfins (Polynemidae), and herring (Clupeidae) (Barros *et al.*, 2004). Part of the feeding strategy for this population may be to induce shoaling of fish by physically corralling them, allowing individuals to forage and feed successfully, even within murky nearshore waters (Sheehy, 2009). In general, the prey species of the humpback dolphin include small fish which are generally not commercially valuable to local fisheries (Barros *et al.*, 2004; Sheehy, 2009).

Little is known about the life history and reproduction of ETS humpback dolphin. In some cases, comparison of the ETS population with other populations may be appropriate, but one needs to be cautious about making these comparisons, as environmental factors such as food availability and habitat status may affect important rates of reproduction and generation time in different populations. A recent analysis of life history patterns for individuals in the Pearl River Estuary (PRE) population is the best proxy for the ETS population. Like the ETS population, the PRE population inhabits estuarine and freshwater-influenced environments in similar proximity to anthropogenic activity (Jefferson *et al.*, 2012). Maximum longevity for the PRE population is estimated to be greater than 38 years (Jefferson *et al.*, 2012). Evidence from multi-year photo-analysis of the ETS population demonstrated that adult survivorship is high, 0.985, suggesting that this population also has a relatively long lifespan (Wang *et al.*, 2012). In general, it is inferred that the population has long calving intervals, between 3 and 5 years (Jefferson *et al.*, 2012). Gestation lasts 10–12 months (Jefferson *et al.*, 2012). Weaning may take up to 2 years, and strong female-calf association may last 3–4 years (Karczmarski *et al.*, 1997; Karczmarski, 1999). Peak calving activity most likely occurs in the warmer months, but exact peak of calving time may vary geographically (Jefferson *et al.*, 2012). Age at sexual maturity is late, estimated at between 12 and 14 years (Jefferson *et al.*, 2012).

DPS Analysis

The following section provides our analysis, based on the best available science and the DPS Policy, to determine whether the ETS humpback dolphin population qualifies as a DPS of the taxon.

Discreteness

The Services' joint DPS Policy states that a population segment of a vertebrate species may be considered discrete if it satisfies either one of the following conditions: (1) It is markedly separated from other populations of the same taxon as a consequence of physical, physiological, ecological, or behavioral factors (quantitative measures of genetic or morphological discontinuity may provide evidence of this separation); or (2) it is delimited by international governmental boundaries within which differences in control of exploitation, management of habitat, conservation status, or regulatory mechanisms exist that are significant in light of section 4(a)(1)(D) of the ESA (61 FR 4722; February 7, 1996).

Individuals from the ETS population exhibit pigmentation that differs significantly from nearby populations along the mainland coast of China, and evidence suggests that pigmentation varies geographically across the species' range (Jefferson *et al.*, 2001; Jefferson *et al.*, 2004a; Wang *et al.*, 2008). Across the species, pigmentation changes as individuals mature. When young, dolphins appear dark grey with no or few light-colored spots; as they age, they transform to mostly white (appearing pinkish), as dark spots decrease with age. In particular, the developmental transformation of pigment differs significantly between ETS and nearby Chinese humpback dolphin populations; specifically, the spotting intensity (density of spots) on the dorsal fin of the ETS population is significantly greater than that of four mainland Chinese populations, including the other nearby populations in the Pearl River Estuary and Jiulong River estuaries (Wang *et al.*, 2008). Significantly greater spotting intensity on the dorsal fin of the ETS population is consistent, regardless of age (Wang *et al.*, 2008). Further, the ETS humpback dolphin never loses the dark dorsal fin spots completely, as has been observed in older individuals of other humpback dolphin populations (Wang *et al.*, 2008). In contrast, dorsal fins of Chinese populations are strikingly devoid of spots, compared to their bodies, throughout most of their lives, except when they are very young or very old (Wang *et al.*, 2008). These differences in

pigmentation can be used to reliably differentiate between the ETS humpback dolphin and nearby Chinese populations (Wang *et al.*, 2008). Thus, we consider these significant differences in pigmentation of the ETS humpback dolphin as evidence of its discreteness.

Several researchers have suggested that the ETS population of the humpback dolphin is physically and geographically isolated from other populations, based on the fact that individuals have not been observed crossing or to have crossed the Strait of Taiwan, despite repeated surveys of Chinese and Taiwanese populations using photo-identification techniques (Wang *et al.*, 2004; Wang *et al.*, 2007b; Chen *et al.*, 2010; Wang *et al.*, 2011; Wang *et al.*, 2012). For instance, a detailed analysis of more than 450 individually-recognizable dolphins catalogued for Taiwanese and Chinese populations revealed no matches among them (Wang *et al.*, 2008). Movement of *Sousa chinensis* is thought to be limited to shallow water and nearshore habitat (Karczmarski *et al.*, 1997; Hung *et al.*, 2004). Water depth and fast-moving currents within the Eastern Taiwan Strait are thought to isolate the ETS population from Chinese populations, despite their relatively close geographic proximity (Wang *et al.*, 2004; Wang *et al.*, 2008; Wang *et al.*, 2011; Wee *et al.*, 2011; Wang *et al.*, 2012). In fact, the ETS population has never been observed in waters greater than 30 meters depth (Wang *et al.*, 2007b). Evidence suggests that the ETS population of the humpback dolphin has a narrow home range, and does not migrate seasonally or mix with Chinese populations (Wang *et al.*, 2011). The population has been shown to inhabit the shallow, narrow habitat on the western coast of Taiwan throughout the year, and exhibits strong site fidelity (Wang *et al.*, 2011).

The evidence for geographic isolation is based on limited survey data collected since 2002, which focused only on nearshore waters at certain times of year and did not survey the Strait waters between mainland China and Western Taiwan (Wang *et al.*, 2004; Wang *et al.*, 2011; Wang *et al.*, 2012). Thus, the possibility for Indo-Pacific humpback dolphin migration or emigration across the Strait cannot be eliminated entirely. However, the best available scientific information indicates that the species is found primarily in shallow nearshore habitat, and the ETS population has never been observed in waters greater than 30 meters, and thus migration or emigration across the deeper Strait is thought to occur rarely, if ever.

The best available data suggest that the ETS humpback dolphin population is discrete from all other populations of the species based on its morphological differences. Although limited, the best available data also suggest that the ETS humpback dolphin population is geographically isolated from other populations. The morphological differences and geographic isolation set this population apart from other populations of the Indo-Pacific humpback dolphin, and thus, we conclude that the ETS humpback dolphin population meets the discreteness criterion of the DPS Policy.

Significance

When the discreteness criterion is met for a potential DPS, as it is for the ETS humpback dolphin population, the second element that must be considered under the DPS Policy is the significance of the DPS to the taxon as a whole. Significance is evaluated in terms of the importance of the population segment to the taxon to which it belongs, in this case the species *Sousa chinensis*. Some of the considerations that can be used under the DPS Policy to determine a discrete population segment's significance to the taxon as a whole include: (1) Persistence of the population segment in an unusual or unique ecological setting; (2) evidence that loss of the population segment would result in a significant gap in the range of the taxon; and (3) evidence that the population segment differs markedly from other populations of the species in its genetic characteristics.

The ETS humpback dolphin population occurs in an ecological setting similar to populations occurring along the coast of mainland China, and many features of its habitat and ecology are similar to those of populations throughout the range of the species, as discussed above. Throughout its range, the Indo-Pacific humpback dolphin is consistently associated with coastal river output and is found in shallow nearshore waters (Jefferson *et al.*, 2001). It displays no apparent preference for clear or turbid waters (Karczmarski *et al.*, 2000). The habitat and ecosystem use of the species differ in some ways geographically, but evidence suggests that the dolphin is an opportunistic piscivore, and thus does not exhibit unique or restricted feeding ecology across its range (Jefferson *et al.*, 2001).

In Thailand, Malaysia, and Indonesia, the species occurs in tropical seagrass, coral, and mangrove lagoons not present in ETS humpback dolphin habitat (Beasley *et al.*, 1997; Smith *et al.*, 2003; Adulyanukosol *et al.*, 2006; Jaroensutasinee *et al.*, 2011; Chersukjai

et al., 2013). In India, the species is associated with nearshore habitat consisting of mangroves, corals, and tidal mudflat, heavily influenced by monsoons that regulate the influx of freshwater to the system (Sutaria *et al.*, 2004). The ETS humpback dolphin habitat is most similar to that of coastal Chinese populations, with more temperate water, soft muddy substrate, and consistent input from river systems. The ETS humpback dolphin habitat differs from the habitat occupied by mainland Chinese populations in some ways, with nearby rivers generally smaller than those in mainland China, and with warmer waters in the winter due to the influence of the Kuroshio Current, which periodically moves into the Strait of Taiwan (Chern *et al.*, 1990; Jan *et al.*, 2002; Wang *et al.*, 2008). However, feeding ecology, prey availability, and prey preference are thought to be similar in mainland China and Taiwan (Barros *et al.*, 2004; Wang *et al.*, 2007a), so these small differences in habitat do not seem to have significant effects on the species' ecology.

The presumed habitat of the ETS humpback dolphin is narrower in offshore width than that of other studied populations of the taxon. For instance, the ETS population is thought to inhabit a small area of coastal shallow waters within 3 km from the shore (Wang *et al.*, 2007b). In contrast, Chinese populations inhabit a broader shallow area ranging tens of kilometers offshore, where dolphins can range farther from the coastline without moving into deeper water (Hung *et al.*, 2004; Chen *et al.*, 2011). While the ETS population exhibits some behavioral differences, such as increased cooperative calf-rearing and social connectivity, as compared to Chinese populations (Dungan *et al.*, 2011), it is uncertain whether or not these differences are adaptive or facultative, and simply based on the population's low abundance. Thus, insufficient evidence exists to suggest significant differences in the dolphin's ecology or adaptation have derived from the differences in the physical parameters of its environment. Therefore, differences in the habitat and ecological setting of the ETS humpback dolphin do not set it apart from the rest of the taxon, and do not appear to relate to significant selection pressures affecting the population's foraging, behavior, or ecology.

There is no evidence to suggest that loss of the ETS humpback dolphin population would result in a significant gap in the range of the taxon. The ETS humpback dolphin population constitutes a small and peripheral

portion of the entire range of the species, and its loss would not inhibit population movement or gene flow among other populations of the species (Lin *et al.*, 2012). The ETS humpback dolphin is distributed throughout only 512 square kilometers of coastal waters off Western Taiwan; this small range is not geographically significant in comparison to the taxon's range throughout the coastal Indo-Pacific and Indian Oceans.

There are no data to show that the genetic characteristics of the ETS humpback dolphin population differ markedly from other populations in a significant way. While pigmentation of the ETS population is significantly different from other populations within the taxon (Wang *et al.*, 2008), whether the pattern is adaptive or has genetic underpinnings is unknown. In other cetacean species, differences in pigmentation have been hypothesized to relate to several adaptive responses, allowing individuals to hide from predators, communicate with conspecifics (promoting group cohesion), and disorient and corral prey (Caro *et al.*, 2011). However, the differences in ETS humpback dolphin pigmentation may be a result of a genetic bottleneck from the small size of this population (less than 100 individuals) and the possibility that it represents a single social and/or family group. Such small populations are more heavily influenced by genetic drift than large populations (Frankham, 1996). Insufficient data exist to determine whether significant differences in ETS humpback dolphin pigmentation relate to the functional divergence of the population, or are simply a product of genetic drift and a genetic bottleneck. The best data available thus lead us to conclude that loss of the ETS humpback dolphin population would not result in significant loss of overall genetic or ecological diversity of the taxon as a whole.

DPS Conclusion and Proposed Determination

According to our analysis, the ETS humpback dolphin population is considered discrete based on its unique pigmentation patterns, which set it apart morphologically from the rest of the taxon, and evidence for its geographic isolation. However, while discrete, the ETS humpback dolphin population does not meet any criteria for significance to the taxon as a whole. The ecological setting it occupies is similar to that of the rest of the species, loss of the population would not constitute a significant gap in the taxon's extensive range, and no genetic or other data have

demonstrated that the population makes a significant contribution to the adaptive, ecological, or genetic diversity of the taxon. As such, based on the best available data, we conclude that the ETS humpback dolphin population is not a DPS and thus does not qualify for listing under the ESA. This is a final action, and, therefore, we do not solicit comments on it.

Dusky Sea Snake

The section below presents our analysis of the status of the dusky sea snake, *Aipysurus fuscus*. Further details can be found in Manning (2014).

Species Description

The dusky sea snake, *Aipysurus fuscus*, is a species within the family Elapidae, which is a very diverse family of venomous snakes. The genus *Aipysurus* contains seven species, six of which are restricted to Australasian waters. The dusky sea snake is brown, blackish-brown, or purplish-brown with wide ventral scales and diamond-shaped body scales that are smooth and imbricate (*i.e.*, overlapping). There are generally 19 scale rows around the neck, 19 around the mid-body, and 155 to 180 ventral scales (Rasmussen, 2000). The dusky sea snake is completely aquatic and, like all sea snakes, has a paddle-like tail for swimming. Its maximum total length is about 90 cm (Rasmussen, 2000). Growth rates for the dusky sea snake have not been documented, but reported growth rates for other sea snakes range from 0.07–1.0 mm per day and decline with age (Heatwole, 1997). The maximum lifespan for dusky sea snakes has been assumed to be about 10 years, and age at first maturity has been assumed to be about 3–4 years (Lukoschek *et al.*, 2010). Generation length is thought to be approximately 5 years (Lukoschek *et al.*, 2010).

Despite its aquatic existence, and like all reptiles, the dusky sea snake lacks gills and must surface to breathe air. Dive durations vary by species, but most sea snakes typically stay submerged for about 30 minutes, and some for up to 1.5–2.5 hours (Heatwole and Seymour, 1975). Maximum dive depth for dusky sea snakes is unknown, but co-occurring members of this genus are considered “shallow” and “intermediate” depth species that dive no deeper than 20 m or 50 m, respectively (Heatwole and Seymour, 1975).

The dusky sea snake is viviparous, meaning embryos develop internally and young undergo live birth. Because this species never ventures on land, mating occurs at sea and young are born alive in the water. Within the genus *Aipysurus*, the number of young per

brood is small, usually less than four, and young are relatively large at birth (Cogger, 1975). Timing and seasonality of the dusky sea snake's breeding cycles are unknown, and very little is known about the juvenile life stage.

The dusky sea snake preys mainly on labrid (*e.g.*, wrasses) and gobiid (*e.g.*, gobies) fishes, and to a lesser extent, fish eggs (McCosker, 1975). Food competition among sympatric sea snakes is thought to be minimal, based on examinations of diet composition for sympatric sea snakes (McCosker, 1975; Voris and Voris, 1983). Feeding behavior of dusky sea snakes has not been thoroughly investigated; however, during surveys at Ashmore Reef, Australia, Guinea and Whiting (2005) commonly saw dusky sea snakes over sand bottom habitat and watched one snake actually force its head and about 15 percent of its body into the sand. However, because it emerged without a prey item (Guinea and Whiting, 2005), it is unclear whether this was foraging or some other behavior. Like their terrestrial relatives, sea snakes swallow their prey whole and therefore must have some strategy for subduing them. McCosker (1975) hypothesized that the highly toxic venom of sea snakes is probably more of a feeding adaptation than a defensive one.

The dusky sea snake is a benthic, coral reef-associated species endemic to several shallow emergent reefs of the Sahul Shelf off the coast of Western Australia in the Timor Sea, between Timor and Australia. These reefs are relatively isolated and lie at the edge of the continental shelf over several hundred kilometers from the mainland. The dusky sea snake has been reported to occur at Ashmore, Scott, Seringapatam, and Hibernia Reefs and Cartier Island; however, individual surveys have not consistently recorded dusky sea snakes at all of these locations. For example, in transect surveys conducted by Minton and Heatwole (1975) over several weeks during December 1972 and January 1973 at Ashmore, Scott, and Hibernia Reefs and Cartier Island, dusky sea snakes were recorded at Scott and Ashmore reefs only. Extensive surveys conducted more recently at Ashmore Reef, where dusky sea snakes were once relatively common, have located no specimens (Guinea, 2013; Lukoschek *et al.*, 2013). Lukoschek *et al.* (2010) estimated that the area of occurrence of dusky sea snakes is probably less than 500 km².

During their surveys, Minton and Heatwole (1975) observed dusky sea snakes in shallow water (<10 m) as well as in the 12 to 25 m depth-zone. They were observed in areas of moderate to

heavy coral growth, but they were also observed to congregate in sandy-bottomed gullies and channels (Minton and Heatwole, 1975). Home-range size and site fidelity of individual dusky sea snakes has not been evaluated.

However, a short-term (6–9 days), telemetry study on the sympatric olive sea snakes (*A. laevis*) and a long-term (8-year), mark-recapture study on the turtle-headed sea snake (*Emydocephalus annulatus*) suggest that home-ranges of sea snakes are small, movement of adults is very limited, and longer-distance dispersal may be due mainly to passive transport, such as by currents and storms (Burns and Heatwole, 1998; Lukoschek and Shine 2012). While it is very plausible that adult *A. fuscus* are similar to these other species, research to evaluate adult and juvenile *A. fuscus* habitat use and movement is needed.

Sea snakes typically have patchy distributions and can be found in very dense aggregations in certain locations within their ranges (Heatwole, 1997). This patchiness complicates efforts to understand habitat use patterns, as seemingly suitable habitat can remain unoccupied. On a smaller spatial scale, distributions of sea snake fauna on Australian reefs appear to be influenced by water depth, substrate type, and feeding strategies (McCosker, 1975; Heatwole, 1975b). Other biotic factors, such as limited juvenile dispersal, may also contribute to the observed patchy distributions (Lukoschek *et al.*, 2007a). Overall, however, causative factors for observed distributions are not completely understood.

Population Abundance, Distribution, and Structure

There are no historical or current population estimates for the dusky sea snake. However, multiple reefs have been surveyed repeatedly, and although survey methodologies have varied, the data provide some indication of population trends for some locations. For Ashmore Reef in particular, the survey data provide a strong indication of severe population decline and possible extirpation. Older surveys dating from 1972 to 2002 by various researchers indicate that the relative abundance of *A. fuscus* was fairly consistent and represented about 10–23 percent of the sea snakes observed (see Table 1, Manning, 2014). A footnote in Smith (1926) also indicates that a sample of 27 dusky sea snakes (out of an ~100-specimen sea snake collection) had recently been collected for him at Ashmore Reef. The dusky sea snake, however, has not been recorded in a single survey conducted at Ashmore

Reef after 2005, despite considerable effort (Lukoschek *et al.*, 2013; Table 1, Manning, 2014). Based on reef area data reported in Skewes *et al.* (1999), Ashmore Reef represents about 40 percent of the dusky sea snake's historical reef habitat. Extirpation from this reef would represent a substantial change in the species' distribution and abundance.

A survey in 2005 at Hibernia Reef indicated a relatively low abundance of *A. fuscus*, and the most recent surveys, conducted in 2012 and 2013, have failed to detect any dusky sea snakes despite extensive survey effort (Guinea, 2005; Guinea, 2013). Dusky sea snakes were observed in surveys conducted at Scott Reef in 1972/73, 2006, 2012 and 2013; however, their relative abundance varies across the surveys, and no trends are detectable given the limited data (see Table 1, Manning, 2014). For example, Guinea (2012) visited Scott Reef in February, 2006, and reported that dusky sea snakes, as the third-most abundant species, made up 15 percent of the total sea snake sightings (Guinea, 2013). Portions of Scott Reef were surveyed again in 2012 and 2013, and dusky sea snakes made up only 3.2 percent and 7.4 percent of the total sightings respectively for each year (Guinea, 2013). At Seringapatam Reef and Cartier Island, *A. fuscus* is rare or potentially absent. Overall, while these limited abundance data are very difficult to interpret, they indicate that dusky sea snakes have not been present in high numbers in any recent reef surveys (Table 1, Manning, 2014).

The dusky sea snake has a restricted range, and structure and connectivity of populations is uncertain. Assuming that *A. fuscus* is extirpated from Ashmore Reef, Sanders *et al.* (2014) recently estimated that the dusky sea snake's range is now less than 262 sq km. Although structure and connectivity of reef populations of *A. fuscus* have not been studied directly, some information may be gleaned from several studies on the olive sea snake, *A. laevis*, a sympatric congener that is larger in size, more common, and more widely distributed than *A. fuscus*, but is very closely related to *A. fuscus* (Sanders *et al.*, 2013b). As mentioned above, a short-term (6–9 days) tracking study on *A. laevis* suggests that adults of this species have small home ranges (1,500–1,800 sq m) and undergo limited active dispersal (Burns and Heatwole, 1998). Results of that study are somewhat supported by analyses by Lukoschek *et al.* (2007b) of mitochondrial DNA (mtDNA) from 354 olive sea snakes collected across its range, including some samples from Hibernia, Scott, and

Ashmore reefs and Cartier Island. Based on their results, Lukoschek *et al.* (2007b) concluded that gene flow among the reefs of the Timor Sea is low, and that olive sea snakes at these reefs have been diverging for some time. A subsequent analysis of microsatellite DNA from the same specimens indicates that two of the most distant Timor reef populations of *A. laevis* are significantly diverged (Lukoschek *et al.*, 2008). However, the degrees of divergence of other reef populations were not statistically significant, and there was no clear isolation-by-distance relationship (Lukoschek *et al.*, 2008). Although not conclusive, the available information for the olive sea snake and the fact that dusky sea snakes also lack a dispersive larval phase, suggest connectivity of *A. fuscus* may be limited among some reefs within the region. Limited inter-population exchange would increase the extinction risk and reduce the recovery potential for local populations that have experienced severe declines or have been lost.

Summary of Factors Affecting the Dusky Sea Snake

Available information regarding current, historical, and potential threats to the dusky sea snake was thoroughly reviewed (Manning, 2014). Although causes for observed declines in dusky sea snake have not been conclusively determined, we found that the species is being threatened by hybridization. Other possible threats include vessels, pollution, climate change, and inadequate regulatory mechanisms. We summarize information regarding each of these threats below according to the factors specified in section 4(a)(1) of the ESA. Available information does not indicate that disease, predation, or overutilization (including bycatch) are operative threats on this species; therefore, we do not discuss those further here. See Manning (2014) for additional discussion of all ESA Section 4(a)(1) threat categories.

The Present or Threatened Destruction, Modification, or Curtailment of Its Habitat or Range

Aipysurus fuscus is dependent on coral reefs for prey and shelter, and loss of live coral is a possible mechanism contributing to the decline of *A. fuscus* at locations such as Ashmore Reef. Coral reefs of the Sahul Shelf experienced widespread bleaching in response to El Niño events in 1998 and 2003. Ashmore Reef experienced bleaching in 1998 and again, to an apparently greater extent, in 2003 (Lukoschek *et al.*, 2013). However, because there are no estimates of coral coverage prior to 1998, the extent of

coral loss following the 1998 event has not been quantified. Widespread mortality of corals was documented in response to the 2003 bleaching event, and average live coral coverage was reduced to 10 percent (Kospartov *et al.*, 2006; as cited in Lukoschek *et al.*, 2013). Surveys conducted in 2005 and 2009 indicated that recovery of corals at Ashmore Reef was rapid but delayed by about 7 years (Ceccarelli *et al.*, 2011). Overall, there has been an eight-fold increase in hard coral coverage from 1998 to 2009 (Hale and Butcher, 2013), with all of the recorded recovery occurring after 2005. Meanwhile, survey data suggest complete loss of dusky sea snakes at Ashmore Reef after 2005. Existing survey data also show sharp declines in total sea snake abundance and species diversity at Ashmore Reef following both the 1998 and 2003 bleaching events (Lukoschek *et al.*, 2013). These patterns are consistent with a hypothesis that loss of live corals affects reef-associated sea snakes.

The patterns at Ashmore Reef are contrasted, however, by those observed at Scott Reef. Following the 1998 bleaching event, a greater than 80 percent loss of hard and soft coral cover occurred, which translated into a reduction of live coral coverage to a total of roughly 10 percent (Smith *et al.*, 2008). The 1998 El Niño event represents the most extreme temperature anomaly recorded for Scott Reef, and involved a rapid rise in water temperatures that remained above normal for two months (NOAA, 2013). Almost 6 years after the bleaching event (in 2004), the hard corals had partially recovered to 40 percent of their pre-bleaching cover, the soft corals showed no sign of recovery, and community composition of corals remained significantly altered (Smith *et al.*, 2008). Within 12 years after the event (by 2010), coral cover, recruitment, community composition, and generic diversity were similar to pre-bleaching years (Gilmour *et al.*, 2010). Several lesser disturbances, including two cyclones and the 2003 El Niño event, occurred during this time period and may have slowed the rate of recovery to some extent (Gilmour *et al.*, 2013). Available sea snake survey data, spanning 1972–2013, with surveys in 1972–73, 2006, 2012, and 2013, do not appear to indicate a major decline in abundance of dusky sea snakes at Scott Reef, which were relatively common during the surveys conducted by Guinea (2012) in 2006. However, the temporal gaps in these survey data, especially from 1973 to 2006, could conceal shorter-term patterns.

A comprehensive understanding of the relationship between live coral cover and dusky sea snake abundance likely requires more detailed information regarding coral species composition, habitat complexity, and coral and prey fish resiliency relative to both the 1998 and 2003 bleaching events. Such an analysis might offer further insights into the differing response patterns at the two reefs, and an indication of whether sea snake abundance is driven by live coral coverage over timescales relevant to these disturbances. At this time, however, because a clear or consistent pattern does not emerge from the available data regarding dusky sea snake abundances at Ashmore and Scott reefs in relationship to these two bleaching events, we cannot conclude that loss of live coral is contributing to the decline of the dusky sea snake.

The reefs where dusky sea snakes are found lie more than several hundred kilometers offshore and thus enjoy a considerable degree of protection from human activities and land-based sources of pollution. Despite this remoteness, the reefs may experience some degradation as a result of vessel traffic. Anchor damage, pollution from contaminated bilge water, and marine debris are among the potential issues identified at Ashmore Reef, which experiences a relatively high level of traffic from Indonesian fishers, yachts, merchant ships, and illegal entry vessels (Whiting, 2000; Lukoschek *et al.*, 2013). The mechanisms for and extent to which these boat-based habitat threats are impacting dusky or any other sea snake species of the Timor Sea reefs are unknown.

The extensive oil and gas industry activity in this region may also be a possible source of disturbance affecting dusky sea snakes and their habitat. Exploration and extraction activities within the Ashmore Platform began in 1968 (Geoscience Australia, 2012) and are expected to continue for some time, given the significant resources within this region. Ashmore Reef and Cartier Islands lie about 50–80 km west of the main offshore wells in the Timor Sea, and the closest exploration wells are 36 km away (Russell *et al.*, 2004). However, Scott Reef lies directly above a significant portion of the Torosa Reservoir, where drilling for natural gas is expected to start by 2017. The development of the natural gas facility in this area will mean increased vessel traffic and potentially light, sound, and chemical pollution. The area is also expected to experience minor subsidence or compaction as the gas is removed (Woodside Energy LTD, 2013).

Whether, and the degree to which, any of these threats or a combination of these threats will impact dusky sea snakes is not yet known.

Unfortunately, extremely limited information also exists regarding the toxic effects of oil exposure on sea snakes. Oil spills, which occur more frequently as a result of vessel or pipeline incidents rather than exploration and drilling activities (www.amsa.gov.au), have also not occurred very often in this region. Some information is available from the August 2009 explosion of the West Atlas oil rig on the Montara Well, which leaked oil and gas uncontrollably into the Timor Sea for 74 days until the well was finally capped in November 2009. Considered one of the worst oil-related spills to have ever occurred in Australia, the Montara leak was analogous in nature to the Deepwater Horizon disaster of April 2010 in the Gulf of Mexico. In an effort to rapidly assess impacts to multiple taxa, Watson *et al.* (2009) conducted ship-based transect surveys in areas around the Atlas drilling platform in September 2009. They did not observe or identify any dusky sea snakes; however, they did observe “lethargic sea snakes lying in thick oil (*i.e.*, not moving much when approached, unable to dive)” and collected a dead horned sea snake (*Acalyptophis peronii*) from oil-affected waters for further analysis (Watson *et al.*, 2009). The necropsy report indicated that this snake was in good physical condition, with no visible external or internal pathologies, and no oil was detected in swab samples of the skin (Gagnon and Rawson, 2010). Chemical analysis of tissues clearly indicated that exposure to crude oil occurred through ingestion of prey and not through inhalation (Gagnon and Rawson, 2010). *Acalyptophis peronii* is considered more of a diet specialist than the dusky sea snake and primarily consumes burrowing gobies (McCosker, 1975; Voris and Voris, 1983). Because they saw no physical damage to the gut structure and no contamination of the tissues, Gagnon and Rawson (2010) concluded it was unlikely that oil ingestion was the primary cause of death. Tests for presence of chemical dispersants used during the spill-response were not conducted.

A necropsy was also performed on a dead sea snake landed by a commercial fisherman operating in the vicinity of the West Atlas spill on September 14, 2009 (Gagnon, 2009). This specimen was identified as *Hydrophis elegans*, which is a relatively widespread and abundant species that preys on eels and other fishes (McCosker, 1975; Voris and

Voris, 1983). The necropsy indicated that the snake had fed recently and that the stomach contents were contaminated with oil (Gagnon, 2009). Relatively high levels of polycyclic aromatic hydrocarbons were also detected in the lungs, trachea, and muscle tissue (Gagnon, 2009). Neither of two dispersant chemicals used to treat the spill were detected in lung samples (Gagnon, 2009). The necropsy report concluded that the likely cause of death for this specimen was exposure to petroleum hydrocarbons (Gagnon, 2009).

In 2012 and 2013, Guinea (2013) conducted surveys to evaluate the potential impacts of the Montara leak on species of marine reptiles. Potentially impacted areas surveyed included Ashmore Reef, Cartier Island, and Hibernia Reef; Scott and Seringapatam reefs were surveyed as control reefs (Guinea, 2013). Ashmore Reef and Cartier Island are 167 km west-north-west and 108 km west from the Montara well, respectively. Seringapatam and Scott reefs are several hundred km south-east of the Montara well and far from modeled oil trajectories (Guinea, 2013). The extensive survey efforts of Guinea (2013) did not indicate any impact of the hydrocarbon release on marine reptiles (sea turtles and sea snakes) of the potentially affected reefs. Of the reefs surveyed, Hibernia Reef and Cartier Island had the highest sea snake density; however, no sea snakes were observed at Ashmore Reef, where sea snake abundance and diversity had already declined to very low levels prior to the 2009 incident (Guinea, 2013). Overall, these data suggest that while there are likely to be acute impacts to sea snakes in response to major spills, it is unlikely that pollution stemming from oil and gas industry activities has contributed to the observed declines of the dusky sea snake.

Overall, based on the existing information, we conclude that there is a low likelihood that these habitat-related threats have contributed to the observed decline of the dusky sea snake. At this time, there is insufficient information to indicate whether and how the dusky sea snake will be affected by these habitat issues in the future. We do expect that each of the various habitat-related issues summarized above will continue well into the future, and some may worsen. Given that El Niño and its associated warming of equatorial Pacific Ocean waters is a natural and reoccurring climate phenomenon, coral bleaching in response to sufficiently strong El Niño events will continue. Furthermore, because climate warming as a consequence of carbon dioxide

emissions is expected to continue (IPCC, 2013), and elevated sea surface temperatures are expected to rise at an accelerated rate (Lough *et al.*, 2012), loss of corals through bleaching events is expected to increase. The expansion of Australia's oil and gas exploration and extraction in the Timor Sea may also result in an increased risk of oil spills and additional habitat threats for dusky sea snakes.

Inadequacy of Existing Regulatory Mechanisms

The dusky sea snake and its habitat receive a significant degree of regulatory protections. The largest potential gap in existing regulatory mechanism may be for threats related to climate change. Oil spills, while rare and unpredictable, and other oil and gas industry activities may also pose threats to the species as a consequence of inadequate management and regulation. We summarize the available information regarding related regulatory protections below; a more in-depth discussion is available in Manning (2014).

Along with all of Australia's other hydrophiine sea snakes, dusky sea snakes are listed under the Commonwealth Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act). The EPBC Act provides a legal framework to protect and manage Australia's nationally and internationally important flora, fauna, ecological communities, and heritage places that are of national environmental significance. Under the EPBC Act, no one may “kill, injure, take, trade, keep or move a member of a native species” within any reserve without a permit (Commonwealth of Australia, 2000). The EPBC Act requires that surveys be conducted for listed marine species. The EPBC Act also provides that the Australian Government Minister for the Environment may make or adopt a recovery plan for a listed species, to set out the research and management actions needed to stop the decline of the species and support its recovery. There are no recovery plans in place for any sea snake species, however (www.environment.gov.au/topics/biodiversity/threatened-species-ecological-communities/recovery-plans). Thus, while the dusky snake receives substantial protection under the EPBC Act, without a recovery plan, that protection may not be enough to help stabilize and recover the species.

Two of the five main reefs within the dusky sea snake's historical range, Ashmore Reef and Cartier Island, are protected reserves. Ashmore Reef National Nature Reserve was established

in 1983, under the National Parks and Wildlife Conservation Act 1975 (a predecessor to the EPBC Act), and later listed as a Ramsar Site in 2000, under the Ramsar Convention, which is an intergovernmental treaty on sustainable use of wetlands. In Australia, Ramsar Sites receive protection under the EPBC Act: Any action that will have or is likely to have a significant impact on a Ramsar Site requires an environmental assessment and approval. The EPBC Act also sets forth national standards for managing, planning, monitoring, involving the community in, and conducting environmental assessments of Ramsar Sites to insure consistent compliance with the Ramsar Convention. Cartier Island, a former British Air Force bombing range, was designated as a Marine Reserve in 2000. These two reserves cover a combined area of 750 km² and are both assigned to IUCN category Ia—strict nature reserve. IUCN category Ia areas are protected to preserve biodiversity and maintain the areas for the benefit of scientific research. Human access to such areas is tightly controlled and limited. A small section of Ashmore Reserve is managed as IUCN category II—national park. Such areas are managed to protect ecosystems and biodiversity, and while still restricted, human visitation is not as limited as for category Ia areas. No fishing or harvest of any biota is allowed within the reserves, with the limited exception of finfish fishing within the category II area of Ashmore Reef, and then only as long as the fish are used for relatively immediate consumption. Given the lack of clearly identified habitat-related or human-disturbance-related threats to the dusky sea snake, there is no indication that these reserves and area protections are inadequate such that they have contributed to the observed decline of the species.

According to the Australia Department of Sustainability, Environment, Water, Population, and Communities (DSEWPC) 2012 Report Card for marine reptiles listed under the EPBC Act, pollution from offshore oil rigs and operations is a potential concern for sea snakes (DSEWPC, 2012). This report also states that Australia has a strong system for regulating the oil and gas industry and that this system was strengthened further in the wake of the Montara oil spill. Details on how any particular processes or regulations were strengthened are not provided in this report and could not be found. Although oil spills pose a potential threat to the health and status of the dusky sea snake, oil spills are relatively

rare, and there is insufficient information to indicate that the existing regulatory mechanisms are inadequate or that they have contributed to the decline of this species.

Potential threats to dusky sea snakes stemming from anthropogenic climate change include elevated sea surface temperature, ocean acidification, and increased coral bleaching events (see below). Impacts of climate change on the marine environment are already being observed in Australia and elsewhere (Melillo *et al.*, 2014; Poloczanska *et al.*, 2012), and the most recent United Nations Intergovernmental Panel on Climate Change (IPCC) assessment provides a high degree of certainty that human sources of greenhouse gases are contributing to global climate change (IPCC, 2013). Ocean temperatures around Australia have increased by 0.68 °C since 1910–1929 (Poloczanska *et al.*, 2012), and carbon dioxide inputs have lowered ocean pH by 0.1 units since 1750 (Howard *et al.*, 2009). Australia and other countries have responded to climate change through various international and national mechanisms. Australia signed on to the Kyoto Protocol in 2007 and has active domestic and international programs to lower greenhouse gas emissions (www.climatechange.gov.au/). However, in Australia, there appear to be no specific actions to address potential climate change effects on marine reptiles beyond monitoring (Fuentes *et al.*, 2012). Because climate change related threats have not been clearly or mechanistically linked to decline of dusky sea snakes, the adequacy of existing or developing measures to control climate change threats is not possible to fully assess, nor are sufficient data available to determine what regulatory measures would be needed to adequately protect this species from climate change. While it is not possible to conclude that the current efforts have been inadequate, such that they have contributed to the decline of this species, we consider it likely that dusky sea snakes will be negatively impacted by climate change, given the predictions of widespread and potentially permanent damage to coral reefs in Australia (IPCC, 2013).

Overall, we do not find there is substantial evidence indicating that *A. fuscus* is currently threatened by the lack of adequate regulatory mechanisms. Beyond the direct protection the species receives through its listing under the EPBC Act, the dusky sea snake receives additional direct and indirect protection within the Ashmore Reef and Cartier Island Marine Reserves. Given the

predictions of worsening damage to coral reefs in Australia in response to climate change (IPCC, 2013), the largest potential future gap in the existing regulatory mechanisms appears to be related to climate change.

Other Natural or Manmade Factors Affecting Their Continued Existence

Elevated sea surface temperature as a consequence of climate change has been proposed as a possible threat to sea snakes, and we have addressed habitat-related effects above. The IUCN Red List assessment for *A. fuscus*, suggests that climate-induced increases in water temperature may actually exceed the upper lethal limit for *A. fuscus*, and thereby pose a threat to the species (Lukoschek *et al.*, 2010). These authors assumed an upper lethal limit of 36 °C, based on data for the pelagic sea snake, *Pelamis platurus*. Experiments to measure the thermal tolerances of *A. fuscus* have not been conducted.

Sea snakes, like all reptiles, are ectotherms, and thus to a great extent are physiologically affected by temperature. On a large geographic scale, the distribution of sea snakes is considered to be dictated by ocean temperatures: Sea snakes generally do not occur in waters below about 18 °C (Davenport, 2011). Most sea snakes can tolerate temperatures up to a mean of about 39–40 °C, but tolerances may vary with the size of the snake and the rate of temperature change (Heatwole *et al.*, 2012). Also, although sea snakes are able to dive to avoid extreme temperatures of surface waters, they have limited capacity to acclimate and cannot thermoregulate (Heatwole *et al.*, 2012).

Sea surface temperatures vary seasonally within the Timor Sea. The highest recorded oceanic water temperature in the Ashmore region is 31 °C, and the highest recorded lagoon water temperature is 35.4 °C (Commonwealth of Australia, 2002). These temperatures are below the assumed upper lethal temperature limit for dusky sea snakes; but Australia's average ocean temperatures have increased by over half a degree since 1910–1929, and the rate of warming has accelerated since the mid-20th century (Poloczanska *et al.*, 2012). Given the thermal tolerances of other sea snakes and the ocean temperatures currently experienced by *A. fuscus* at present, it is very unlikely that elevated ocean temperature has been a source of mortality. However, it is plausible that a continuation of the observed rate of ocean warming would, in the distant future, result in negative physiological consequences for *A. fuscus*.

Hybridization and introgression have recently been identified by Sanders *et al.* (2014) as a threat to the continued existence of *A. fuscus*. Hybridization, or the production of viable offspring through the crossing of genetically distinct taxa or groups, occurs in the wild for about 10 percent of animal species (Mallet, 2005). Hybridization can lead to introgression, or the integration of foreign genetic material into a genome. The conservation concern in this particular case is that reproductive barriers between the olive sea snake, *A. laevis*, and the dusky sea snake, *A. fuscus*, appear to be breaking down, potentially allowing *A. fuscus* to undergo reverse speciation.

The dusky sea snake co-occurs with the closely-related olive sea snake throughout its range, and the two species are thought to have shared a common ancestor approximately 500,000 years ago (Sanders *et al.*, 2013b). The olive sea snake is a relatively abundant and much more widely distributed species compared to the dusky sea snake. Although similar in appearance, the two species can be distinguished based on body scale rows, body size, and color pattern. Sanders *et al.* (2014) analyzed 11 microsatellite markers for *A. fuscus* and *A. laevis* across four reefs (Ashmore, Hibernia, Scott, and Seringapatam) to assess inter-specific gene flow and introgression. Results of their genetic analyses indicate significant and asymmetric gene flow, with higher rates of introgression from *A. laevis* into the smaller *A. fuscus* population (Sanders *et al.*, 2014). A high frequency of hybrids was also found at each of the four reefs included in the study area. Forty-three percent of the snakes sampled (n=7) at Ashmore, 55 percent of the snakes sampled (n= 42) at Scott Reef, and 42 percent of the snakes sampled (n=12) at Seringapatam Reef were identified as hybrids (Sanders *et al.*, 2014). At Hibernia Reef, 95 percent of the snakes sampled (n=19) were hybrids (Sanders *et al.*, 2014). Phenotypically, the majority of hybrids resembled the olive sea snake (Sanders *et al.*, 2014). Whether the observed hybridization is a purely natural process or has human causes is not yet known. Regardless, the high rates of hybridization of *A. fuscus* with another species across its range may lead to the eventual disappearance of this taxonomic species and is a threat to its survival.

Extinction Risk

Although accurate and precise data for many demographic characteristics of dusky sea snakes are lacking, the best available data provide multiple lines of

evidence indicating that this species currently faces a high risk of extinction. The probable extirpation of the dusky sea snake from Ashmore Reef, which constitutes about 40 percent of the historical reef habitat, represents a contraction of an already limited range for this species. Loss of dusky sea snakes from Ashmore Reef and low relative abundances at all other reefs, coupled with high rates of hybridization throughout the range and a presumed low rate of dispersal, suggest that the species is declining and unlikely to recover without intervention. The interaction of the threats of low and declining abundance, limited dispersal, and high rates of hybridization all suggest a high risk of extinction in the near term.

Protective Efforts

As mentioned previously, all of Australia's hydrophiine sea snakes are listed and protected under the EPBC Act, making it illegal to kill, injure, take, trade, or move dusky sea snakes in Commonwealth waters without a permit (DSEWPC, 2012a). The EPBC Act also requires that surveys be conducted for listed marine species.

Sea snakes are also identified as a "conservation value" in Australia's North-west Marine Bioregional Plan (DSEWPC, 2012b). Marine bioregional plans are meant to improve the way decisions are made under the EPBC Act, particularly with respect to balancing protection of marine biodiversity with the sustainable use of natural resources. The North-west Plan identifies activities that may affect sea snakes and thus require prior approval. National heritage places are also listed and protected under the EPBC Act. Ashmore, Scott, and Seringapatam reefs are all listed on Australia's Commonwealth Heritage List, and under the EPBC Act, approval must be obtained before any action takes place that could have a significant impact on the national heritage values of these areas.

Also mentioned previously were the various habitat protections currently in place that directly and indirectly protect the coral reefs within the dusky sea snake's range. For example, the Ashmore Commonwealth Marine Reserve, which includes 583 km² of sandy islands, coral reefs, and surrounding waters up to 50 m deep (Commonwealth of Australia, 2002), is almost completely closed to the general public. Permits may be issued to authorize visits for tourism or recreation. There are 1–2 visits per year by commercial tourism vessels to view wildlife, and about 15–20 recreational yachts that visit each year (Hale and

Butcher, 2013). Indonesians have fished this site for centuries and subsistence fishing is allowed in only the IUCN category II portion of the reserve (Hale and Butcher, 2013). No commercial fishing is allowed in any part of the Reserve. The relatively pristine state of the site makes it attractive for the long-term monitoring and other scientific projects that are conducted there (Hale and Butcher, 2013). Starting in the late 1980's, Environment Australia (EA) contracted a private vessel and crew to undertake on-site management at the Reserve; however, as of 2000, Australian Customs Service took over this responsibility (Whiting, 2000). Enforcement of protections at the Reserve depends largely on the presence of Customs officials, which is not quite continuous (Lukoschek *et al.*, 2013; Whiting, 2000).

The Cartier Island Commonwealth Marine Reserve, designated in 2000 under the EPBC Act, is completely closed to the public. No commercial or recreational fishing is allowed. General access and several specific activities, such as scientific research, photography and tourism, may be allowed with prior approval from the Director of National Parks issued under the EPBC Act (see <http://www.environment.gov.au/topics/marine/marine-reserves/north-west/cartier-activities>).

Since the early 18th century, Indonesian fishers have visited and fished reefs within the Timor Sea, mainly in search of trepang, trochus, turtle, shark fin, and reef fishes (Commonwealth of Australia, 2002). In 1974, a Memorandum of Understanding (MOU) was established between Australia and Indonesia that set out arrangements by which traditional fishers may access resources in Australia's territorial sea. Because of its shape, the area covered by this MOU is often referred to as the MOU Box. The MOU Box, which covers an area of about 50,000 km², includes the five main reefs where the dusky sea snake occurs (Skewes *et al.*, 1999). The marine resources within this area are managed by the Australian Government, and traditional fishing by Indonesian fishers is allowed. However, as discussed above, certain restrictions apply within the Marine Reserves. Traditional Indonesian fishers may access parts of the Ashmore Reserve for shelter and freshwater and to visit grave sites, but, as mentioned previously, fishing is prohibited in both the Cartier Island and Ashmore Marine Reserves, with the limited exception for fishing for immediate consumption within the category II area of the Ashmore Reserve. There is no evidence that sea snakes

have been targeted by Indonesian fishers (Hale and Butcher, 2011; Lukoschek *et al.*, 2013).

Because sea snakes are listed under the EPBC Act, all Australian fisheries are required to demonstrate that direct and indirect interactions with sea snakes are sustainable (Zhou *et al.*, 2012). Commercial trawls take over a dozen species of sea snakes (Heatwole 1997; Wassenberg *et al.*, 2001; Zhou *et al.*, 2012), and in the absence of bycatch reduction devices (BRDs), an estimated 48.5 percent of all incidentally captured sea snakes will die (Wassenberg *et al.*, 2001). BRDs are required in the prawn trawl fishery to minimize bycatch mortality and help conserve protected species. The only trawl fishery that operates within the range of the dusky sea snake is the North West Slope Trawl Fishery (NWSTF). The Australian Fisheries Management Authority (AFMA) reports that the NWSTF, which targets three scampi species (lobsters), is a low effort fishery with a very low level of bycatch and no documented interactions with threatened, endangered, or protected species (AFMA, 2012). The NWSTF is also a deep-water fishery, and thus unlikely to encounter the reef-associated dusky sea snake (Fry *et al.*, 2001; Lukoschek *et al.*, 2007a; Lukoschek *et al.*, 2013). As discussed here and in further detail in the status review report (Manning, 2014), there is no indication that direct harvest or incidental capture poses a threat to the dusky sea snake.

Sea snake products have been traded internationally since the 1930s (Marsh *et al.*, 1994), but no sea snake species is currently listed under the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). Australia's Wildlife Protection Act 1982 restricts the export of sea snake products out of Australia (Marsh *et al.*, 1994). There are no data to suggest that the dusky sea snake is threatened by past, present, or future trade.

Despite their apparent substantiveness, these existing and ongoing conservation efforts seem unlikely to prevent further decline of the dusky sea snake, because they have failed to prevent the decline of the species to date. For example, decades of protections at Ashmore Reef, while maintaining this as a relatively pristine reef (Hale and Butcher, 2013), have not prevented the severe decline and likely extirpation of dusky sea snakes there. Furthermore, the threat posed by hybridization is beyond the scope of existing protections. We are thus not able to conclude that the existing protective efforts alter the extinction risk for the dusky sea snake. We are not

aware of any additional, planned or not-yet-implemented conservation measures that would protect this species; thus, we did not conduct an analysis under the PECE. We seek additional information on other conservation efforts in our public comment process (see below).

Proposed Determination

Based on our consideration of the best available data, as summarized here and in Manning (2014), and protective efforts being made to protect the species, we conclude that the dusky sea snake, *A. fuscus*, is currently at high risk of extinction throughout its range. We therefore propose to list it as endangered under the ESA.

Banggai Cardinalfish

The following section describes our analysis of the status of the Banggai cardinalfish, *Pterapogon kauderni*. More details can be found in Conant (2014).

Species Description

The Banggai cardinalfish is a species within the family Apogonidae and genus *Pterapogon*. It was discovered in 1920 by Walter Kaudern and described by Koumans (1933). The genus *Pterapogon* contains one other species, *P. mirifica*, from northwestern Australia (Allen and Donaldson, 2007).

The Banggai cardinalfish is a relatively small marine fish. Adults generally do not exceed 55 to 57 mm standard length (Vagelli, 2011). The species is distinguished from all other apogonids by its tasseled first dorsal fin, elongated anal and second dorsal fin rays, and deeply forked caudal fin (Allen, 2000). It is brilliantly colored, with contrasting black and light bars with whitish spots over a silvery body.

The Banggai cardinalfish has an exceptionally restricted natural range (approximately 5,500 km²) within the Banggai Archipelago, Indonesia. Populations have been introduced in areas of Indonesia outside of the Banggai Archipelago, including Luwuk Harbor (Bernardi and Vagelli, 2004), Palu Bay (Moore and Ndobe, 2007), Lembah Strait (Erdmann and Vagelli, 2001), Tumbak (Ndobe and Moore, 2005), Kendari Bay (Moore *et al.*, 2011), and north Bali (Lilley, 2008). These introductions are a result of discards from the ornamental live reef aquarium trade and introductions by dive-resort operators to support the tourist industry (Vagelli, 2011). The introduced populations are an artifact of the commercial ornamental live reef trade and are not part of any conservation program to benefit the native populations. Because we interpret the ESA as conserving species and the

ecosystems upon which these species depend, we consider the natural range to be biologically and ecologically important to the species' viability to persist in the face of threats. Distances between non-introduced populations range from less than 1 km (Vagelli, 2011) up to 153 km (Vagelli *et al.*, 2009). Distribution of populations is discontinuous, with deep water, strong currents, or coast exposed to severe weather serving as effective ecological barriers to migration (Bernardi and Vagelli, 2004; Ndobe *et al.*, 2012; Ndobe and Moore, 2013). The Banggai cardinalfish exhibits the highest known degree of genetic structure of any marine fish (Bernardi and Vagelli, 2004; Hoffman *et al.*, 2005; Vagelli *et al.*, 2009). Populations occurring on the same reef, separated by only a few kilometers, are genetically isolated from one another (Bernardi and Vagelli, 2004; Hoffman *et al.*, 2005; Vagelli *et al.*, 2009).

The Banggai cardinalfish is generally found in calm waters of sheltered bays or on the leeward side of islands (Allen and Donaldson, 2007). It inhabits a variety of shallow (from about 0.5 to 6 m) habitats including coral reefs, seagrass beds, and less commonly, open areas of low branching coral and rubble. To avoid predators, it associates with microhabitats such as sea urchins and anemones (Vagelli, 2011). Banggai cardinalfish are found in waters ranging from 26–31 °C, but averaging 28 °C (Ndobe *et al.*, 2013).

The Banggai cardinalfish, like many apogonids, exhibits reversed sex roles, where males provide parental care and brood eggs in their mouths. It lacks a planktonic larval stage and extends the brooding of larvae for about 7 days after hatching, which results in the release of fully formed juveniles. Spawning occurs year round but peaks around September through October, which is a period of fewer storms in the region (Ndobe *et al.*, 2013). The Banggai cardinalfish has the lowest fecundity reported for any apogonid (Vagelli, 2011). Generation length (the age at which half of total reproductive output is achieved by an individual) is estimated to be 1.5 years (Vagelli, New Jersey Academy for Aquatic Sciences (NJAAS), personal communication cited in Allen and Donaldson (2007)) to 2 years (Ndobe *et al.*, 2013). Its lifespan in the wild has been estimated at approximately 2.5–3 years (Vagelli, 2011), with a maximum lifespan up to 3–5 years (Ndobe *et al.*, 2013). Based on a conservative estimate, a male could incubate/brood approximately 400 to 640 offspring over his lifespan (Vagelli, personal communication, 2014), of which less

than 5 percent may survive to adulthood (Vagelli 2007 as cited in CITES (2007)). High mortality occurs during the first days after release from the brood pouch due to predation, including parental and non-parental cannibalism (Vagelli, 1999).

Banggai cardinalfish form stable groups. Natural group size is difficult to know because group size decreases with fishing pressure, and most populations are not pristine. However, one bay (oyster pearl farm) in private ownership in the Banggai Islands had, until 2006, never been fished, and group size averaged about 13 fish, but varied from 2–33 fish per group (Lunn and Moreau, 2002). At the same site in 2004, group size varied from 1 to over 200 fish per group (Moore, unpublished data, 2014). Group size is typically less than 25 individuals, although smaller groups are common and vary by age class and habitat type (Vagelli, 2011).

The first scientific surveys of Banggai cardinalfish estimated population abundance and density between 1.7 million, with a mean density of 0.03 fishes per m², based on a census at three sites in 2001 (Vagelli, 2002; Vagelli and Erdmann, 2002), and 2.4 million, with a mean density of 0.07 fishes per m², based on an expanded census of 34 sites conducted in 2004 (CITES, 2007). In 2007, population the density estimate of the expanded survey sites indicated a mean density of 0.08 fishes per m² (Vagelli, 2008); however, overall population abundance was not reported for the 2007 survey. By 2011–2012, Ndobe *et al.* (in press) estimated the population abundance at 1.5–1.7 million, with a mean observed density of 0.05 fishes per m², reportedly for the 24 of the 34 sites that were surveyed in 2004 and 2007. The 2011–2012 estimates does not include locations in Toado where the habitat was limited and density was very high (Ndobe *et al.*, in press); thus, the population abundance estimate likely is biased low. However, 7 of the major sites first surveyed in 2004 have declined in abundance and mean density (Ndobe *et al.*, in press), indicating the population has likely decreased from the 2.4 million estimated in 2004. Although the mean observed density estimate of 0.03 fishes per m² found in the 2001 survey (Vagelli, 2002; Vagelli and Erdmann, 2002) is less than the 2011–2012 survey, the 2001 survey was based on only three sites, while the 2011–2012 survey was based on 24 sites of the 34 sites. Ndobe (*et al.*, in press) selected the expanded survey sites from 2004 and 2007 for their 2011–2012 survey based on the author's previous work on habitat conditions and to better compare trends,

over time, in density and abundance. Ndobe (*et al.*, in press) stated that their 2011–2012 estimate of 1.5–1.7 million represented 62–71 percent of the abundance estimate of 2.4 million from the 2004 survey. A total abundance estimate was not provided for the 2007 survey, however mean observed density decreased approximately 38 percent between 2007 (0.08 fishes per m²) and 2011–2012 (0.05 fishes per m²).

Historical data on abundance are lacking, as surveys were done after harvest began in the early to mid-1990s. The private oyster pearl farm mentioned above is thought to represent a proxy for historical abundance by several researchers, though others disagree that the site is representative of historical abundance. The private oyster farm exists within a privately owned bay in Banggai Island, and fishing has been prohibited there since trade began, although illegal poaching in the bay was reported in 2006 (Talbot *et al.*, 2013). The habitat in the bay may be similar to other sites that support the Banggai cardinalfish; thus, several researchers claim this population can be used as a proxy for a baseline of population abundance (Allen and Donaldson, 2007; Vagelli, 2008). In 2001, densities of fish in the private oyster pearl farm averaged 0.63 ± 0.39 fishes per m² (1 standard deviation, SD) (range: 0.28 to 1.22 fishes per m²) (Lunn and Moreau 2002) and 0.58 fishes per m² in 2004 (Vagelli 2005). When these densities are compared to the densities found in the 2001 and 2004 survey data discussed above, they indicate that the Banggai cardinalfish abundance has declined up to 90% from historical levels (Allen and Donaldson, 2007; Vagelli, 2008). However, several researchers (Moore, Sekolah Tinggi Perikanan dan Kelautan (STPL), personal communication 2014; Ndobe, Tadulako University, personal communication 2014) caution against the use of this bay as a baseline for population trends. Banggai cardinalfish population distribution is inherently patchy, and density is highly variable between and within sites of the Banggai Archipelago, including this bay (Moore, unpublished data, 2004). The researchers also question whether the habitat in the bay is comparable to other sites. The bay has been protected from degradation because it is privately owned and contains significant amounts of sheltered habitat and good quality microhabitat/habitat, with limited suitable habitat for predators of the cardinalfish, such as groupers and other larger reef fish. We acknowledge the debate regarding the use of the data from the private oyster farm as a

baseline for historical abundance. However, even without that data, it is clear that population abundance estimates at sites throughout the Banggai Archipelago declined significantly between 2004 and 2011–2012.

Declines and extirpations of local populations have been observed across years, likely due to directed harvest and, more recently, habitat destruction. In the 2001 survey, Bakakan Island had about 6,000 fish, but by the 2004 census, only 17 fish remained (Vagelli, 2008). In the 2007 survey, 350 individuals were found at Bakakan Island, but this was still well below the 6,000 fish found in the 2001 survey (Vagelli, 2008). In 2014, Moore (personal communication) reported that local fishers characterize the cardinalfish population on Bakakan Island as small and declining. Between the 2001 and 2004 surveys, the population density at Masoni Island doubled from 0.03 to 0.06 fish per m² (an increase of approximately 150 fish in 3 years) (Vagelli, 2005). This increase is thought to have occurred in response to a collecting ban that the local people imposed in early 2003. However, in the 2007 survey, the population was found to have declined to 0.008 fish per m², with 38 fish recorded over the entire census site (the largest group consisted of 2 individuals). An extensive search around the entire island identified only 150 fish (Vagelli, 2008). A population in southeast Peleng Island had 159 and 207 fish in 2002 and 2004, respectively (Vagelli, 2005). However, by 2007, it had been practically extirpated, with only 27 fish found (Vagelli, 2008). Overharvest of microhabitat, such as *Diadema* sea urchins and sea anemones, and coral mining have resulted in local population depletions on an island off Liang, which was surveyed in 2004, and was extirpated by 2012 (Ndobe *et al.*, 2013). Extirpation of local populations has been documented in areas with increased harvest of microhabitat, combined with fishing pressure on Banggai cardinalfish. Interviews with locals and visits to several sites in 2011 and 2012 indicate populations are declining in the Banggai Archipelago (Ndobe *et al.*, 2013).

Summary of Factors Affecting the Banggai Cardinalfish

Next we consider whether any one or a combination of the five threat factors specified in section 4(a)(1) of the ESA are contributing to the extinction risk of the Banggai cardinalfish. We discuss each of the five factors below, as all factors pose some degree of extinction risk. More details are available in Conant (2014).

Present or Threatened Destruction, Modification, or Curtailment of Habitat or Range

The illegal use of fish bombs (typically made with fertilizer and phosphorus) and cyanide to catch fish has resulted in significant loss of coral reef habitat within the Banggai cardinalfish range (Allen and Werner, 2002). Damage to coral reefs due to fish bombs is prevalent, even in protected areas (Talbot *et al.*, 2013). Cyanide is used to catch fish for the live reef fish trade, and the practice kills corals (*e.g.*, see Jones and Steven, 1997; Mous *et al.*, 2000). Boats have degraded the coral reefs in the area, and clear-cutting of wooded slopes and mangroves has occurred, increasing sedimentation, which degrades coral reef habitat (Lilley, 2008). Other upland activities, such as agriculture and human population growth, have increased the amount of waste and nitrates in the marine environment, promoting algal blooms (Lilley, 2008), which may destroy coral reefs by outcompeting them for vital resources such as light and oxygen (reviewed by Fabricius, 2005). Significant plastic, styrofoam, and other human-made debris occurs in the area (Lilley, 2008). This information indicates destruction of habitat is occurring within the Banggai cardinalfish's range. Although quantitative data on impacts to cardinalfish populations are lacking, considerable qualitative information exists indicating that where habitat has been degraded (*e.g.*, Tanjung Nggasuang and Toropot surveyed in 2004 and 2012, and Mbuang-Mbuang, on Boka Island, surveyed in 2012), large and thriving Banggai cardinalfish populations spread over large areas can be reduced to isolated remnants crowded into small remaining patches of habitat with some protective microhabitat (Ndobe, personal communication, 2014).

Coral reef conditions in the Central Sulawesi Province, including the Banggai Archipelago, were examined from 2001 through 2007 in seven Districts in the region (Moore and Ndobe, 2008). Average condition of the reefs was poor, and major impacts included coral mining, sedimentation, fishing, and predation (Moore and Ndobe, 2008). Population explosions of the crown-of-thorns starfish (*Acanthaster planci*), a coral predator, have been observed in the area, indicating an ecological imbalance, likely due to overharvest of natural predators and changes in hydrology and water quality (Moore *et al.*, 2012). Surveys conducted at five sites around Banggai Island from 2004 through 2011

showed coral reef cover declined by more than half, from 25 percent to 11 percent (Moore *et al.*, 2011; 2012). Major causes of the coral reef decline around Banggai Island were attributed to destructive fishing methods and general fishing pressure, coastal development, and the replacement of traditional homes with concrete and breeze-block dwellings, which increases the demand for mined coral and sand. Loss of coral reef cover may increase mortality of Banggai cardinalfish recruits due to cannibalism (Moore, personal communication, 2014; Ndobe *et al.*, in press).

Climate change may also impact Banggai cardinalfish habitat as a result of coral bleaching. Coral bleaching events due to warming temperatures are anticipated to increase by 2040 in areas of the Indian Ocean, including waters of Indonesia (van Hooidonk *et al.*, 2013). Coral bleaching due to elevated water temperatures has not been observed around Banggai Island up through December 2011; however, extensive bleaching was observed in nearby Tomini Bay in 2010 (Moore *et al.*, 2011; 2012). The Banggai cardinalfish is restricted to shallow waters with ambient temperatures ranging from 28 to 31 °C. Thus, warming temperatures may render habitat unsuitable, but specific data on impacts to the Banggai cardinalfish are lacking.

Sea urchins and anemones are experiencing intensive and increasing harvest pressure, which negatively impacts the Banggai cardinalfish (Moore *et al.*, 2012; Ndobe *et al.*, 2012). Sea anemones were once abundant but were drastically reduced from Tinakin Laut, Banggai Island, which resulted in a collapse of the Banggai cardinalfish population in the area (Moore *et al.*, 2012). Heavy harvest of sea anemones at Mamboro, Palu Bay, resulted in a drastic reduction of new recruits and juvenile Banggai cardinalfish (observed since 2006) in 2008 (Moore *et al.*, 2011). Moore *et al.* (2011; 2012) report that intensive harvesting of shallow water invertebrates, including sea anemones and sea urchins, is increasing and is linked to socio-economic trends associated with consumption by local seaweed farmers and use as feed for carnivorous fish destined for the ornamental live reef trade.

In addition, a disease of unknown origin may be damaging hard corals in habitat occupied by the Banggai cardinalfish. The disease affects the top sections of long-branched *Acropora* species as well as species of *Porites*, both of which are important microhabitat for the Banggai cardinalfish (Vagelli, 2011). Data are

lacking on the extent of impact the disease poses to Banggai cardinalfish habitat.

Overutilization for Commercial, Recreational, Scientific, or Educational Purposes

The Banggai cardinalfish is traded internationally as a live marine ornamental reef fish. It has been collected in the Banggai Islands, Indonesia, since 1995 (Marini and Vagelli, 2007). The United States, Europe, and Asia are the major importers of the Banggai cardinalfish for the aquarium trade (CITES, 2007). The Banggai cardinalfish is the tenth most common ornamental fish imported into the United States (Rhyne *et al.*, 2012). Banggai cardinalfish exports for the ornamental live reef fish trade may be decreasing, although systematic data are lacking. In 2001, up to 118,000 Banggai cardinalfish were sold to trade centers each month, with a total estimate of 700,000–1.4 million fish traded (Lunn and Moreau, 2002, 2004). From 2004 through 2006, around 600,000–700,000 fish were traded yearly (Moore *et al.*, 2011). In 2008 and 2009, 236,373 and 330,416 fish, respectively, were traded at Bone Bone, Toropot, and Bone Baru trade centers (Moore *et al.*, 2011, 2012). However, these numbers do not include trading data from Bone Bone in 2008 and other active centers (*e.g.*, Panapat for 2008 and 2009). These collections centers each reported about 15,000 fish per month in 2007 (Vagelli, 2008; 2011). Vagelli (personal communication, 2014) estimates that 1,000,000 Banggai cardinalfish are currently captured each year for the ornamental live reef trade.

The ornamental live reef fish trade has resulted in decreases in cardinalfish population density and extirpation of local populations. By 2000 (after less than a decade of trade), negative impacts on the Banggai cardinalfish from the trade were observed. The trade results in high mortality of cardinalfish collected. Based on interviews with collectors, Lilley (2008) estimated that only one out of every four to five fish collected makes it to the buyer for export due to high mortality and discard practices. Density and group size of cardinalfish and sea urchins are negatively impacted by the trade (Kolm and Berglund, 2003). Ndobe and Moore (2009) also found that populations were exploited, but observed high population density in areas where collection had been ongoing for some years with rotation between sites, indicating some harvest sustainability. Unfortunately, habitat destruction and collection and destruction of microhabitat (unrelated to the Banggai cardinalfish fishery) have

now greatly reduced cardinalfish populations at sites which had previously sustained periodic collection for more than a decade (Moore, personal communication, 2014). Decreases in population density are also evidenced by significant declines in the catch per unit effort (Vagelli, personal communication, 2014). In Bone Baru, from 1993–2000, fishers were catching an average of 1,000–10,000 fish per day, but by 2003 they only averaged 100–1,000 per day, with most catching between 200–300 fish (EC-Prep Project, 2005). Prior to 2003, collectors from Bone Baru typically required one day to capture approximately 2,000 specimens. In 2007, they reported requiring one week to capture the same number (Vagelli, 2011). Vagelli (2011) reports similar declines for Banggai Island, where between 2000 and 2004, the reported mean catch declined from about 1,000 fish/hour to 25–330 fish/hour.

Information suggests the number of active participants in the trade may have dropped. In 2001, there were 12 villages that collected the Banggai cardinalfish, but only 3 were active in 2011 (Moore *et al.*, 2011, 2012), and at least 5 villages were active in 2014 (Moore, personal communication, 2014). Reported as number of collectors, the data indicate a decline in participation as well, from about 130 in 2001 (Lunn and Moreau, 2004) to about 80 in 2007 (Vagelli, 2011) and 2012 (Vagelli, personal communication, 2014).

In 2012, a large-scale aquaculture facility based in Thailand began to breed Banggai cardinalfish in captivity for export, which may alleviate some of the pressure to collect fish from wild populations (Talbot *et al.*, 2013; Rhyne, Roger Williams University, unpublished data 2014). In 2013, approximately 120,000 Banggai cardinalfish were imported into the United States from the Thailand facility. The volume represents a significant portion of overall United States imports of the cardinalfish and may even exceed the number of wild fish currently imported (Rhyne, unpublished data, 2014). Efforts to captive-breed the species in the United States are also ongoing, which may alleviate dependence on wild-caught cardinalfish. In the United States, the Florida Department of Agriculture and Consumer Services has certified eight aquaculture facilities that are beginning to culture and market farm-raised Banggai cardinalfish (Knickerbocker, Florida Department of Agriculture and Consumer Services, personal communication 2014). In-situ breeding by the fishing communities in the endemic area may also alleviate

pressure on the natural population, but the concept requires further research before it can be implemented at a local community level (Ndobe, personal communication, 2014).

Disease or Predation

Predation and cannibalism are high among new recruits (Moore *et al.*, 2012). However, specific data are lacking on whether predation pressure is increasing or impacting the Banggai cardinalfish population growth beyond natural levels.

A virus known as the Banggai cardinalfish iridovirus (genus *Megalocytivirus*) is linked to high mortality of wild-caught fish imported for the ornamental live reef fish trade (Vagelli, 2008; Weber *et al.*, 2009). The virus causes necrosis of spleen and renal tissue, which appears as darkened tissue. Other symptoms are lethargy and lack of appetite. Surveys of wild populations have not reported symptoms of the disease. Necropsies of over 100 fish collected in the wild and at holding facilities showed no indication of the virus (Talbot *et al.*, 2013). Thus, the virus is likely transmitted from other specimens at containment centers, or is carried by the Banggai cardinalfish and is only expressed as a result of stress incurred during the long transport process (Weber *et al.*, 2009; Talbot *et al.*, 2013) and may not be a concern for wild fish.

Inadequacy of Existing Regulatory Mechanisms

Current Indonesian legislation requires that all trade in Banggai cardinalfish go through quarantine procedures before crossing internal administrative borders or prior to export (Moore *et al.*, 2011). Compliance historically has been low, but is improving (Moore, personal communication, 2014; Moore *et al.*, 2011). However, reported collection through the Fish Quarantine Data system, which records fish that go through quarantine procedures, was well below the total reported collection from Bone Baru, Toropot, and Bone Bone for 2008 and 2009. Bone Baru, Toropot, and Bone Bone reported collection of 236,373 fish in 2008 and 330,416 fish in 2009. Whereas in 2008 and 2009, the Fish Quarantine Data reported collection of 83,200 and 215,950 fish, respectively (Moore *et al.*, 2011). Enforcement of the Fish Quarantine procedures is weak, and illegal, unregulated, and unreported capture and trade are still a major problem, especially in remote areas (Ndobe, personal communication, 2014).

Legislation is needed to establish fishing quotas and size limits; however, no legally binding regulations have been passed or implemented (Moore *et al.*, 2011). Indonesia prohibits the use of chemicals or explosives to catch fish (Fisheries Law No. 31/2004, Article 8(1)). However, the practice continues (Vagelli, 2011), and damage to coral reefs due to fish bombs is prevalent, even in protected areas (Talbot *et al.*, 2013).

In 2011, Indonesia had proposed to list the Banggai cardinalfish for restricted protected status under domestic law. But the proposal stalled when the Indonesian Institute for Science argued that the introduced populations meant the species was no longer endemic, and thus did not meet the criteria for protected status (Moore, personal communication, 2014; Ndobe, personal communication, 2014). In 2007, the Banggai cardinalfish was proposed for listing under CITES Appendix II. However, the proposal failed. The species is listed in Annex D of the European Wildlife Trade Regulations, which only requires monitoring of European Union import levels through import notifications.

Based on the weaknesses discussed above, regulatory mechanisms on the commercial harvest industry do not appear adequate to ensure the population will be sustainable.

Other Natural or Manmade Factors Affecting Continued Existence

Global averaged combined land and ocean surface temperatures show a warming of 0.85 °C over the period 1880 to 2012 (IPCC, 2013). As discussed earlier (see Present or Threatened Destruction, Modification, or Curtailment of Habitat or Range), warming temperatures may destroy or modify habitat, but data are lacking on specific direct impacts to the Banggai cardinalfish.

The Banggai Archipelago sits at the junction of three tectonic plates (Eurasian, Indian-Australian, and Pacific-Philippine Sea) and is vulnerable to earthquakes. An earthquake measuring 7.6 on the Richter scale occurred in 2000 and destroyed coral reefs in the region (Vagelli, 2011). Frequent earthquakes within the Banggai Archipelago may have impacted localized Banggai cardinalfish populations (CITES, 2007), but specific data are lacking.

Extinction Risk

The life history characteristics (*i.e.*, low fecundity, high degree of parental care and energetic investment in offspring, high new recruit mortality, no

planktonic dispersal, high site fidelity) of the Banggai cardinalfish render it less resilient and more vulnerable to stochastic events than marine species that are able to disperse over large areas and recolonize sites that have been lost due to these events. Because the Banggai cardinalfish also has an exceptionally restricted natural range (approximately 5,500 km²), these demographic traits become more important in terms of the extent to which the threats appreciably reduce the fitness of the species. The Banggai cardinalfish lacks dispersal ability and exhibits high site fidelity, and new recruits stay within parental habitat. Thus, recolonization is unlikely once a local population is extirpated. Local populations off Liang and Peleng Island are reported extirpated, and interviews with local fishermen indicate extirpation of small local populations throughout the Banggai Archipelago. The Banggai cardinalfish also exhibits high genetic population substructuring; thus, extirpation of local populations from overharvest and/or loss of habitat can result in loss of genetic diversity and further fragmentation of spatial distribution. In considering the demographic risks to the species, its growth rate/productivity, spatial structure/connectivity, and diversity are assigned to the high risk of extinction category. However, the overall population abundance (estimated at 1.5 to 1.7 million) is assigned to the moderate risk of extinction category, because the abundance may allow some resilience against stochastic events.

In considering the threats, we rely on the best available data to assess how the threats are currently impacting or likely to impact the species in the foreseeable future. The best available data indicate that several threats to the Banggai cardinalfish will continue and increase, with the species responding negatively, but other threats will decrease, with the species responding favorably. Habitat degradation has occurred and is anticipated to continue and increase in the foreseeable future. Although Indonesia prohibits the use of chemicals or explosives to catch fish, historically, compliance has been low, and data indicate compliance is not improving. Data also indicate that by 2007, harvest of microhabitat (sea urchins and sea anemones) had negatively impacted cardinalfish populations, and the harvest had increased by 2011. Moore *et al.* (2011, 2012) concluded that it would be difficult to establish and enforce local regulations for controlling the overharvest of microhabitat. Thus, it is reasonable to expect that microhabitat harvest will continue and increase in

the foreseeable future, which negatively impacts the Banggai cardinalfish and its ability to avoid predators. Overutilization from direct harvest for the ornamental live reef fish trade has significantly impacted the Banggai cardinalfish and remains a concern. Trade continues resulting in high mortality, and in areas of heavy overexploitation, populations have been extirpated. However, an increase in compliance with the Fish Quarantine regulations and improved trade practices have occurred in recent years, and we anticipate compliance and trade practices will likely continue to improve in the future, which may mitigate impacts through sustainable trade. Participation in collection of Banggai cardinalfish for the live ornamental reef trade has dropped in recent years. Captive-bred facilities have recently started in the United States and Thailand and are anticipated to decrease the threat of directed harvest of the wild populations in the future. Predation of new recruits is high. Mortality from disease in wild-caught fish imported for the ornamental live reef fish trade and disease affecting the Banggai cardinalfish habitat are both plausible threats. However, data are lacking on how these threats impact the population and what, if any, impacts will occur and at what rate in the future. Climate change within the Banggai cardinalfish range will continue to affect coral reefs in the future, and it is reasonable to expect that future earthquakes that may destroy or modify habitat within the species' range will occur at the current rate.

The Banggai cardinalfish is exposed, and negatively responds to some degree, to the five threat factors discussed above. Although quantitative analyses are lacking, it is reasonable to expect that when these exposures are combined, synergistic effects may occur. For example, the ornamental live reef fish trade likely causes the expression of the iridovirus in the Banggai cardinalfish, which results in increased mortality. The indiscriminate harvest of sea anemones and sea urchins and destruction of coral reefs eliminates important cardinalfish shelter and substrate and increases the likelihood of predation. Interactions among these threats may lead to a higher extinction risk than predicted based on any individual threat.

In sum, based on the life history characteristics of the Banggai cardinalfish, which indicate high vulnerability to demographic risks (due to trends in population growth/productivity, spatial structure and connectivity, and diversity), coupled

with ongoing and projected threats to habitat and microhabitat, commercial use, inadequate regulatory mechanisms, disease and predation, and additional natural or manmade factors, we conclude that demographic risks and the combination of threats to the species may contribute to the overall vulnerability and resiliency of the Banggai cardinalfish. The Banggai cardinalfish has experienced a decline in abundance as evidenced by the decrease in mean density at survey sites between 2004 and 2012. Moreover, at least some researchers believe that the population may have experienced a dramatic decline from historical abundance due to overharvest based on comparisons between populations in a private bay and other populations. Most of the species' demographic characteristics put it at a high risk of extinction. However, the threat of overharvest has been and will likely continue to be reduced in the future. Further, the overall population abundance (1.5 to 1.7 million) may allow some resilience against stochastic events; thus, placing the Banggai cardinalfish at an overall moderate risk of extinction.

Protective Efforts

The Banggai cardinalfish is listed as 'endangered' by the World Conservation Union (IUCN; Allen and Donaldson, 2007). Although listing under the IUCN provides no direct conservation benefit, it raises awareness of the species. In addition, the Banggai cardinalfish was one of the first entrants into the Frozen Ark Project, which is a program to save the genetic material of imperiled species (Williams, 2004; Clarke, 2009).

In 2007, Indonesia developed a national multi-stakeholder Banggai cardinalfish action plan (BCF-AP), which focused on conservation, trade, and management issues (Ndobe and Moore, 2009). As part of the BCF-AP, annual stakeholder meetings are held to share data, review progress, and set goals (Moore *et al.*, 2011). The BCF-AP called for biophysical and socio-economic monitoring of trade, population status, and habitat, and several organizations have begun to report on these activities. However, there is no integrated or comprehensive monitoring system, and long-term data sets are lacking (Moore *et al.*, 2011). Several aspects of the BCF-AP appear to have improved the sustainability of the Banggai cardinalfish trade. Fishermen groups have gained legal status (allowing them access to various benefits such as funding or loan support), which has led to socialization of sustainable harvest in Bone Baru. The

legally-established fishermen's group Kelompok BCFLestari, in Bone Baru, implemented collection practices designed to prevent capture of brooding males (Moore *et al.*, 2011). Workshops have been held on improving capture methods and post-harvest care, and several community members have become active in conservation efforts. However, the BCF-AP officially ended in 2012 and so did the funding. Some of the stakeholders are still active and are likely to continue to be so, despite lack of government support (Moore, personal communication, 2014).

As discussed earlier, compliance with the Fish Quarantine regulations has increased, which is largely due to the development and implementation of the BCF-AP (Moore *et al.*, 2011). In 2004, one Banggai cardinalfish trader followed Fish Quarantine procedures. By 2008, there was a marked increase in legal trade, but unreported fishing still occurs (Moore *et al.*, 2011). With the lapse of the BCF-AP, legislation is needed to support and restart the goals described in the BCF-AP, and although efforts have been ongoing to establish fishing quotas and size limits, no legally binding regulations have been passed or implemented (Moore *et al.*, 2011).

In 2007, the Banggai Cardinal Fish Centre (BCFC) was established in the Banggai Laut District to serve as a central point for sharing information and managing the species over a wider community area (Lilley, 2008; Moore *et al.*, 2011). As of 2011, the BCFC had no electricity, no operational budget, and was operated on a voluntary basis (Moore *et al.*, 2011). Further inhibiting the continued operation of the BCFC is that in 2013, the region was split into two Districts by constitutional law (UU No. 5/2013). The BCFC will need to be officially approved under the new District to maintain its legal status (Ndobe, personal communication, 2014).

A marine protected area (MPA) consisting of 10 islands was declared by Indonesia in 2007, with conservation of the Banggai cardinalfish as the primary goal of the Banggai and Togong Lantang Islands (Ndobe *et al.*, 2012). However, Banggai cardinalfish populations are not found at Togong Lantang Island, while for three other islands within the proposed MPA with known populations, Banggai cardinalfish conservation is not included as a conservation goal in the designation (Ndobe *et al.*, 2012). In addition, based on genetic analysis, only 2 of 17 known populations occur within the MPA, which led Ndobe *et al.* (2012) to conclude the MPA design was ill-suited for conserving the Banggai cardinalfish.

It is uncertain whether the MPA will be changed in the foreseeable future to better suit the species.

Although no longer active, the Marine Aquarium Council (MAC), an international non-governmental organization, developed a certification system to improve the management of the marine aquarium trade. MAC developed best practices for collectors and exporters, including those in Indonesia. Best practices include improvement of product quality, reduction in mortality rates, safer practices for collectors, and fairer prices paid to collectors. By applying the MAC standards, traders could be certified as meeting these international standards (Lilley, 2008). Building on the MAC efforts, the Yayasan Alam Indonesia Lestari (LINI) has worked in the Banggai Islands to promote a sustainable fishery for the Banggai cardinalfish and to protect habitat (Talbot *et al.*, 2013). LINI focuses on surveys, capacity building, and training of local suppliers and reef restoration (Lilley, 2008). LINI's training and education efforts may raise awareness of needed conservation efforts to benefit the Banggai cardinalfish. For example, more benign collection methods have been implemented at Bone Baru, the species has been adopted as a mascot, and local citizens craft and market items related to the fish. LINI is also trying to set up a mechanism for hobbyists to buy only from distributors who use best practices and are sustainable (Talbot *et al.*, 2013). However, continued funding for the program is a concern (Moore, personal communication, 2014).

In addition to the protective efforts described above, Indonesia has committed to develop a comprehensive management plan for the Banggai cardinalfish under the auspices of Indonesia's national plan of action under the Coral Triangle Initiative on Coral Reefs, Fisheries, and Food Security (CTI-CFF). The CTI-CFF specifies a goal to use an ecosystems-based approach to managing fisheries (EAFM), including a more sustainable trade in live reef fishes. In 2013, World Wide Fund for Nature (WWF), in partnership with STPL, implemented a pilot project in Central Sulawesi Province under the ecosystems-based approach and chose the Banggai cardinalfish as one of five fisheries case studies in Banggai Laut District. The goal is to draft local regulations for an EAFM for two Districts—Banggai Laut District (which encompasses the majority of the endemic Banggai cardinalfish populations) and Banggai Kepulauan District (which includes the Peleng Island Banggai cardinalfish

populations). The STPL EAFM Learning Centre team will be implementing this component through January 2015. These efforts are likely to introduce local measures to sustain the Banggai cardinalfish trade (Moore, personal communication, 2014; Ndobe, personal communication, 2014).

Under the PECE, conservation efforts not yet implemented or not yet shown to be effective must have certainty of implementation and effectiveness before being considered as factors decreasing extinction risk. The effort described above does not satisfy the PECE criteria of having a certainty of implementation and effectiveness. Although a pilot project in Central Sulawesi Province under the ecosystems-based approach is underway with the Banggai cardinalfish as one of five fisheries case studies, we lack information on how this effort will yield measures that will be funded, regulated, or regularly practiced to sustain the Banggai cardinalfish trade in the future; thus, this effort cannot be considered to alter the risk of extinction of the Banggai cardinalfish. We seek additional information on other conservation efforts in our public comment process (see below).

Proposed Determination

Based on the best available scientific and commercial information discussed above, we find that the Banggai cardinalfish is at a moderate risk of extinction, but the nature of the threats and demographic risks identified do not suggest the species is presently in danger of extinction, and therefore, it does not meet the definition of an endangered species. We do find, however, that both the species' risk of extinction and the best available information on the extent of and trends in the major threats affecting this species (habitat destruction and overutilization) make it likely this species will become an endangered species within the foreseeable future throughout its range. We therefore propose to list it as threatened under the ESA.

Harrison's Dogfish

The following section describes our analysis of the status of the gulper shark, Harrison's dogfish (*Centrophorus harrissoni*). More details can be found in Miller (2014).

Species Description

Centrophorus harrissoni, or Harrison's dogfish, is a shark belonging to the family Centrophoridae (order Squaliformes). The Centrophoridae contain two genera: *Deania* (long-snouted or bird-beak dogfishes) and

Centrophorus, usually referred to as gulper sharks. “Gulper shark” is also the common name for the largest species, *C. granulatus* (White *et al.*, 2013).

Harrison’s dogfish is endemic to subtropical and temperate waters off eastern Australia and neighboring seamounts. Specimens identified as *C. harrissoni* have also been collected along the Three Kings, Kermadec, and Norfolk Ridges north of New Zealand, and it has also possibly been identified off New Caledonia (Duffy, 2007). It is a demersal species, primarily found along the upper- to mid-continental and insular slopes off eastern Australia, from north of Evans Head in northern New South Wales (NSW) to Cape Hauy on the island of Tasmania, and on the Tasmantid Seamount Chain off NSW and southern Queensland (hereafter referred to as its “core range”). It occurs in depths of 180 to 1000 m, with a principal depth range of 200 to 900 m (White *et al.*, 2008; Last and Stevens, 2009; Williams *et al.*, 2013a). However, specimens have been collected in deeper waters from the seamounts and ridges north of New Zealand and off southeastern Australia and in shallower depths off eastern Bass Strait (Daley *et al.*, 2002; Graham and Daley, 2011; Williams *et al.*, 2013a). Gulper sharks, including Harrison’s dogfish, are thought to conduct diel vertical feeding migrations, whereby the sharks ascend the continental slope near dusk to around 200 m depths to feed and then descend before dawn (Williams *et al.*, 2013a), which helps to explain the large depth distribution for the species. Small bathypelagic bony fishes (particularly myctophids, lantern fishes), cephalopods, and crustaceans have been found in the stomachs of *C. harrissoni* (Daley *et al.*, 2002).

Research studies indicate that *C. harrissoni* may also exhibit spatial sexual segregation (Graham and Daley, 2011), based on the evidence that males tend to dominate the sex ratios on survey grounds and assumption that females must be more abundant elsewhere to compensate for the uneven sex ratios. Specifically, sex ratios varied from 1.5:1 to 4.9:1 along the east coast of Australia, illustrating the predominance of males (Graham and Daley, 2011). Two notable sites, however, did show females outnumbering males and were located off northern NSW, from Newcastle to Danger Point, and off Taupo Seamount (Graham and Daley, 2011), providing some support for spatial sexual segregation. Interestingly, Graham and Daley (2011) found no evidence of sexual or age segregation by depth, with males dominating throughout all depth

zones sampled (with the exception of the two sites noted above) and juveniles evenly interspersed with adults across all depths.

In terms of mating and reproductive behavior, which could provide some insight into potential spatial structuring, very little information is available. It is known that Harrison’s dogfish is viviparous (*i.e.*, gives birth to live young), with a yolk-sac placenta. Females have litters of one or (more commonly) two pups, with size at birth around 35–40 cm TL (Graham and Daley, 2011). Although the gestation period is unknown, a 2 to 3 year period has been estimated for other *Centrophorus* species, with continuous breeding from maturity to maximum age (Kyne and Simpfendorfer, 2007; Graham and Daley, 2011). Female *C. harrissoni* mature at sizes around 98 cm TL and reach maximum sizes of 112–114 cm TL, while males mature around 75–85 cm TL and reach maximum sizes of 95–99 cm TL (Graham and Daley, 2011). Female age at maturity is estimated between 23 and 36 years of age (Daley *et al.*, 2002; Wilson *et al.*, 2009; Last and Stevens, 2009; Graham and Daley, 2011). Longevity is estimated at over 46 years of age (Wilson *et al.*, 2009). Current breeding sites for Harrison’s dogfish are thought to include waters off eastern Australia, from Port Stephens to 31 Canyon, areas off North Flinders and Cape Barren in southeastern Australia, and waters around Taupo Seamount (Williams *et al.*, 2012). These are areas where mature males, mature females, and juveniles have been recorded, and thus are likely to be areas that support viable populations where mating and pupping occur (Williams *et al.*, 2012). However, more extensive sampling, as well critical information regarding the aspects of the Harrison’s dogfish breeding cycle (including necessary sex ratios for successful reproduction, preferred mating and breeding grounds, and mating and breeding behaviors), is needed to identify and fully comprehend the spatial dynamics of Harrison’s dogfish.

For management purposes, Harrison’s dogfish in Australia have been separated into two stocks that are considered to be “distinct” populations: A “continental slope” stock that occurs continuously along the Australian eastern continental margin, and a “seamount stock” that occurs on the Tasmantid Seamount Chain off NSW and southern Queensland, including the Fraser, Recorder, Queensland, Britannia, Derwent Hunter, Barcoo, and Taupo Seamounts. However, to date, no genetic studies have been conducted to confirm that these two populations are

genetically distinct, and tagging studies are limited, with insufficient recapture rates to make any determination regarding the connectivity of the populations. In addition, there are a number of other uncertainties associated with the assumption of two separate Harrison’s dogfish stocks, including necessary sex ratios and other successful reproduction requirements, which are further discussed in Miller (2014). Due to these uncertainties, we do not find conclusive evidence of separate populations of Harrison’s dogfish. Therefore, we consider the available information for these two stocks, including estimates of depletion rates and protection benefits of management measures, together when we determine the status of the entire species throughout its range.

Because species-specific historical and current abundance estimates are not available, Williams *et al.* (2013a) used a variety of methods and analyses to estimate the pre-fishery (pre-1980s) and current abundance (in biomass units) at fishery stock and sub-regional scales (detailed information on the data sources and methods can be found in Williams *et al.* (2013a)). Results from the various analyses revealed that Harrison’s dogfish is currently estimated to be at 21 percent of its pre-fishery population size throughout its core range (with a lower estimate of 11 percent and upper estimate of 31 percent). The authors note that this overall estimate of decline is strongly influenced by the small declines estimated on seamounts (Williams *et al.* 2013a). The continental margin population is estimated to be at 11 percent of its pre-fishery population size (range of 4 to 20 percent; with the estimate influenced by uncertainty surrounding the level of cumulative fishing effort off the northern NSW slope). The seamount population is estimated to be at 75 percent of its pre-fishery population size (range 50 percent to 100 percent).

Summary of Factors Affecting Harrison’s Dogfish

Available information regarding current, historical, and potential threats to Harrison’s dogfish were thoroughly reviewed (Miller, 2014). We find that the main threat to the species is overutilization for commercial purposes, with the species’ natural biological vulnerability to overexploitation exacerbating the severity of the threat, and hence also identified as a secondary threat contributing to the species’ risk of extinction. We summarize information regarding these threats and their

interactions below, according to the factors specified in section 4(a)(1) of the ESA. Available information does not indicate that habitat destruction, modification, or curtailment, disease, or predation are operative threats on this species; therefore, we do not discuss those further here. Because new regulatory measures were just recently implemented, the adequacy and effectiveness of existing regulatory measures is discussed in the “Protective Efforts” section below. See Miller (2014) for full discussion of all threat categories.

Overutilization for Commercial, Recreational, Scientific, or Educational Purposes

Historically, Harrisson’s dogfish and other gulper sharks were taken in both Australian Commonwealth-managed commercial trawl fisheries (those that are managed by the Australian Federal Government, in coordination with Australian State fisheries agencies, through the Australian Fisheries Management Authority (AFMA) (Kyne and Simpfendorfer, 2007)) and State-managed commercial trawl fisheries operating on the upper slope off eastern Australia, within the core range of Harrisson’s dogfish. Unfortunately, little information is available on the specific catch of these deep-water sharks, primarily due to the historical inaccuracy of data reporting and species identification issues. These Commonwealth and State-managed commercial trawl fisheries developed off NSW in the 1970s and off Victoria and Tasmania in the 1980s. By the early 1980s, more than 100 trawlers were operating off NSW, with around 60 percent regularly fishing on the upper slope. In fact, between 1977 and 1988, catches from these upper-slope trawl operations comprised more than half of the total trawl landings in NSW (Graham *et al.*, 2001). Large numbers of *C. harrissoni* were likely caught and discarded off NSW during this time, due to the absence of a market for deepwater shark carcasses (a result of mercury content regulations and preference for more marketable bony fishes) (Daley *et al.*, 2002; Graham and Daley, 2011). Similarly, trawlers operating on the upper-slope off eastern Victoria reported minimal catches of *Centrophorus* dogfishes, but also likely discarded substantial numbers due to Victorian State restrictions on mercury content in shark flesh (Daley *et al.*, 2002). Graham and Daley (2011) estimate that landings of *Centrophorus* spp. were around several hundred tonnes per year during the 1980s and early 1990s.

Daley *et al.* (2002) note that in the early 1990s significant quantities of *Centrophorus* spp. were also caught off eastern Victoria by fishermen using droplines targeting blue-eye trevalla (*Centrolophus antarctica*) and ling (*Genypterus blacodes*). In addition, some Southern and Eastern Scalefish and Shark Fishery (SESSF) operators off Victoria used deep-set gillnets to target *Centrophorus* species for their livers in the 1990s (Daley *et al.*, 2002). Squalene oil, which is extracted from the liver of deep-sea sharks, is used in a number of cosmetics and health products, and the livers of *Centrophorus* species have the highest squalene oil content (67–89 percent) of any deep-sea shark. Fishermen would keep the livers of the *Centrophorus* spp. and discard the carcasses due to their mercury content. However, by the time the mercury restrictions were eased in 1995 (allowing for carcasses to also be sold), very few *Centrophorus* species were being caught off eastern Victoria, with targeting of these sharks having essentially ceased (Daley *et al.*, 2002). Since 2002, total catch of gulper sharks by Commonwealth licensed vessels has been less than 15 t per year (Woodhams *et al.*, 2013).

In 2001, Graham *et al.* (2001) quantified the effects of the historical trawling on the abundance of gulper sharks off NSW using data from fishery-independent surveys conducted along the upper slope before and after the expansion of the commercial trawl-fishery (Andrews *et al.*, 1997). The initial pre-fishery survey was carried out during 1976 and 1977. There were three trawling survey grounds: (1) Sydney-Newcastle, (2) Ulladulla-Batemans Bay, and (3) Eden-Gabo Island and eight depth zones (covering depths of 200–650 m). The two northern grounds (Sydney and Ulladulla) were surveyed twice in 1976 and twice in 1977; the southern (Eden) ground was surveyed three times in 1977. These surveys were repeated in 1996–1997, (with two surveys conducted off Sydney and Ulladulla and three off Eden) using the same vessel and trawl gear and similar sampling protocols, to examine the changes in relative abundances of the main species (number and kg per trawling hour) after 20 years of trawling (see Andrew *et al.*, 1997; Graham *et al.*, 2001). Results from these surveys show that Harrisson’s dogfish were present and, at one time, were caught across all of the survey grounds and depth zones. In 1976, catches of Harrisson’s dogfish were combined with southern dogfish (*C. zeehaani*) in the initial two surveys off Sydney and one off Ulladulla. When

these species were separated in the later 1976 surveys, and in 1977, southern dogfish comprised around 75 percent and Harrisson’s dogfish comprised 25 percent of the combined catch. In 1976–77, Harrisson’s and southern dogfishes combined represented around 9 percent, 18 percent, and 32 percent of the total fish catches off Sydney, Ulladulla, and Eden, respectively. The overall mean catch rate (for all grounds and depths) was 126 kg/hour. This is in stark contrast to the 0.4 kg/h catch rate in 1996–1997, when only 14 southern and 8 Harrisson’s dogfishes were caught, comprising 0.18 percent of the total fish catch weight (Graham *et al.*, 2001). For the 1976–77 surveys where the two species were separated, the mean catch rate of Harrisson’s dogfish was 28.8 kg/hr caught over the course of 173 tows. In 1996–97, the mean catch rate of Harrisson’s dogfish was 0.1 kg/hr over the course of 165 tows (Graham *et al.*, 1997; 2001). These decreases in survey catch rates provide compelling evidence of declines of over 99.7 percent in relative abundance of *C. harrissoni* on the upper-slope of NSW, a core part of their range, after 20 years of trawling activity (Graham *et al.*, 2001).

In Australia, the commercial trawl fisheries are still active, as are demersal line fisheries, which also incidentally catch Harrisson’s dogfish. In terms of Commonwealth-managed fisheries, Harrisson’s dogfish are primarily caught as bycatch by the SESSF, which operates over an extensive area of the Australian Fishing Zone (AFZ) around eastern, southern, and southwestern Australia. The distribution of recent (2006–2010) commercial fishing effort in the SESSF shows that there is still substantial fishing effort on Commonwealth upper-slope grounds using demersal gears, specifically trawl and auto-longline operations (see Miller (2014) for more details). According to Graham (2013), around 30 trawlers and 3 auto-longliners in the SESSF still operate along the upper-slopes. Since auto-longline vessels, which deploy up to 15,000 hooks per vessel per day, can operate on the steep and rough ground that would potentially be a refuge for *C. harrissoni* from trawling (R. Daley, Commonwealth Scientific and Industrial Research Organization (CSIRO), personal communication, 2014), the combined operation of both the trawl and auto-longline fisheries within the range of Harrisson’s dogfish significantly increases the likelihood of incidental catch of the species. Catch rates of Harrisson’s dogfish in the SESSF have been minimal in recent years, likely due to their low abundance

on the continental margin; however, the combined operation of these demersal gears on the upper-slope grounds may further decrease abundance of the remaining population. For the 2012–2013 season, reported gulper shark (*C. harrissoni*, *C. moluccensis*, *C. zeehaani*) landings (in trunk weight) were 0.9 t with discards of 1.2 t (Woodhams *et al.*, 2013). This is a decrease from the previous year, which reported landings of 3.8 t. Given the evidence of substantial depletion of both Harrison's and southern dogfishes in Australian waters over the years, high risk of overfishing in the SSSF, with no current indication of recovery (based on 2012–2013 season data), the Australian Government Department of Agriculture classified the above three gulper sharks as “overfished” in 2012, with the current level of fishing mortality noted as “uncertain” (Woodhams *et al.*, 2013). In fact, upper-slope gulper sharks have been classified as overfished since they were first included in Australia's Fishery Status Reports in 2005 (Woodhams *et al.*, 2011). In February 2013, a zero retention limit was implemented for Harrison's dogfish (Woodhams *et al.*, 2013), along with other management measures detailed in AFMA's Upper-Slope Dogfish Management Strategy (AFMA, 2012) and evaluated in the “Protective Efforts” section below.

In terms of state-managed fisheries, the range of Harrison's dogfish extends within NSW, Victoria, and Tasmania jurisdictions. In both Victorian and Tasmanian fisheries, catch records of Harrison's dogfish are rare and interactions with these fisheries are considered to be unlikely, based on their respective fishing operations (Threatened Species Scientific Committee (TSSC), 2013). In NSW commercial fisheries, Harrison's dogfish may be caught by the Ocean Trap and Line Fishery and the Ocean Trawl Fishery. According to Graham (2013), there are up to five trawlers in the Ocean Trawl Fishery that fish seasonally between Newcastle and Sydney and may incidentally catch Harrison's dogfish, and only minimal line fishing effort on the upper-slope (K. Graham, Australian Museum, personal communication, 2014). In 2013, a zero retention limit was implemented for Harrison's dogfish (unless for scientific purposes as agreed by Fisheries NSW) (NSW DPI, 2013).

Because of their low productivity, sustainable harvest rates of gulper sharks are estimated to be less than five percent of their virgin biomass, and maybe even as low as one percent (reflecting the proportion of total

population that can be caught and still maintain sustainability of the population; Forrest and Walters, 2009). However, these harvest levels can only be sustained by a population in a significantly less depleted state (Woodhams *et al.*, 2011). In the case of Harrison's dogfish, Woodhams *et al.* (2013) notes that even low levels of mortality can pose a risk because of its significantly depleted state. Although total fishing mortality on gulper sharks is unknown, the level of catch and observed discards in recent years was deemed likely to result in further population declines (Woodhams *et al.* 2011; 2012; 2013). In the 2012–13 fishing season, discards actually outnumbered landings (1.2 t compared to 0.9 t; Woodhams *et al.*, 2013). Thus, even with the prohibition on retention of the species, there is still a potential for discards based on the significant overlap of current fishing effort within the core range of the species (Woodhams *et al.*, 2013). This is a concern because Harrison's dogfish suffers from high at-vessel mortality in trawl gear and potentially high at-vessel mortality in auto-longline gear (Williams *et al.*, 2013a). Therefore, the continued fishing effort on the upper-slope and potential for incidental capture of Harrison's dogfish in the trawl and line fisheries described above, which will likely result in mortality of the species, is considered a threat that is currently contributing to the overutilization of the species and its risk of extinction.

In the areas off New Zealand where *C. harrissoni* have been observed (Three Kings Ridge, Norfolk Ridge, and Kermadec Ridge), there is limited fishing effort (Graham, 2013). The fishing activities include trawling on the West Norfolk Ridge, drop-lining for large bony fishes on the Three Kings Rise, West Norfolk Ridge, and Wanganella Bank, and minimal longlining and close to no trawling on the Kermadec Ridge. No bycatch of gulper sharks has been reported from these fishing activities (based on a personal communication from C. Duffy in Graham (2013)). Given the uncertainty surrounding the *C. harrissoni* abundance in this area, it is currently unknown if these fishing activities are impacting Harrison's dogfish populations or significantly contributing to its extinction risk (Graham, 2013).

Other Natural or Manmade Factors Affecting the Continued Existence of Harrison's Dogfish

Many sharks are biologically vulnerable to overexploitation due to

their life history parameters. Species with slow population growth rates, late age at maturity, long gestation times, low fecundity, and higher longevity are especially sensitive to elevated fishing mortality (Musick, 1999; García *et al.*, 2008; Hutchings *et al.*, 2012). These life history traits increase the species' susceptibility to depletion by decreasing the species' ability to rapidly recover from exploitation. Harrison's dogfish exhibits these same life history traits, with late maturity, long gestation times, small litter sizes, and high longevity. These life history traits have exacerbated the overall impact of the historical overutilization of the species on its extinction risk, leading to the substantial decline in Harrison's dogfish abundance, and will continue to place the species at increased risk of demographic stochasticity.

Extinction Risk

It is clear that the species faces current demographic risks that greatly increase its susceptibility to extinction. Due to the significant decline, the species is no longer found in approximately 19 percent of its Australian range and, furthermore, throughout the rest of its core range, is estimated to be at 21 percent of its total virgin population size (with separate estimates of 11 percent for the continental margin population and 75 percent for the seamount population) (Williams *et al.*, 2013a). Although the population on the seamounts may be less depleted, it also likely comprises a significantly smaller portion of the entire Harrison's dogfish population, based on the amount of available habitat and corresponding carrying capacity. In fact, the continental margin habitat, where the population is estimated to be at only 11 percent of its total virgin population size, represents 86 percent of Harrison's dogfish's estimated extent of occurrence and 84 percent of its estimated area of occupancy (TSSC, 2013), indicating significant depletion throughout most of the species' range. In addition, the existing Harrison's dogfish populations along the continental margin and off the seamounts in Australia and New Zealand are small and fragmented, with only three identified remnant populations that are thought to be viable (due to presence of mature males, females, and/or juveniles within the same area). Two of these populations are located off the continental margin and the third is off Taupo Seamount. It is unclear the extent to which these populations can help recover Harrison's dogfish, as breeding behavior, stock structure, inter-

population exchange, and general movement of individuals is currently unknown. Due to their size and isolation, these populations may be at an increased risk of random genetic drift and could experience the fixing of recessive detrimental alleles that could further contribute to the species' extinction risk (Musick, 2011). In addition, the patchy distribution of these populations throughout the species' entire range increases susceptibility to local extirpations from environmental and anthropogenic perturbations or catastrophic events. Given the apparent spatial structuring of the species and dominance of males in the sex ratios at many locations, a further reduction in the numbers of females at any given site may decrease reproductive success and prevent population replacement. The species has extremely low fecundity (2–3 year gestation period resulting in 1 to 2 pups), slow growth rates, and late maturity, all of which contribute to a long population doubling time. In a severely depleted state, these traits may contribute to increasing the species' extinction risk, especially if the species is still subject to threats that further reduce its abundance. Thus, although the species' biological characteristics have allowed it to successfully thrive in the past, under the current conditions of severely fragmented populations and low abundance throughout its range, questionable population viability, and risk of incidental mortality from fisheries, the species' natural life history traits are presently threatening its continued existence. Specific information is lacking on interactions among threats.

Without considering the effectiveness of the recently implemented management measures in reducing the threat of overutilization and improving the status of Harrisson's dogfish in Australian waters (discussed in the "Protective Efforts" section below), Miller (2014) concluded that Harrisson's dogfish is presently at a high risk of extinction due to threats of overutilization exacerbated by its natural biological vulnerability to depletion, the interaction of which has resulted in significant demographic risks to the species. We agree with this analysis and find that the species is presently in danger of extinction throughout its range. Below we evaluate formalized conservation efforts that have yet to be implemented or to show effectiveness to determine whether these efforts contribute to making listing the species as endangered unnecessary.

We evaluate these conservation efforts using the criteria outlined in PECE.

Protective Efforts

The EPBC Act, the Australian Government's central piece of environmental legislation, applies to any group or individual whose actions may have a significant impact on a "matter of national environmental significance." Any proposed action that meets this standard must then be assessed to determine its environmental impact. Species listed as "vulnerable," "endangered," and "critically endangered" under the EPBC Act are considered to be matters of national environmental significance and receive these provisions.

In 2009, Harrisson's dogfish was nominated for listing under the EPBC Act. Its status was reviewed by the Threatened Species Scientific Committee (TSSC), a committee established under the EPBC Act to advise the Australian Minister for the Environment on the amendment and updating of lists of threatened species, threatened ecological communities, and key threatening processes, and with the making or adoption of recovery plans and threat abatement plans. In 2013, the TSSC concluded that Harrisson's dogfish was eligible for listing as endangered under the EPBC Act because the species had suffered a severe reduction in numbers, with a suspected population decline of between 74 and 82 percent (TSSC, 2013). However, the TSSC concluded that the species was also eligible for listing as a conservation dependent species under the EPBC Act because it is the "focus of a plan of management [the Strategy] that provides for managed actions necessary to stop the decline of, and support the recovery of, the species so that its chances of long term survival in nature are maximized" (TSSC, 2013). In May 2013, based on the TSSC recommendation, the Minister of the Environment officially listed Harrisson's dogfish as a conservation dependent species under the EPBC Act. This listing means that the species is not considered a matter of national environmental significance in the context of the EPBC Act, and, as such, Harrisson's dogfish are exempt from the EPBC Act protective provisions.

In 2012, AFMA published the Upper-Slope Dogfish Management Strategy (the "Strategy"; see AFMA, 2012) to satisfy the aforementioned management requirements for a conservation dependent listing of Harrisson's Dogfish and Southern Dogfish under Australia's EPBC Act. The Strategy, which we evaluate below according to the guidelines in the PECE (68 FR 15100;

March 28, 2003), includes regulatory management measures designed to rebuild the Harrisson's dogfish population above a limit reference point of 25 percent of its unfished biomass (B_{25}). Setting a recovery time frame was deemed not feasible until further research on the species is completed; however, an interim time frame to reach this reference point was estimated based solely on the biological characteristics of the species (three generation times) and equal to 85.5 years (SWG, 2012).

The outcomes and the effectiveness of the Strategy are expected to be measured on a biennial basis, as detailed in AFMA's "Upper-Slope Dogfish Research and Monitoring Workplan." The workplan for the period of 2014–2016 (Workplan 1) focuses on the development of a cost-effective method for measuring baseline relative abundance of gulper sharks and recovery over time (AFMA, 2014). This output will be assessed as part of the Research and Monitoring Workplan 2014–16 review (proposed time frame of July 2014–Dec 2016). Once the methodology has been developed, the next output (Workplan 2) is expected to produce baseline relative abundance estimates for Southern and Harrisson's dogfish (proposed time frame for output: Jan 2017–Dec 2019). Subsequent workplans will provide estimates of rebuilding over time and will be periodically assessed to ensure that the actions within the workplans are achieving the desired outputs. Hence, it appears it will be a number of years before the effectiveness of the Strategy will be able to be quantified. As outlined in the PECE, we must evaluate these conservation efforts that have not yet demonstrated effectiveness at the time of listing to determine whether these efforts are likely to be effective at reducing or eliminating threats and improving the status of Harrisson's dogfish. Below are the regulatory measures from the Strategy that have already been implemented by AFMA for the conservation of the species (under the legal authority of section 41A of the Australian Fisheries Management Act 1991 and implemented under "SESSF Fishery Closures Direction No. 1 2013;" satisfying the first criteria of the PECE) and our subsequent evaluation of their likely effectiveness at improving the status of Harrisson's dogfish (the second criteria of the PECE). The figures and tables referenced below can be found in the PECE supplement (Miller, 2014b).

Prohibition on the Commercial Retention of Gulper Sharks

The Strategy implements a complete prohibition on the commercial retention

of all gulper sharks. However, even before the prohibition, reported catch rates of Harrison's dogfish in the SESSF have been minimal in recent years, likely due to the low abundance of the species on the continental margin where the fisheries operate. Harrison's dogfish are not a targeted species, but rather taken as incidental catch. Although this prohibition will decrease the numbers of sharks being landed, it is worth noting that discards have outnumbered landings in recent years and at a rate that was deemed likely to result in further declines of the species (Woodhams *et al.*, 2011). Additionally, in the latest Fishery Status Report for Commonwealth-managed fish stocks, it states: "[t]here is potential for unreported or underestimated discards (based on the large degree of overlap of current fishing effort with the core range of the species [Harrison's dogfish]), and low levels of mortality can pose a risk for such depleted populations" (Woodhams *et al.*, 2013). Based on the above discarding trends, the fact that it is the Commonwealth Trawl Sector of the SESSF which is the main fishery operating within the species' core continental margin range, and the evidence that Harrison's dogfish are not expected to survive after incidental capture in trawl gear (Rowling *et al.*, 2010), the new retention prohibition may only have a minor impact on decreasing current fisheries-related mortality.

Network of Spatial/Area Closures

Prior to the Strategy, a number of closures were implemented across the SESSF operational area (AFMA, 2012); however, there were concerns that these closures were too small in relation to the historical distribution of the species to prevent further declines or recover the species (Musick, 2011; Woodhams *et al.*, 2011). Musick (2011) estimated that the closures protected Harrison's dogfish from all forms of industrial fishing in only 9.8 percent of its habitat. In response to these concerns, AFMA evaluated options for closures in the Strategy and created a new network of spatial/area closures in 2013, taking into account the species' distribution and habitat potential, which would protect the species from various forms of fishing and prevent further declines.

Regulations that are the most effective in protecting the species from threats of overutilization (*i.e.*, incidental catch) are those that prohibit all types of fishing methods. An analysis of already implemented conservation efforts from the Strategy estimates that 26.3 percent of the core Harrison's dogfish seamount habitat (weighted by carrying capacity—

the habitat area's ability to support dogfish populations) and 5.5 percent of the continental margin habitat are closed to all types of fishing (see Table 1; Figures 1 and 4 in Miller, 2014b). In terms of the areas that support Harrison's dogfish populations, this coverage translates to protection for 26.3 percent of the current biomass of the seamount population (provided by the new Derwent Hunter closure) and 19.1 percent of the biomass of the continental margin population. Contributing to the protection of the continental margin population are the Strategy's extension of the Flinders Research Zone closure and revision to the Harrison's Gulper closure that prohibits fishing in the depth range of Harrison's dogfish. The fact that these closures encompass areas critical to population viability further increases the effectiveness of this regulation in improving the status of the species. For example, the Extended Flinders Research Zone (see Figures 2a and 2b in PECE supplement) protects the only known potentially reproducing population of Harrison's dogfish found south of Sydney. Specifically, this closure protects the mature male population found around Babel Island, the mature female population found around Cape Barren, and the likely migration route between these two populations that is thought to support mating activities (Middle Ground). Prior to this closure, only the Babel and Cape Barren grounds were protected, leaving the closely adjacent Trawl Corridor and Middle Ground open to fishing activities (and the potential for incidental catch). Now, this closure has been extended and prohibits all fishing methods from 200 to 1000 m deep, covering the entire depth range of Harrison's dogfish.

If we also consider closures that prohibit all high-risk fishing methods (permitting only power hand-line), the protection coverage increases to 24 percent of Harrison's dogfish's entire core habitat (see Table 1; Figures 1–4 in Miller, 2014b). The effectiveness of these regulations in improving the status of Harrison's dogfish partly depends on the handling of the species in fishing gear and subsequent post-release mortality rates of the shark. In other words, these regulations are only likely to be effective in decreasing threats if they reduce incidental catch altogether or reduce mortality rates of Harrison's dogfish when incidentally caught. As these closures prohibit all fishing with the exception of power-handline methods, we need to consider the selectivity and post-release mortality

of power-handline methods on Harrison's dogfish in order to evaluate the effectiveness of these closures. Based on findings from Graham (2011) and Williams *et al.* (2013b), there is a high selectivity rate for target species (and consequently low bycatch) when using the power handline technique. For example, in one of the experiments designed to replicate normal power-handline fishing operations for harvesting blue-eye trevalla (the target species for power-handline fishing), results showed that Harrison's dogfish could be successfully avoided. Out of a total of 1,435 individual line drops, 25,509 hooks, and over 10 fishing trips, no Harrison's dogfish were taken as bycatch. This is in contrast to the 6,819 blue-eye trevalla that were caught using the power-handline method (Williams *et al.*, 2013b). Likely contributing to this high degree of selectivity using the power handline method and avoidance of Harrison's dogfish is the fact that fishing for blue-eye trevalla is normally conducted during daylight hours, in depths of 280–550 m. Based on Harrison's dogfish's diel-migration patterns, the species is normally found in depths greater than 550 m during daylight hours, deeper than the normal power handline operating depths.

Insight into post-release mortality was also provided from the Williams *et al.* (2013b) study, as exploratory fishing for Harrison's dogfish was conducted to determine the occurrence of the species on the seamounts. A total of 105 Harrison's dogfish were captured during this exploratory component of the survey and Williams *et al.* (2013b) observed that many of these sharks, when brought to the surface, were in good physical condition. All but one shark were released back into the water alive and actively swam away. Williams *et al.* (2013b) attribute this potentially low post-release mortality to the short soak times associated with power-handline fishing. In addition, this type of fishing method consists of a high degree of spatial targeting and small gear size, which also likely contribute to a high survival rate of Harrison's dogfish when caught on lines (Williams *et al.*, 2013b). Based on these findings, we consider closures that prohibit all high-risk fishing methods (permitting only power hand-line), as effectively decreasing the threat of overutilization (*i.e.*, mortality from incidental catch) of Harrison's dogfish (see Table 1; Figures 1–4 in Miller, 2014b). The coverage of these closures, when broken out by continental margin and seamount proportions and weighted by carrying capacity, translates to protection for

Harrison's dogfish over 18.4 percent of its core continental margin habitat and 77.6 percent of its seamount habitat (see Table 1 in Miller, 2014b). Contributing to the protection of the continental margin population is the Strategy's extension of the Endeavour closure, and for the seamount population, the newly created Queensland and Britannia seamount closures.

If we look at the closures that prohibit trawling operations next, it is estimated that 29.5 percent of the species' core habitat range is protected from trawling activities (see Table 1 in Miller, 2014b). With these regulations, almost all of the Harrison's dogfish's core seamount habitat would be protected. As Harrison's dogfish are not expected to survive when caught in trawl gear, these closures are likely to be effective in decreasing mortality rates from incidental catch in trawls. In fact, there is already evidence of rebuilding in areas that were extensively trawled but have seen significantly less activity recently. Graham and Daley (2011) note the presence of a high numbers of juveniles (<80 cm TL, including neonates) that were caught during a 2009 long-line survey at sites off Port Stephens NSW. This area had been extensively trawled during the first 20 years of the upper-slope fishery, but over the last 10 years has seen significantly less trawling activity (Graham *et al.*, 2001; Graham and Daley, 2011). The authors of the study attribute the increase in juvenile sightings as potentially a re-establishment of the population in this area.

NSW closures and regulations may also offer additional protection to the species (TSSC, 2013). Specifically, the NSW "North of Sydney closure" (see Figure 3 in Miller, 2014b) prohibits all fishing methods except for power-handline, but allows trawling in depths over 650 m (which overlaps with the Harrison's dogfish depth range). The NSW trawl restriction areas 4 and 6 (see Figure 5 in Miller, 2014b) also provide some protection by prohibiting trawling, but are open to line methods. Overall, these additional regulations protect 2.4 percent of the core habitat (and 3 percent of the core continental margin habitat), mainly from trawling, except at the shallowest depths (TSSC, 2013).

Many uncertainties surround these estimates. We currently do not know the locations of important foraging grounds or nursery areas that are critical for population viability. In addition, we have no information regarding the movement of Harrison's dogfish in and out of these protective closures, or the connectivity between the seamounts and continental margin populations.

However, preliminary tagging studies of a closely related species, *C. zeehaani*, inside a fishery closure off southern Australia suggest that the home ranges of deep-water dogfish sharks may be small, with evidence of resident female populations that can be effectively protected by fishery closures (Daley *et al.*, 2014). Furthermore, as new information becomes available that improves the understanding of Harrison's dogfish biology and stock structure, the management arrangements in the Strategy can be adapted as necessary to ensure the effectiveness of the Strategy over time.

Compliance and Enforcement

In addition to the actual spatial extent of the closure network, the certainty of effectiveness of these regulatory measures in decreasing threats to the species also depends on the compliance and enforcement of these closures. For the Commonwealth fisheries, AFMA has created a compliance team to assist with issues such as quota evasion and balancing, Vessel Monitoring System (VMS) requirements, and compliance with fisheries closures and interactions with protected species. In terms of VMS requirements (a key monitoring provision in the Strategy), compliance rates have significantly increased over the years, thanks to outreach material to vessel operators. Compliance rates for the requirement for vessels to have an operational VMS averaged around 97 percent for the 2012–2013 year (AFMA, 2013a).

Another key to the successful and effective conservation of the Harrison's dogfish population so that it may rebuild in the future is compliance with fishing prohibitions inside closures. In 2010–2011, AFMA identified the activity of fishing boats entering and/or fishing inside closures as an occasional but significant risk. To combat this, they developed a "show cause" program whereby breaches inside closures were identified from VMS, and the operators of these vessels were sent a letter asking them to explain or "show cause" for their activity. Within a year of running the program, the incidence of fishing or navigating inside fishery closures had decreased from an average of 11 breaches per month to less than 2 breaches per month (AFMA, 2013b).

Conclusion

After consideration of the evaluation criteria for certainty of effectiveness under the PECE, we find that these existing regulatory measures are likely to be effective in improving the present status of the species. The network of implemented closures addresses the

threat of overutilization by prohibiting high-risk fishing methods, which decreases fishery-related mortality from bycatch. Based on a prior review by Musick (2011), it was recommended that closures include at least 20 to 35 percent of important Harrison's dogfish habitat in order to prevent further decline of the species and potentially support recovery. Overall, the closures evaluated above appear to provide the species with effective protection from high-risk fishing methods over 24 percent of its core habitat range (see Table 1 in Miller, 2014b). Specifically, the core habitat of the much-less-depleted seamount population is significantly protected from high-risk fishing methods and almost entirely protected (98.2 percent) from trawling activities (see Table 1 in Miller, 2014b). In fact, 77.6 percent of the seamount population biomass is protected from all high-risk fishing methods by the new closures created by the Strategy. These conservation efforts are likely to effectively improve and protect the status of this population so that it is no longer presently in danger of extinction. In terms of the continental margin population, the new network of spatial closures provides protection from high-risk fishing methods over 18.4 percent of the core margin habitat. The closures protect 32.4 percent of the current biomass, including the only known viable population found south of Sydney, from all fishing activities, which will be critical for improving the status of the population (see Table 1; Figure 1 in Miller, 2014b). Although incidental fishing mortality may occur outside of these closures, based on the best available information, we consider the current network of closures effective in adequately decreasing the present threat of overutilization throughout the species' range to the point where the species is not currently in danger of extinction.

As mentioned previously, these conservation efforts have been designed with the explicit objective to stop the decline of Harrison's dogfish and rebuild the population above 25 percent of its unfished biomass. AFMA's "Upper-Slope Dogfish Research and Monitoring Workplan" details the provisions for monitoring and reporting progress on the objective and effectiveness (based on evaluation of quantifiable parameters and using principles of adaptive management) of the implemented conservation efforts. Specifically, the outcomes and the effectiveness of the Strategy are expected to be measured on a biennial basis. However, as noted below,

certainty that the above conservation efforts will remain in place after 5 years cannot be predicted at this time. As it stands, the Strategy, and conservation efforts therein, are only a force under Australian law if AFMA continues to implement the closures under section 41A of the Fisheries Management Act 1991. These closures are implemented under "Directions" (for example, the current fishery closures to protect Harrison's dogfish have been implemented under "SESSF Fishery Closures Direction No. 1 2013"). These legal instruments are only in effect for 5 years, after which AFMA may choose to extend the closures by creating a new Direction. If AFMA does not take action after 5 years, these closures will expire.

Although the Upper-Slope Dogfish Research and Monitoring Workplan details AFMA's commitment to stop the decline of Harrison's dogfish and work to rebuild the population, the protection of the species is not required under the EPBC Act since the species was listed as conservation dependent instead of endangered. In addition, in the case where any part of this Strategy ceases to exist or changes, the species would not automatically be listed as endangered under the EPBC Act. Rather, the TSSC would be convened and asked to evaluate how the changes impact the status of the species and provide recommendations on listing eligibility to the Minister for the Environment, with the ultimate decision on whether to list the species in a given category made by the Minister.

While we conclude that the present conservation efforts are currently effective in preventing the extinction of the species, we have no certainty that they will remain in place after 5 years. Taking into account the present state and life history of the species, we do not consider 5 years to be sufficient time for the status of the species to improve to where it is no longer in danger of extinction without the continued implementation of these efforts. In other words, the removal of these conservation efforts after 5 years will once again subject the species to the threats described previously, and based on the information from the extinction risk analysis (e.g., substantial depletion, fragmented populations, extremely low productivity, sensitivity to low levels of mortality), we find that the species will likely become in danger of extinction at that time.

In conclusion, after consideration of the evaluation criteria under the PECE, we are sufficiently certain that the implemented conservation efforts will effectively decrease the threat of overutilization by fisheries in the near

term to the point where the species is no longer presently in danger of extinction. However, given that the implementation of these conservation efforts is only certain for 5 years, a time frame that is insufficient to increase the species' chances of survival when faced again with prior threats, we conclude that the species will likely be in danger of extinction in the foreseeable future. We specifically seek additional information from the public comment process on these conservation efforts and their certainty of implementation and effectiveness (see below).

Proposed Determination

We assessed the ESA section 4(a)(1) factors and conclude that the species faces ongoing threats from overutilization, with the species' natural biological vulnerability to overexploitation exacerbating the severity of the threats. The species faces demographic risks, such as small and fragmented populations with low productivity, which make it likely to be influenced by stochastic or compensatory processes throughout its range and place the species in danger of extinction from the aforementioned threats. We deem ongoing conservation efforts as currently effective in decreasing the main threat of overutilization to the point where the species is no longer presently in danger of extinction. However, the time frame over which these conservation efforts will certainly be in place is insufficient to increase the species' chances of survival or prevent its extinction through the foreseeable future. Therefore, based on the best available scientific and commercial information as presented in the status report and this finding, we find that *C. harrissoni* is not currently in danger of extinction throughout its range, but is likely to become so in the foreseeable future. We propose to list Harrison's dogfish as a threatened species under the ESA.

Corals

The three coral species considered herein are all marine invertebrates in the phylum Cnidaria. The phylum is called Cnidaria because member species use cnidae (capsules containing stinging nematocysts) for prey capture and defense. All are tropical, shallow water, scleractinian ("stony") corals that secrete a calcium carbonate skeleton. Two of the three have the typical stony coral symbiosis with zooxanthellae (photosynthetic) algae that reside in gastrodermal cells of the coral tissue. All are non-reef building corals that live in small colonies or as solitary individuals. The following section

describes our analysis of the status of the three species. Information on many of the species is sparse, so we cannot provide complete descriptions of their natural history. More details can be found in Meadows (2014).

Species Description of *Cantharellus noumeae*

Cantharellus noumeae is a fungiid or mushroom coral that was the first described species of its genus, in 1984 (Hoeksema and Best, 1984). It received its own new genus name because, unlike most other fungiid corals, it is stalked and not free-living as an adult. Other species in the genus have since been discovered and named, so the genus is no longer monotypic. Polyps are relatively small for a fungiid coral, ranging from 25 to 65 mm in diameter (Hoeksema and Best, 1984). The polyps are cup-shaped when fully developed and have wavy margins (AIMS, 2013a). The primary septa are thin. The species may be solitary or colonial; colonies consist of a few contorted polyps. Their typical color is mottled brown.

Cantharellus noumeae was thought to occur only in a restricted area of less than 225 km² on reefs in sheltered bays in New Caledonia, on the southern tip of the main island of Grand Terre (Hoeksema *et al.*, 2008). Recent research by the French Institut de Recherche pour le Développement (IRD) has found that the species also occurs on fringing reefs farther up the southeast coast at Noumea and at Balabaio in the northeastern part of New Caledonia (www.lagplon.ird.nc; Antoine Gilbert, Ginger Soproner, personal communication, 2013). It is found in waters 10 to 35 m deep, close to soft sediment habitats that are in sheltered bays and lagoons (Hoeksema and Best, 1984). There are records of it in western, northern, and eastern parts of the island of New Guinea that includes Papua New Guinea and West Papua, Indonesia, with details likely to be published soon on a new Web site (<http://coraloftheworld.com>; Charlie Veron, personal communication). There are also reports of it from Papua New Guinea in the International Union for Conservation of Nature (IUCN) assessment, but the assessment questions the validity of this record (Hoeksema *et al.*, 2008). The IUCN assessment and the researcher whose published record is in question (Doug Fenner) suggest further confirmation is necessary (Hoeksema *et al.*, 2008; Fenner, personal communication). Fossil records from over 5 million years ago indicate that this species was at one time found as far west as East Kalimantan, on the island of Borneo,

Indonesia (Hoeksema, 1989; Hoeksema, 1993).

Scleractinian corals have diverse reproductive strategies, including both asexual and sexual modes of reproduction (see Brainard *et al.*, 2011). Individual reproductive modes for these three species have not been studied. *Cantharellus noumeae* may be a sequential sex-changing species like other members of its family. Because of their relationship with symbiotic zooxanthellae, *C. noumeae* needs to live in shallow water to be exposed to light the symbiotic algae use to photosynthetically fix carbon.

There is no quantitative species-specific population or trend information available for *C. noumeae* (Hoeksema *et al.*, 2008; Gilbert, personal communication). The current and continuing presence of the species in New Caledonia was confirmed by Bert Hoeksema (personal communication) in 2012 and in one murky location in Prony Bay on the southern tip of Grand Terre in 2013 (Andrew Bruckner, personal communication). In addition, Antoine Gilbert (personal communication) notes that from surveys he has done over the past 4 years, the species is “uncommon and usually found in fringing reefs where sedimentation is quite intense.” He also noted that the species is “usually found in low density, [but] it was observed in relative[ly] high density on the slope of artificial shores (embankment) in the biggest (commercial and industrial) harbour of New Caledonia: la Grande Rade.” We found no information on abundance or trends on New Guinea. Its presence at one site in Milne Bay (Fenner, 2003) is uncertain; Charlie Veron may publish information from New Guinea on his Web site soon (see above).

Species Description of Siderastrea glynni

Siderastrea glynni was described in 1994 (Budd and Guzmán, 1994). It occurs in non-reef-forming spherical colonies that are 70 to 100 mm in diameter (AIMS, 2013b). They have polygonal corallites that are 2.5 to 3.5 mm in diameter (Budd and Guzmán, 1994). The species is a light reddish-brown in color and occurs on coarse sand-rubble substrates. Recent genetic work by Forsman *et al.* (2005) has shown that *S. glynni* is genetically very similar to the Caribbean species *S. siderea*, though there are differences between the species. Their study could not differentiate between two possible explanations of the species' evolution: (1) that *S. siderea* and *S. glynni* are the same species and that *S. glynni* may

have recently passed through or been carried across the Panama Canal to the Pacific Ocean side; or (2) the alternate possibility that *S. glynni* evolved from *S. siderea*, likely about 2 to 2.3 million years ago during a period of high sea level, when the Isthmus of Panama may have been breached, allowing inter-basin transfer of the species' ancestors. Because the available information to reclassify the species is inconclusive, we determine that *S. glynni* is a valid and unique species.

The range of *S. glynni* is a small area of the Pacific Ocean near the small island of Uraba in Panama Bay, a few kilometers from the opening of the Panama Canal (Guzmán and Edgar, 2008). Identified colonies of *S. glynni* were reported to be unattached and occur “along the upper sand-coral rubble reef slope at a depth of 7 to 8.5 meters” (Budd and Guzmán, 1994). All the islands around the site, as well as another set of islands to the south, were searched several times without finding any additional colonies (Fenner, 2001).

The reproductive mode for this species has also not been studied. Because of their relationship with symbiotic zooxanthellae, *S. glynni* need to live in shallow water to be exposed to light the symbiotic algae use to photosynthetically fix carbon.

Only five colonies of *S. glynni* have ever been found. All were found by Budd and Guzmán (1994) when they discovered the species in 1992. All five colonies occurred within a small area of less than 10 m², with each colony within 1 m of another (Budd and Guzmán, 1994). Each colony was no more than 20 cm² in size. One colony was sacrificed in order to provide material for the species' description. During the 1997–98 El Niño event, the four surviving colonies started to deteriorate, displaying signs of bleaching and tissue loss. Due to their unhealthy state, the four colonies were moved to Smithsonian Tropical Research Institute (STRI) aquaria in Panama City, Panama, where they remain to this day (Guzmán and Edgar, 2008; Hector Guzmán, STRI, personal communication, 2013). According to Guzmán (personal communication, 2013) the colonies were fragmented to increase the number of specimens, but their growth rate has been very slow, and some fragments did not survive. From the original colonies, only one survives, with less than 4 cm² of living tissue. Nine of the fragmented colonies also survive in the lab and all are less than 9 cm² in area (Guzmán, personal communication, 2013). No known colonies exist in the wild; however, there is a possibility that it still exists

elsewhere in the wild and is yet undiscovered (Guzmán and Edgar, 2008). There are no plans to re-introduce the species, as existing colonies are too small to survive, though three of the fragments are being considered for cryopreservation, which would further reduce the population size (Guzmán, personal communication, 2013).

Species Description of Tubastraea floreana

Tubastraea floreana was first described by Wells (1982). It is an azooxanthellate species, which means it lacks the symbiotic photosynthetic zooxanthellae that most scleractinians have. It has a bright pink color while alive, but turns deep red-black when dead out of water. Corallites in the species are closely spaced (Cairns, 1991) and about 4–6 mm in size (Wells, 1983).

Tubastraea floreana is endemic to a few sites on a number of islands in the Galapagos Islands chain. It is mostly found in cryptic habitats, including on the ceilings of caves, and on ledges and rock overhangs (Hickman *et al.*, 2007). It has been reported to occur at depths of 2 to 46 m (Hickman *et al.*, 2007).

The reproductive mode of this species has not been studied, but other *Tubastraea* species reproduce asexually. Other *Tubastraea* species are invasive and productive (Riul *et al.*, 2013), so *T. floreana* is also likely to be moderately productive.

According to Hickman *et al.* (2007), prior to the 1982–83 El Niño Southern Oscillation (ENSO) this species was known from six sites on four islands in the Galapagos. Since the 1982–83 ENSO, specimens have only been observed at two sites. At one of these two sites, the species has not been seen since 2001, leaving only a single confirmed site with living specimens (Hickman *et al.*, 2007). Recent reports indicate the species is still present in at least one site (Stuart Banks, Charles Darwin Foundation, personal communication, 2013). We know of no other published information on distribution or abundance for this species.

Summary of Factors Affecting the Three Species of Coral

Next we consider whether any one or a combination of the threat factors specified in section 4(a)(1) of the ESA are contributing to the extinction risk of these three corals. Available information does not indicate that overutilization is an operative threat for these species; therefore, we do not discuss this factor further here. We discuss each of the remaining four factors and their

interaction in turn below, with species-specific information following a general discussion relevant to all of the species. A full review of all of the ESA section 4(a)(1) threat factors can be found in Meadows (2014b) and our final rule listing 20 corals (20-coral listing rule) under the ESA (79 FR 53851; September 10, 2014), which provides a general global summary of threats facing corals. Our 20-coral listing rule identified ocean warming, ocean acidification, sea-level rise, disease, sedimentation, nutrient enrichment, and fishing as the major global threats to coral reefs. The information about these threats and the species' responses to these threats is described in the 20-coral listing rule and incorporated herein by reference. Species-specific information regarding applicability of these threats to the three species considered here is discussed below, where available. The extent to which the risks discussed in the 20-coral listing rule are similar to the risks to these three corals is discussed for each species.

The Present or Threatened Destruction, Modification, or Curtailment of Its Habitat or Range

Habitat modification from climate change is a potential threat to all three species of corals (79 FR 53851; September 10, 2014). Coral bleaching occurs when the photosynthetic zooxanthellae symbionts of corals are damaged by light at higher than normal temperatures. The resulting damage leads to the expulsion of these important organisms from the coral host, depriving the host of the nutrients and energy provided by the zooxanthellae. While corals can survive mild to moderate bleaching, repeated, severe, or prolonged bleaching can lead to colony mortality. Bleaching events have been increasing both in intensity and geographic extent due to worldwide anthropogenic climate change (Hoegh-Guldberg, 2006; Eakin *et al.*, 2009). Certain genera and growth forms, particularly branched species, are more sensitive to bleaching than others (Wooldridge, 2013). Many corals are physiologically optimized to their local long-term seasonal variations in temperatures and an increase of only 1–2 °C above the normal local seasonal maximum can induce bleaching (Brainard *et al.*, 2011; Logan *et al.*, 2013). The United States NOAA Coral Reef Watch satellite bleaching database shows that the range of all three species occurs in areas that frequently have bleaching alerts, with alerts being more frequent and severe in the ranges of *S. glynni* and *T. floreana*, than in the range of *C. noumeae*.

Ocean acidification threatens to slow or halt coral growth and reef building entirely if the pH of the ocean becomes too low for corals to form their calcite skeletons, but tolerance appears to vary by species for those that have been studied (see Brainard *et al.*, 2011). In addition, bioerosion of reefs is likely to accelerate as coral skeletons become more fragile as a result of the effects of acidification, but effects are highly species-specific. Since the petitioned species are not reef-building, this effect is likely to be less significant.

Sea-level is also likely to rise as a result of climate change, but effects on corals are highly uncertain, owing to uncertainty in both the likely rate and extent of sea-level rise as well as the ability of corals generally (or the petitioned species specifically) to keep pace with the rise in sea level (Brainard *et al.*, 2011; 79 FR 53851; September 10, 2014).

While climate change effects are likely to be serious for many corals, Brainard *et al.* (2011) and our final rule listing 20 corals under the ESA (79 FR 53851; September 10, 2014) show that adaptation and acclimatization of corals to increased ocean temperatures are possible, that there is intra-genus and inter-species variation in susceptibility to bleaching, ocean acidification, and sedimentation, that at least some species have already expanded their range in response to climate change, and that not all species are seriously affected by ocean acidification. In addition, a more recent paper by Logan *et al.* (2013) examined the potential for coral adaptation and acclimatization to climate change and found that these processes can reduce the frequency of mass bleaching events in the future. Their modeling results suggest some adaptation or acclimatization may even have already occurred. A study by Wooldridge (2014) provides support that a suite of morphological and physiological traits relate to bleaching vulnerability. These include symbionts' type, metabolic rate, colony tissue thickness, skeletal growth form, mucus production rates, fluorescent pigment concentrations, and heterotrophic feeding capacity. According to Wooldridge (2014), these traits tend to correlate with the ends of the dichotomy of branching and plate corals with thin tissue layers versus massive and encrusting corals with thick tissue layers. The species under consideration here are not necessarily the most vulnerable, based on those traits (see below). Therefore, while climate change is generally considered a potential threat to these candidate corals, the likelihood and magnitude of threats

from climate change are largely species-specific and must be examined on that basis to fully assess extinction risk (79 FR 53851; September 10, 2014).

In addition to the general global threats identified in our status review of 82 corals and final rule listing 20 corals under the ESA (Brainard *et al.*, 2011; 79 FR 53851; September 10, 2014), there are some species-specific threats for which we have detailed information at the scale of these species' ranges that are discussed below.

Cantharellus noumeae

Cantharellus noumeae is exposed to deforestation, urbanization, and mining activity that causes sedimentation and water pollution throughout its range in New Caledonia (Hoeksema *et al.*, 2008; David *et al.*, 2010; McKenna *et al.*, 2011). The mining activity is a result of nickel and smaller amounts of other metal mining (cobalt and chromium especially) on land throughout the main island of Grand Terre (McKenna *et al.*, 2011; Hoeksema, personal communication). Grand Terre holds 25 percent of the world's known nickel deposits (McKenna *et al.*, 2011). Nickel mining started there in the 1870s. Currently, most mining is done by open-cast strip mining, which has caused deforestation and increased erosion and runoff of sediments leading to varying degrees of sedimentation and light attenuation throughout the lagoon of Grand Terre, including in areas in and adjacent to the species' range (Ouillon *et al.*, 2010). Labrosse *et al.* (2000) estimate that 300 million m³ of soil has been displaced since the beginning of mining activities. Mines are located across the country, including the large new Goro complex, which includes mines, processing facilities, and a port. The complex began production in late 2010 and is very near the most abundant population of *C. noumeae*. The Goro complex has already had three incidents affecting the environment, involving spills or releases of sulfuric acid solutions used in the processing of the nickel ore (Sulfuric Acid on the Web, 2013). Runoff of heavy metals from the mining operations has greatly increased concentrations of those metals in the marine environment (Fichez *et al.*, 2010). Nickel has been shown to affect fertilization success of four reef coral species in the families Acroporidae and Faviidae (Reichelt-Brushett and Harrison, 2005) and to affect settlement and cause mortality of larvae in the coral *Pocillopora damicornis* (Goh, 1991). Gilbert (personal communication, 2013) reports that the species is common in areas of high sedimentation and in the largest harbor, so it may be

tolerant to environmental stressors like sedimentation. The species may have the ability to actively remove sediments, as has been shown in some other fungiid corals (Bongaerts *et al.*, 2012), but this is uncertain. Mitigation measures for mining operations are required by legislation and include reef monitoring requirements (UNESCO, 2011; Gilbert, personal communication, 2013), but this monitoring is not at the species level (Gilbert, personal communication, 2013). It is unclear how effective the mitigation methods are, as sedimentation and pollution remain concerns (David *et al.*, 2010).

Despite the frequency of bleaching alerts, heat-related bleaching is apparently not a significant current threat in the range of *C. noumeae* in New Caledonia, as water temperatures there are relatively low (Hoeksema, Naturalis Biodiversity Center, personal communication, 2013) and the ReefBase coral bleaching database only reports events with low bleaching severity as the worst past events to ever occur there. We have found no species-specific information on the susceptibility of this species to bleaching or ocean acidification; however, its growth form suggests it is not among the most susceptible species (Wooldridge, 2014).

Anthropogenic eutrophication occurs in the range of the species near the capital of Noumea and is attributed mostly to inadequately treated sewage (Fichez *et al.*, 2010), although 19 aquaculture farms on the west coast and island-wide agriculture may also play roles (David *et al.*, 2010). Storm events and flooding have also recently occurred in the range of the species (EMR, 2013), and there is concern that climate change may make such events more frequent in New Caledonia (Gilbert, personal communication, 2013).

The biggest threats to New Guinea's coral reef resources include sedimentation and pollution from inland sources (*e.g.*, forest clearance, sewage, and erosion), climate change, and dynamite fishing (Burke *et al.*, 2011; PNG, 2009; PNG, 2012). There is little specific data on these threats in New Guinea in the above references.

Siderastrea glynni

Should *S. glynni* ever be restored to the wild, it faces considerable habitat degradation threats from coastal development, oil production, sedimentation, eutrophication and other pollution, and increased transportation activities in the Panama City area, the Gulf of Panama, and the enlarged Panama Canal, which is due to open in

2016 (Mate, 2003; Guzmán and Edgar, 2008). Almost continuous dredging and release of oil-based compounds (bunker oil, diesel, gasoline, etc.) that are spilled from nearby port facilities and commercial vessels anchored near the species' natural range are other reasons why it was decided to transfer and then keep in captivity the remaining known colonies (Guzman, personal communication, 2013). "During the 1997–98 ENSO event, the four known colonies of *S. glynni* began to deteriorate, displaying bleaching and tissue loss" (Guzmán and Edgar, 2008). This suggests this species is vulnerable to increased ocean temperatures, though there is no specific research on this point. As discussed above, the area of the species' range is subject to a high frequency of bleaching warnings. We have found no species-specific information on the susceptibility of this species to ocean acidification.

Tubastraea floreana

For *T. floreana*, there is a lack of information on thermal tolerances, but "the dramatic reduction in its distribution immediately after the 1982–83 [ENSO] event suggests that this mortality resulted from the event" (Hickman *et al.*, 2007). This is true despite the fact that this species is azooxanthellate, suggesting that other mechanisms besides loss of calorie subsidy from symbionts are involved. Edgar *et al.* (2010) document a series of drastic ecosystem changes in the Galapagos following the 1982–83 ENSO event, including dramatic declines in dissolved nutrients and phytoplankton productivity, leading to declines across the food chain and resulting in heavily grazed reefs with crustose coralline algae ("urchin barrens") replacing former macroalgal and coral habitats. A total of 95–99 percent of reef coral cover was lost from the Galapagos between 1983 and 1985 (Edgar *et al.*, 2010). All known coral reefs based on calcareous frameworks died and subsequently disintegrated to rubble and sand (Glynn, 1994). These changes led to large decreases in biodiversity. The urchin *Eucidaris galapagensis* now appears to be present in sufficient numbers to prevent re-establishment of coral and macroalgal habitat, thereby facilitating a regime shift in local benthic habitats (Edgar *et al.*, 2010). Moreover, the Galapagos Islands sit near the center of the most intense El Niño events in the region (Glynn and Ault, 2000) and are regularly included in bleaching threat warnings issued by NOAA (see above). Therefore, future ENSO events and inhibition of recruitment are likely to remain threats to *T. floreana*. We have

found no species-specific information on the susceptibility of this species to ocean acidification.

Disease and Predation

Coral disease has been linked to the effects of climate change (see Brainard *et al.*, 2011), especially indirectly as a synergistic effect, as climate change and other threats potentially increase stress on corals, making them more susceptible to disease. Coral diseases also appear to be increasing worldwide (Roessig *et al.*, 2004). Nevertheless, susceptibility of coral species to disease is highly species-specific and no generalizations can be made. We found no species-specific information on disease in *C. noumeae* or *T. floreana*. Black-band, dark spot, and white plague diseases in the Caribbean occur in *S. siderea*, which is closely related to *S. glynni* (Sekar *et al.*, 2008; Brandt and McManus, 2009; Cardenas *et al.*, 2012), suggesting *S. glynni* may be susceptible to similar coral diseases, but we have no solid information.

With respect to predation, we found no information on predation threats to *S. glynni* or *T. floreana*. For *C. noumeae*, one potential predation threat is *Acanthaster planci* (crown-of-thorns starfish). *Acanthaster planci* does not appear to be a major cause of coral mortality in New Caledonia (Adjeroud, 2012), but several remote reefs surveyed during the Global Reef Expedition in November 2013 on the outer-slope of Guilbert's atolls showed evidence of past outbreaks (LOF, 2013).

Inadequacy of Existing Regulatory Mechanisms

The petitioners discussed regulation of trade in corals under CITES as a threat to these species. All of the species considered in this petition were listed in Appendix II of CITES in 1989, when all scleractinian corals were listed. While only some scleractinians were in trade at the time, the 1989 listing rationale for including all scleractinians in Appendix II was because of identification difficulties where non-traded species resemble species in trade. According to Article II of CITES, species listed on Appendix II are those that are "not necessarily now threatened with extinction but may become so unless trade in specimens of such species is subject to strict regulation in order to avoid utilization incompatible with their survival." Based on the CITES definitions and standards for listing species on Appendix II, the species' listing on Appendix II is not itself an inherent indication that these species may now warrant threatened or endangered status under the ESA. The

significance of any threat from international trade would depend on the amount of international trade relative to the population size of the species, as well as any other factors related to the trade, such as habitat damage caused in the collecting process, or synergistic effects of other threats. We have no information any of these three species is traded internationally.

Because each of the species considered herein exists in small ranges that do not overlap with each other, and they are not otherwise managed or regulated under any other common international regimes, additional discussion of this factor is left for the species-specific entries for this section, below.

Cantharellus noumeae

Since the Organic Law (No. 99–209) on March 19, 1999, New Caledonia has been recognized as an “Overseas Country” of France. This status gives New Caledonia extensive autonomy with respect to France. In particular, the national laws in force within France are no longer applicable to New Caledonia, and New Caledonia now manages the ocean resources of its Exclusive Economic Zone. The territorial sea and the maritime public domain (coastal terrestrial and nearshore aquatic zone originating under French colonial law) depend on management from New Caledonia’s three provinces (David *et al.*, 2010). In the two provinces where *C. noumeae* occurs, collection of live corals (and other marine resources) is restricted to scientists and licensed fishers who can only collect for a domestic market.

The range of *C. noumeae* is included in the United Nations Education, Scientific and Cultural Organization (UNESCO) World Heritage Site designation for the “Lagoons of New Caledonia” site, specifically within the South Grand Lagoon area. The World Heritage Site implementation is supported by specific legislation on fisheries, land and water use planning, urban development, and mining (Morris and Mackay, 2008). A wide monitoring program of the heritage site all around New Caledonia was created (Andréfouët 2008), but this suffers from a lack of sampling at a species level (Gilbert, personal communication, 2013). In 2011, the World Heritage Committee of UNESCO (the organizing body for World Heritage Sites) issued Decision 35Com 7B.22, which expressed concern regarding permits granted to the mining company GEOVIC to explore for cobalt in mineral sands in areas adjacent to the site and near the range of *C. noumeae*. The committee requested that New

Caledonia submit Environmental Impact Assessments for the proposed exploration and possible exploitation of cobalt sands to the World Heritage Centre. We have no evidence this has occurred. The New Caledonian Mining Code prescribes mitigation measures to mitigate the impacts of mining activities (see above), and abandoned mines are being restored using indigenous plant species (UNESCO, 2011).

In Papua New Guinea, there is a variety of legislation to protect biodiversity and habitat, including a mandate to ensure marine resource sustainability, and a plan of action directed at coral reef conservation (PNG, 2009). However, as noted above, threats remain. Resources and capacity may not be adequate to ensure full implementation of the laws and plan (PNG, 2009; PNG, 2012).

Overall, we do not believe that the threat to *C. noumeae* from habitat modification, destruction, and pollution is adequately addressed or mitigated by existing regulatory mechanisms.

Siderastrea glynni

A national law in Panama prohibits coral extraction or mining (Guzmán, 2003), but enforcement is weak and the law may not fully protect rare species (Guzmán, personal communication, 2013). The range of *S. glynni* is adjacent to the Bay of Panama, which is designated an internationally important wetland under the Ramsar Convention and contains extensive mangrove beds that are critical nursery grounds for many marine species. The Bay is a protected Wildlife Refuge under Panamanian law. However, developers seek to open the area for tourism, and Panamanian authorities have requested a reduction of the Ramsar area of the bay (AIDA, 2013). We were not able to find any other species-specific information on this threat. Based on the available information, it is not clear that existing regulatory mechanisms would be adequate to protect *S. glynni*, should it be reintroduced into the wild or found in additional locations.

Tubastraea floreana

The Galápagos Marine Reserve was established in 1986 and expanded to its current size around all the islands in 1998. The reserve has a zoning plan with both limited and multiple use zones. Rules prohibit removing or disturbing any plant, animal, or remains of such, or other natural objects.

Tubastraea floreana also occurs inside the Galapagos Island World Heritage Site (expanded to include Galapagos Marine Reserve areas in 2001) and the Galápagos Island Man and Biosphere

Reserve (1984), both designations of UNESCO. The area was also designated a Galápagos Archipelago Particularly Sensitive Area in 2005. This is a designation by the International Maritime Organization (IMO) that recognizes the area as having ecological, socio-economic, or scientific attributes that make the area vulnerable to damage by international shipping activities. Based on this designation, the IMO instituted special navigation rules in the area. In addition, Ecuador’s “Ley de Gestión Ambiental” (Law of Environmental Management) establishes principles and directives for environmental management, land-use planning, zoning, sustainable use, and natural heritage conservation. Ecuador’s fisheries law states that no harm may be caused to areas that are declared protected, with corals included under those protections (MCA Toolkit, 2013). While the above laws and protected area designations provide a great deal of protection for resources in the area in principal, in practice, illegal activities and incomplete and difficult enforcement, as discussed in the status review report (Meadows, 2014), could threaten *T. floreana*. Moreover, the threats from climate change and ENSO events are outside the scope of these protections.

Other Natural or Manmade Factors Affecting Their Continued Existence

The range of *C. noumeae* in New Caledonia is exposed to eight tropical storms per year on average (David *et al.*, 2010). Specific effects of storms on this species are not documented, but the petitioner submitted an undated Web page that claims Cyclone Erica destroyed between 10 and 80 percent of live coral in New Caledonia in 2003 (EDGE, Undated; Guillemot *et al.*, 2010). We were not able to find any other species-specific information available regarding this threat category for *C. noumeae*. Based on this information, we consider tropical storms an additional potential natural threat to the species, for which we seek additional information (see below).

For *S. glynni* and *T. floreana*, both species have such a small number of colonies, they are susceptible to all of the problems of species with low genetic diversity and population size, including inbreeding depression, population bottlenecks, Allee effects, and density-independent mortality, among others.

Extinction Risk

The extinction risk analyses of Meadows (2014) found all three species to be at either a moderately high or high

risk of extinction. The extinction risk for *C. noumeae* was found to be moderately high, based on the species' small, restricted range, likely low growth rate and genetic diversity, and potential threats from development, water pollution, possibly sedimentation at some level, and potential illegal activities, mitigated by consideration of potential resilience to sedimentation threats and uncertainty regarding sensitivity to heavy metals. Based on the current information, this is the case whether or not the species' range includes New Guinea. The extinction risk for *S. glynni* was found to be high, due to the lack of known populations in the wild, a small captive population in a single location, likely low growth rates and genetic diversity, and potential increased threats from El Niño, climate change, disease, and other development and habitat degradation, should the species be reintroduced to Panama. The extinction risk for *T. floreana* was found to be high, based on its small, restricted range, documented declines, likely low levels of genetic diversity, and threats from El Niño, climate change, development, and illegal activities, mitigated by potential for moderate productivity.

After reviewing the best available scientific data and the extinction risk evaluations of the three species of coral, we concur with Meadows (2014) and conclude that the risk of extinction for all three species is currently high.

Protective Efforts

We evaluated conservation efforts we are aware of to protect and recover coral that are either underway but not yet shown to be effective, or are only planned. We were not able to find any information on conservation efforts specific to *C. noumeae* or *T. floreana*, or their habitat, that are not yet implemented or shown to be effective and that would potentially alter the extinction risk for the species. For *S. glynni*, we are aware that Dr. Hector Guzmán, who maintains the only surviving colonies of this species in captivity at the STRI laboratories, is planning to cryopreserve some specimens to provide an additional means to recover the species and preserve its genetic information. The certainty that this effort will be implemented is unclear. Further, the effectiveness of a cryopreservation effort for species recovery is largely unknown, and thus it is impossible to determine whether this effort will be effective in conserving or improving the status of this species. We are thus not able to conclude that any current conservation efforts would alter the extinction risk for

any of these three species. We seek additional information on other conservation efforts in our public comment process (see below).

Proposed Determination

Based on the best available scientific and commercial information as presented in the status report and this finding, we find that all three species of coral are in danger of extinction throughout all of their ranges. We assessed the ESA section 4(a)(1) factors and conclude that *Cantharellus noumeae*, *Siderastrea glynni*, and *Tubastraea floreana* all face ongoing threats from habitat alteration, small ranges and/or population sizes, and the inadequacy of existing regulatory mechanisms throughout their ranges. *C. noumeae* also faces risks from pollution and *S. glynni* may be at risk from disease. We therefore propose to list all three species as endangered.

Effects of Listing

Conservation measures provided for species listed as endangered or threatened under the ESA include recovery actions (16 U.S.C. 1533(f)); concurrent designation of critical habitat, if prudent and determinable (16 U.S.C. 1533(a)(3)(A)); Federal agency requirements to consult with NMFS under section 7 of the ESA to ensure their actions do not jeopardize the species or result in adverse modification or destruction of critical habitat should it be designated (16 U.S.C. 1536); and prohibitions on taking (16 U.S.C. 1538). Recognition of the species' plight through listing promotes conservation actions by Federal and state agencies, foreign entities, private groups, and individuals. The main effects of the proposed endangered listings are prohibitions on take, including export and import.

Identifying Section 7 Conference and Consultation Requirements

Section 7(a)(2) (16 U.S.C. 1536(a)(2)) of the ESA and NMFS/USFWS regulations require Federal agencies to consult with us to ensure that activities they authorize, fund, or carry out are not likely to jeopardize the continued existence of listed species or destroy or adversely modify critical habitat. Section 7(a)(4) (16 U.S.C. 1536(a)(4)) of the ESA and NMFS/USFWS regulations also require Federal agencies to confer with us on actions likely to jeopardize the continued existence of species proposed for listing, or that result in the destruction or adverse modification of proposed critical habitat of those species. It is unlikely that the listing of these species under the ESA will

increase the number of section 7 consultations, because these species occur outside of the United States and are unlikely to be affected by Federal actions.

Critical Habitat

Critical habitat is defined in section 3 of the ESA (16 U.S.C. 1532(5)) as: (1) The specific areas within the geographical area occupied by a species, at the time it is listed in accordance with the ESA, on which are found those physical or biological features (a) essential to the conservation of the species and (b) that may require special management considerations or protection; and (2) specific areas outside the geographical area occupied by a species at the time it is listed upon a determination that such areas are essential for the conservation of the species. "Conservation" means the use of all methods and procedures needed to bring the species to the point at which listing under the ESA is no longer necessary. Section 4(a)(3)(A) of the ESA (16 U.S.C. 1533(a)(3)(A)) requires that, to the extent prudent and determinable, critical habitat be designated concurrently with the listing of a species. However, critical habitat shall not be designated in foreign countries or other areas outside U.S. jurisdiction (50 CFR 424.12 (h)).

The best available scientific and commercial data as discussed above identify the geographical areas occupied by *Aipysurus fuscus*, *Cantharellus noumeae*, *Centrophorus harrissoni*, *Pterapogon kauderni*, *Siderastrea glynni*, and *Tubastraea floreana* as being entirely outside U.S. jurisdiction, so we cannot designate critical habitat for these species.

We can designate critical habitat in areas in the United States currently unoccupied by the species, if the area(s) are determined by the Secretary to be essential for the conservation of the species. Regulations at 50 CFR 424.12(e) specify that we shall designate as critical habitat areas outside the geographical range presently occupied by the species only when the designation limited to its present range would be inadequate to ensure the conservation of the species. The best available scientific and commercial information on these species does not indicate that U.S. waters provide any specific essential biological function for any of the species proposed for listing. Based on the best available information, we have not identified unoccupied area(s) in U.S. water that are currently essential to the conservation of any of the corals proposed for listing. Therefore, based on the available

information, we do not intend to designate critical habitat for *Aipysurus fuscus*, *Cantharellus noumeae*, *Centrophorus harrissoni*, *Pterapogon kauderni*, *Siderastrea glynni*, and *Tubastraea floreana*.

Identification of Those Activities That Would Constitute a Violation of Section 9 of the ESA

On July 1, 1994, NMFS and FWS published a policy (59 FR 34272) that requires us to identify, to the maximum extent practicable at the time a species is listed, those activities that would or would not constitute a violation of section 9 of the ESA.

Because we are proposing to list all three corals and the dusky sea snake as endangered, all of the prohibitions of section 9(a)(1) of the ESA will apply to these species. These include prohibitions against the import, export, use in foreign commerce, or “take” of the species. These prohibitions apply to all persons subject to the jurisdiction of the United States, including in the United States, its territorial sea, or on the high seas. Take is defined as “to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct.” The intent of this policy is to increase public awareness of the effects of this listing on proposed and ongoing activities within the species’ range. Activities that we believe could result in a violation of section 9 prohibitions for these species include, but are not limited to, the following:

(1) Possessing, delivering, transporting, or shipping any individual or part (dead or alive) taken in violation of section 9(a)(1);

(2) Delivering, receiving, carrying, transporting, or shipping in interstate or foreign commerce any individual or part, in the course of a commercial activity;

(3) Selling or offering for sale in interstate commerce any part, except antique articles at least 100 years old;

(4) Importing or exporting;

(5) Releasing captive animals into the wild without a permit issued under section 10(a)(1)(A). Although animals held non-commercially in captivity at the time of listing are exempt from the prohibitions of import and export, the individual animals are considered listed and afforded most of the protections of the ESA, including most importantly, the prohibition against injuring or killing. Release of a captive animal has the potential to injure or kill the animal. Of an even greater conservation concern, the release of a captive animal has the potential to affect wild populations through introduction of

diseases or inappropriate genetic mixing;

(6) Harming captive animals by, among other things, injuring or killing a captive animal, through experimental or potentially injurious care or conducting research or sexual breeding activities on captive animals, outside the bounds of normal animal husbandry practices. Captive sexual breeding of corals is considered potentially injurious. Furthermore, the production of coral progeny has conservation implications (both positive and negative) for wild populations. Experimental or potentially injurious care or procedures and research or sexual breeding activities of corals or dusky sea snakes may, depending on the circumstances, be authorized under an ESA 10(a)(1)(A) permit for scientific research or the enhancement of the propagation or survival of the species.

Identification of Those Activities That Would Not Constitute a Violation of Section 9 of the ESA

We will identify, to the extent known at the time of the final rule, specific activities that will not be considered likely to result in a violation of section 9 of the ESA. Although not binding, we are considering the following actions, depending on the circumstances, as not being prohibited by ESA section 9:

(1) Take authorized by, and carried out in accordance with the terms and conditions of, an ESA section 10(a)(1)(A) permit issued by NMFS for purposes of scientific research or the enhancement of the propagation or survival of the species;

(2) Continued possession of parts that were in possession at the time of listing. Such parts may be non-commercially exported or imported; however the importer or exporter must be able to provide evidence to show that the parts meet the criteria of ESA section 9(b)(1) (*i.e.*, held in a controlled environment at the time of listing, in a non-commercial activity);

(3) Continued possession of live corals or sea snakes that were in captivity or in a controlled environment (*e.g.*, in aquaria) at the time of this listing, so long as the prohibitions under ESA section 9(a)(1) are not violated. Facilities must provide evidence that the animals were in captivity or in a controlled environment prior to listing. We suggest such facilities submit information to us on the animals in their possession (*e.g.*, size, age, description of animals, and the source and date of acquisition) to establish their claim of possession (see **FOR FURTHER INFORMATION CONTACT**);

(4) Provision of care for live corals or sea snakes that were in captivity at the time of listing. These individuals are still protected under the ESA and may not be killed or injured, or otherwise harmed, and, therefore, must receive proper care. Normal care of captive animals necessarily entails handling or other manipulation of the animals, and we do not consider such activities to constitute take or harassment of the animals so long as adequate care, including veterinary care, when such practices, procedures, or provisions are not likely to result in injury, is provided; and

(5) Any interstate and foreign commerce trade of animals already in captivity. Section 11(f) of the ESA gives NMFS authority to promulgate regulations that may be appropriate to enforce the ESA. NMFS may promulgate future regulations to regulate trade or holding of these species (if any), if necessary. NMFS will provide the public with the opportunity to comment on future proposed regulations.

Protective Regulations Under Section 4(d) of the ESA

We are proposing to list *Pterapogon kauderni*, and *Centrophorus harrissoni* as threatened species. In the case of threatened species, ESA section 4(d) leaves it to the Secretary’s discretion whether, and to what extent, to extend the section 9(a) “take” prohibitions to the species, and authorizes us to issue regulations necessary and advisable for the conservation of the species. Thus, we have flexibility under section 4(d) to tailor protective regulations, taking into account the effectiveness of available conservation measures. The 4(d) protective regulations may prohibit, with respect to threatened species, some or all of the acts which section 9(a) of the ESA prohibits with respect to endangered species. These 9(a) prohibitions apply to all individuals, organizations, and agencies subject to U.S. jurisdiction. We will consider potential protective regulations pursuant to section 4(d) for the proposed threatened species. For example, we may consider future regulations on trade of wild-caught Banggai cardinalfish (see number 7 below). We seek public comment on potential 4(d) protective regulations (see below).

Public Comments Solicited

To ensure that any final action resulting from this proposed rule to list six species will be as accurate and effective as possible, we are soliciting comments and information from the public, other concerned governmental

agencies, the scientific community, industry, and any other interested parties on information in the status review and proposed rule. Comments are encouraged on these proposals (See **DATES** and **ADDRESSES**). We must base our final determination on the best available scientific and commercial information when making listing determinations. We cannot, for example, consider the economic effects of a listing determination. Final promulgation of any regulation(s) on these species' listing proposals will take into consideration the comments and any additional information we receive, and such communications may lead to a final regulation that differs from this proposal or result in a withdrawal of this listing proposal. We particularly seek:

(1) Information concerning the threats to any of the six species proposed for listing;

(2) Taxonomic information on any of these species;

(3) Biological information (life history, genetics, population connectivity, etc.) on any of these species;

(4) Efforts being made to protect any of these species throughout their current ranges;

(5) Information on the commercial trade of any of these species;

(6) Historical and current distribution and abundance and trends for any of these species; and

(7) Information relevant to potential ESA section 4(d) protective regulations for any of the proposed threatened species, especially the application, if any, of the ESA section 9 prohibitions on import, take, possession, receipt, and sale of the Banggai cardinalfish which is currently in international trade.

We request that all information be accompanied by: (1) Supporting documentation, such as maps, bibliographic references, or reprints of pertinent publications; and (2) the submitter's name, address, and any association, institution, or business that the person represents.

Role of Peer Review

In December 2004, the Office of Management and Budget (OMB) issued a Final Information Quality Bulletin for Peer Review establishing a minimum peer review standard. Similarly, a joint NMFS/FWS policy (59 FR 34270; July 1, 1994) requires us to solicit independent expert review from qualified specialists, concurrent with the public comment period. The intent of the peer review policy is to ensure that listings are based on the best scientific and commercial data available. We solicited peer review

comments on each of the status review reports, including from: four scientists with expertise on sea snakes or the dusky sea snake specifically, five familiar with the Banggai cardinalfish, five familiar with Harrison's dogfish, and ten scientists familiar with corals. For these species, we received comments from the scientists, and their comments are incorporated into the draft status review reports for each species and this 12-month finding.

Proposed Revisions to the NMFS Lists

We propose to revise and add table subheadings to accommodate the proposed listings in our lists of threatened and endangered species at 50 CFR 223.102 and 50 CFR 224.101, respectively. We propose to revise the subheading of "Sea Turtles" in both tables by changing the subheading to "Reptiles." This new subheading will encompass all currently listed sea turtles as well as other marine reptiles like the dusky sea snake. In addition, we propose to add the subheading "Corals" to our table at 50 CFR 224.101. This subheading has already been added to our table at 50 CFR 223.102 in a previous rulemaking (79 FR 20802; April 14, 2014). These revisions and additions are not substantive changes, but having these headings will help the public identify and locate species of interest in a more efficient manner.

References

A complete list of the references used in this proposed rule is available upon request (see **ADDRESSES**).

Classification

National Environmental Policy Act

The 1982 amendments to the ESA, in section 4(b)(1)(A), restrict the information that may be considered when assessing species for listing. Based on this limitation of criteria for a listing decision and the opinion in *Pacific Legal Foundation v. Andrus*, 675 F. 2d 825 (6th Cir. 1981), NMFS has concluded that ESA listing actions are not subject to the environmental assessment requirements of the National Environmental Policy Act (NEPA) (See NOAA Administrative Order 216-6).

Executive Order 12866, Regulatory Flexibility Act, and Paperwork Reduction Act

As noted in the Conference Report on the 1982 amendments to the ESA, economic impacts cannot be considered when assessing the status of a species. Therefore, the economic analysis requirements of the Regulatory Flexibility Act are not applicable to the listing process. In addition, this

proposed rule is exempt from review under Executive Order 12866. This proposed rule does not contain a collection-of-information requirement for the purposes of the Paperwork Reduction Act.

Executive Order 13132, Federalism

In accordance with E.O. 13132, we determined that this proposed rule does not have significant Federalism effects and that a Federalism assessment is not required. In keeping with the intent of the Administration and Congress to provide continuing and meaningful dialogue on issues of mutual state and Federal interest, this proposed rule will be given to the relevant governmental agencies in the countries in which the species occurs, and they will be invited to comment. We will confer with the U.S. Department of State to ensure appropriate notice is given to foreign nations within the range of all three species. As the process continues, we intend to continue engaging in informal and formal contacts with the U.S. State Department, giving careful consideration to all written and oral comments received.

List of Subjects in 50 CFR Parts 223 and 224

Administrative practice and procedure, Endangered and threatened species, Exports, Imports, Reporting and record keeping requirements, Transportation.

Dated: December 8, 2014.

Samuel D. Rauch, III.

Deputy Assistant Administrator for Regulatory Programs, National Marine Fisheries Service.

For the reasons set out in the preamble, 50 CFR parts 223 and 224 are proposed to be amended as follows:

PART 223—THREATENED MARINE AND ANADROMOUS SPECIES

■ 1. The authority citation for part 223 continues to read as follows:

Authority: 16 U.S.C. 1531–1543; subpart B, § 223.201–202 also issued under 16 U.S.C. 1361 *et seq.*; 16 U.S.C. 5503(d) for § 223.206(d)(9).

■ 2. In § 223.102, amend the table in paragraph (e) by:

■ A. Revising the table subheading of "Sea Turtles" to "Reptiles"; and

■ B. Adding new entries for two species in alphabetical order under the "Fishes" table subheading to read as follows:

§ 223.102 Enumeration of threatened marine and anadromous species.

* * * * *

(e) The threatened species under the jurisdiction of the Secretary of Commerce are:

Species ¹		Description of listed entity	Citation(s) for listing determination(s)	Critical habitat	ESA rules
Common name	Scientific name				
*	*	*	*	*	*
REPTILES ²					
*	*	*	*	*	*
FISHES					
Cardinalfish, Banggai	<i>Pterapogon kauderni</i>	Entire species	Insert Federal Register citation and date when published as a final rule].	NA	NA
*	*	*	*	*	*
Shark, Harrison's dogfish	<i>Centrophorus harrissoni</i> ...	Entire species	Insert Federal Register citation and date when published as a final rule].	NA	NA

¹ Species includes taxonomic species, subspecies, distinct population segments (DPSs) (for a policy statement, see 61 FR 4722, February 7, 1996), and evolutionarily significant units (ESUs) (for a policy statement, see 56 FR 58612, November 20, 1991).

² Jurisdiction for sea turtles by the Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, is limited to turtles while in the water.

PART 224—ENDANGERED MARINE AND ANADROMOUS SPECIES

■ 3. The authority citation for part 224 continues to read as follows:

Authority: 16 U.S.C. 1531–1543 and 16 U.S.C. 1361 *et seq.*

■ 4. In § 224.101, paragraph (h), amend the table by:

- A. Revising the table subheading of “Sea Turtles²” to “Reptiles²”;
- B. Adding an entry for the dusky sea snake in alphabetical order under the new “Reptiles²” table subheading;
- C. Adding a “Corals” table subheading to follow the “Molluscs” table subheading; and
- D. Adding entries for three species of coral in alphabetical order by scientific

name under the “Corals” table subheading to read as follows:

§ 224.101 Enumeration of endangered marine and anadromous species.

* * * * *

(h) The endangered species under the jurisdiction of the Secretary of Commerce are:

Species ¹		Description of listed entity	Citation(s) for listing determination(s)	Critical habitat	ESA rules
Common name	Scientific name				
*	*	*	*	*	*
REPTILES ²					
Sea snake, dusky	<i>Aipysurus fuscus</i>	Entire species	Insert Federal Register citation and date when published as a final rule].	NA	NA
*	*	*	*	*	*
MOLLUSCS					
*	*	*	*	*	*
CORALS					
Coral, [no common name]	<i>Cantharellus noumeae</i>	Entire species	Insert Federal Register citation and date when published as a final rule].	NA	NA
Coral, [no common name]	<i>Siderastrea glynni</i>	Entire species	Insert Federal Register citation and date when published as a final rule].	NA	NA
Coral, [no common name]	<i>Tubastraea floreana</i>	Entire species	Insert Federal Register citation and date when published as a final rule].	NA	NA

¹ Species includes taxonomic species, subspecies, distinct population segments (DPSs) (for a policy statement, see 61 FR 4722, February 7, 1996), and evolutionarily significant units (ESUs) (for a policy statement, see 56 FR 58612, November 20, 1991).

² Jurisdiction for sea turtles by the Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, is limited to turtles while in the water.

* * * * *