

BEFORE THE SECRETARY OF COMMERCE



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**PETITION TO LIST THE
DWARF SEAHORSE (*HIPPOCAMPUS ZOSTERAE*) AS
THREATENED OR ENDANGERED UNDER THE
ENDANGERED SPECIES ACT**

CENTER FOR BIOLOGICAL DIVERSITY



NOTICE OF PETITION

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Pursuant to section 4(b) of the Endangered Species Act (“ESA”), 16 U.S.C. § 1533(b), section 553(3) of the Administrative Procedure Act, 5 U.S.C. § 553(e), and 50 C.F.R. § 424.14(a), the Center for Biological Diversity (“Center”), Tierra Curry and Noah Greenwald hereby petition the Secretary of Commerce, through the National Marine Fisheries Service (“NMFS”), to list the dwarf seahorse (*Hippocampus zosterae*) as a threatened or endangered species under the ESA, 16 U.S.C. §§ 1531 *et seq.*

The Center is a non-profit, public interest environmental organization dedicated to the protection of imperiled species and their habitats through science, policy, and environmental law. The Center has 320,000 members and online activists throughout the United States.

NMFS has jurisdiction over this Petition. This Petition sets in motion a specific process, placing definite response requirements on NMFS. Specifically, NMFS must issue an initial finding as to whether the Petition “presents substantial scientific or commercial

information indicating that the petitioned action may be warranted.” 16 U.S.C. § 1533(b)(3)(A). NMFS must make this initial finding “[t]o the maximum extent practicable, within 90 days after receiving the petition.” *Id.* Petitioners need not demonstrate that listing of the dwarf seahorse is warranted, rather, Petitioners must only present information demonstrating that such listing may be warranted. While Petitioners believe that the best available science demonstrates that listing of the dwarf seahorse is in fact warranted, there can be no reasonable dispute that the available information indicates that listing the species may be warranted. As such, NMFS must promptly make a positive initial finding on the petition and commence and complete a status review as required by 16 U.S.C. § 1533(b)(3)(B). Petitioners also request that critical habitat be designated for the dwarf seahorse concurrently with the species being listed as endangered or threatened, pursuant to 16 U.S.C. § 1533(a)(3)(A) and 50 C.F.R. § 424.12.



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EXECUTIVE SUMMARY

The dwarf seahorse is the third smallest seahorse species in the world, reaching an adult height of only 2.5 cm. This tiny seahorse lives in seagrass beds in the Gulf of Mexico, the Atlantic Coast of Florida, and the Caribbean. Due to widespread decline of seagrass, pollution from the BP Deepwater Horizon oil spill, and commercial collection, the dwarf seahorse is now threatened with extinction. The dwarf seahorse was common in the 1970s, but the species has been in decline since that time, with numerous authors reporting that the dwarf seahorse is declining (Sheridan et al. 1997, Matheson et al. 1999, Thayer et al. 1999, Thayer et al. 2000, Alford and Grist 2005, Bruckner et al. 2005, Masonjones et al. 2010). The Endangered Species Act states that a species shall be determined to be endangered or threatened based on any one of five factors (16 U.S.C. § 1533 (a)(1)). The dwarf seahorse is threatened by four of these factors and thus warrants listing as a threatened or endangered species:

Loss and Degradation of Habitat

The shallow coastal waters which support the dwarf seahorse are highly vulnerable to degradation, and seagrass loss has been extensive throughout the species' range. Since 1950 the state of Florida has lost more than half of its seagrasses, with loss in some areas exceeding 90 percent. Seagrass loss has also been dramatic in Alabama, Mississippi, Louisiana, Texas, and the Bahamas. Because the dwarf seahorse is a habitat specialist, loss of seagrass equates directly to seahorse population declines. Seahorses are particularly vulnerable to habitat loss because they have low mobility, high site fidelity, patchy spatial distribution, and complex social and reproductive behavior. The dwarf seahorse was declining even before the BP Deepwater Horizon oil spill contaminated a significant portion of the species' range. The BP catastrophe released 205.8 million gallons of oil and 205,000 metric tons of methane into the Gulf, causing fishing closures of 88,500 square miles and oiling more than one thousand linear miles of shoreline. The scientists at Project Seahorse warned that the spill and its aftermath could cause the dwarf seahorse to become extinct due to direct mortality and loss of seagrass habitat. Seahorses are sensitive to oil pollution, and contamination from the spill could persist for decades or longer, threatening both the seahorse and the seagrass on which it depends. In addition to oil pollution, seagrass is threatened throughout the dwarf seahorse's range due to declining water quality, development and human population growth, damage from boat propellers and shrimp trawlers, and global climate change.

Overutilization

Globally, the commercial trade of seahorses is threatening wild populations with depletion. Seahorses are exploited for the aquarium trade, for use as trinkets and curiosities, and for use in traditional medicines and tonic foods. As of 2001 nearly 80 countries were involved in the seahorse trade. Seahorse populations can decline rapidly when faced with exploitation due to their low population densities, highly structured social and reproductive behavior, and low rates of population increase. The dwarf seahorse is exploited for the aquarium trade, for use in curios, and potentially for use in traditional medicine. The dwarf seahorse has been a popular aquarium fish since the 1960s, and is the second most commonly exported marine ornamental fish from Florida.

The harvest of live dwarf seahorses from Florida for the aquarium trade ranges from 2,000 to nearly 100,000 individuals per year, and averages 49,000 individuals per year. Seahorses are collected by divers and as by-catch in shrimp trawlers. Due to collection and habitat loss, the seahorse can no longer be found in some areas where it was previously collected, and it is now difficult to find in other areas. Florida also has a substantial trade in dried seahorses for the curio trade, and most of the seahorses harvested in Florida may enter the curio market (Bruckner et al. 2005). Dwarf seahorses have been collected for use in curios since the 1950s. Globally traditional medicine accounts for the largest consumption of seahorses, with global trade exceeding 20 million dried seahorses annually. Use in traditional medicine poses an emerging threat to the dwarf seahorse because the market demand has become so great that small species and juveniles which were previously undesired are now sold in prepackaged mixtures. The dwarf seahorse is also threatened by collection in the Caribbean and in Mexico. Collection has contributed to the decline of the dwarf seahorse, with collectors and shrimp fishers in Florida and Mexico reporting declining seahorse harvests.

Inadequacy of Existing Regulatory Mechanisms

There are no existing regulatory mechanisms which adequately protect this species from commercial overharvest or from loss and degradation of its seagrass habitat. All seahorses are listed under Appendix II of the Convention on International Trade in Endangered Species, but this designation does not protect the dwarf seahorse from the domestic trade by which it is primarily threatened, and does not ensure that current collection levels are sustainable. A range-wide stock assessment is not available for this species, and all available data indicate that it is declining. The state of Florida limits the commercial harvest of dwarf seahorses to 400 per licensed collector per day, but there are no data which demonstrate that this quota is sufficient to protect the seahorse from overharvest. Existing regulatory mechanisms have failed to prevent the widespread loss of the seagrass habitat which supports the dwarf seahorse, and have failed to prevent the contamination of its habitat from oil pollution and numerous other water-quality issues.

Other Factors

Several other factors threaten the dwarf seahorse including noise pollution, bycatch mortality, hurricanes and tropical storms, and unknown factors. Low frequency boat motor noise negatively impacts the health, behavior, and reproductive success of dwarf seahorses (Masonjones and Babson 2003). Seahorses are commonly caught as bycatch and may not survive, even if thrown back. Hurricanes and tropical storms threaten the dwarf seahorse with direct mortality and with loss of habitat. Seahorses can also be washed into areas that lack suitable habitat, and monogamous pair bonds can be disrupted. Factors that are not yet understood could also be contributing to the decline of the dwarf seahorse, as worldwide many seahorse populations are declining for reasons that are not clear.

Due to these threats, the dwarf seahorse meets the criteria for listing and urgently needs the protections afforded by the ESA.

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INTRODUCTION

At only 2.5 cm adult height, the dwarf seahorse is the smallest of the four seahorse species that are found in U.S. waters. The dwarf seahorse occurs only in shallow seagrass areas in the Gulf of Mexico, the Atlantic Coast of Florida, and the Caribbean. Seahorses have elaborate social and reproductive behaviors, with males giving birth and partners forming monogamous bonds that are reinforced each morning with a greeting ritual. The unique life history characteristics and habitat specificity of seahorses make them particularly vulnerable to decline when faced with exploitation and habitat degradation.

The wellbeing of the seahorse is dependent on the health of the seagrass beds which it needs to survive. There has been widespread loss of seagrass habitat due to pollution, damage from boats and trawls, and other factors. The best available scientific information indicates that the dwarf seahorse is declining. In addition to its shrinking seagrass habitat, the seahorse is threatened by collection for use in the aquarium trade, as a curio, and potentially for prepackaged traditional medicines.

Because of these threats, the dwarf seahorse was imperiled even before the BP Deepwater Horizon oil spill degraded a significant portion of its range. In the aftermath of the spill, the seahorse now faces extinction and needs the protection of the Endangered Species Act to ensure its survival.

This petition summarizes the natural history of the dwarf seahorse, the population information available on the species, and the threats to the seahorse and its habitat. The petition then shows that, in the context of the ESA's five statutory listing factors, the dwarf seahorse warrants listing as endangered or threatened under the Act due to loss or curtailment of habitat or range, overutilization, the inadequacy of existing regulatory mechanisms to safeguard the species, and other factors including global climate change, bycatch in trawl fisheries, and inherent vulnerability to population decline. Lastly, the Center requests that critical habitat be designated for *H. zosterae* concurrently with listing.

NATURAL HISTORY

TAXONOMY

Hippocampus zosterae (Jordan and Gilbert 1882) is a currently recognized species (Robins et al. 1991) commonly known as the dwarf seahorse or pygmy seahorse. It is also known by the synonyms *H. rosamondae* (Borodin 1928) and *H. regulus* (Ginsburg 1933, Lourie et al. 2004). It is classified as a bony fish in the class Actinopterygii, order Gasterosteiformes, and family Syngnathidae. Seahorses make up one genus of the family Syngnathidae, which otherwise includes 55 genera of seadragons, pipefishes, and pipehorses (Foster and Vincent 2004).

DESCRIPTION

Hippocampus zosterae is the third smallest seahorse species in the world, reaching a maximum recorded adult height of 2.5 cm, and a height at first maturity of 2.0 cm, with height defined as the vertical distance from the tip of the coronet to the tip of the tail (Lourie et al. 2004). Other sources report larger sizes for *H. zosterae*. Masonjones and Lewis (1996) report that adult size in *H. zosterae* ranges from 1.6-3.8 cm, measured as the linear distance from the top of coronet to the end of tail. Ginsburg (1937) reports the total mean length of *H. zosterae* as 4.4 cm, and Vari (1982) reports the total mean length as 4.7 cm. Robbins et al. (1986) report that large specimens may reach 5 cm, and Buettner (1983) reports that *H. zosterae* can grow to 7.5 cm. It is unclear if these larger measurements are based on accurately identified specimens, given that Lourie et al. (2004), in the identification guide to the world's seahorses, report the maximum height of *H. zosterae* as 2.5 cm. Seahorses range in size from the tiny *H. denise* with a maximum height of 2.1 cm (Lourie and Randall 2003), and *H. bargibanti*, with a maximum height of 2.4 cm, to *H. abdominalis* which can reach a height of 35.0 cm (Lourie et al. 2004).

Seahorse species are distinguished in part based upon the number of bony rings and fin rays. *Hippocampus zosterae* has 9-10 trunk rings, 31-32 tail rings, 12 dorsal fin rays, and 11-12 pectoral fin rays (Lourie et al. 2004). The dorsal fin is supported by 2 trunk rings and no tail rings. Bruckner et al. (2005) report that the dark margin of the dorsal fin is a distinguishing characteristic for this species. The coronet is high and columnar or knob-like, and lacks spines or projections. The body spines are low and knob-like, and the skin is often covered in tiny warts. The snout is short, at less than one-third of the head length, and males have longer snouts and longer tails than females. Males and females are commonly the same size, and males are distinguished by the presence of a brood pouch, or marsupium, an elevated ventral ridge around the height of the top few caudal segments (Masonjones and Lewis 1996). Seahorses have eyes that swivel independently and prehensile tails (Foster and Vincent 2004).

Hippocampus zosterae individuals can be beige, yellow, green, or nearly black, and have variable mottling which can range from absent to distinct. Some individuals have white markings, and some have dark spots (Lourie et al. 2004). Seahorses use camouflage, changing color and skin filament patterns to blend with their environments, and individuals removed from the wild and held in aquaria usually lose their skin filaments within a few days (Ibid.) They may change color rapidly when interacting with one another, with males commonly becoming brighter during courtship (Foster and Vincent 2004).

There are at least four species of seahorses which occur in the United States including *H. erectus*, *H. ingens*, *H. reidi*, and *H. zosterae* (Lourie et al. 2004). *H. zosterae* can be distinguished from *H. reidi* and *H. erectus*, with which it may co-occur, by its body ring and dorsal ray counts, small size, short snout, and high, columnar coronet (Alford and Grist 2005).

RANGE

Hippocampus zosterae occurs along the Atlantic coast of Florida, the entire Gulf of Mexico including eastern Mexico, and the Caribbean including the Bahamas, Cuba, and Bermuda (Masonjones and Lewis 1996, Beck et al. 2000, Musick et al. 2000, Lourie et al. 2004). CITES (2010) reports that *H. zosterae* occurs in Belize, though this is likely erroneous (p. 31). Though the species has a relatively wide range, its distribution is patchy and restricted to appropriate habitat types, and abundance is generally low (Bruckner et al. 2005).

In Florida, *H. zosterae* occurs in most estuaries, but is more abundant in south Florida and the Keys (Bruckner et al. 2005). *Hippocampus zosterae* is known to occur at Timucuan Ecological and Historic Preserve (NPS 1996, p. 62), J.N. "Ding" Darling National Wildlife Refuge Complex (FWS 2010, p. 289), Terra Ceia Aquatic Preserve (FDEP 2009, p. 75), Mosquito Lagoon Aquatic Preserve (FDEP 2009b, p. 112), Cockroach Bay Preserve State Park (FDEP 2004), Cedar Key Scrub State Reserve (FDEP 2005, p. A 4-13), Faka Union Bay, Fakahatchee Bay, and Pumpkin Bay (FDEP 2010).

Bruckner et al. (2005) report that *H. zosterae* becomes increasingly uncommon west of Florida. Beck et al. (2000) report that within the Gulf, there are 7 independent collection records for *H. zosterae* in Lower Laguna Madre and in South Apalachee Bay. There are 4 independent records for North Apalachee Bay and Corpus Christi Bay. There are two records for St. George Sound and East Mississippi Sound, and one independent collection record each for *H. zosterae* in Aransas Bay, Terrebonne/Timbalier Bays, Chandeleur Sound, Perdido Bay, and Pensacola Bay (p. 29).

In Mexico, *H. zosterae* occurs south to the Gulf of Campeche, and reported Mexican localities include Laguna Madre de Tamaulipas, Laguna de Tamiahua in Veracruz, and Laguna de Términos and Champotón in Campeche (Bruckner et al. 2005, p. 69).

HABITAT

Hippocampus zosterae is restricted to tropical and subtropical/warm-temperate waters where it is primarily associated with seagrass beds in shallow water (Lourie et al. 2004, Alford and Grist 2005). It is particularly associated with eelgrass (*Zostera sp.*) from which its name is derived. Other seagrass species with which it is associated include manatee grass (*Syringodium filiforme*), turtlegrass (*Thalassia testudinum*), star grass (*Halophila engelmanni*), widgeon grass (*Ruppia maritima*), and shoal grass (*Halodule beaudettei*) (Strawn 1958). This seahorse is more abundant in areas with denser seagrass and higher seagrass canopy cover (Bruckner et al. 2005). Sogard et al. (1987) found that total seagrass shoot density is positively correlated with density of *H. zosterae*. Foster and Vincent (2004) report that the maximum depth of detection for this species is 2 meters. Though the dwarf seahorse is primarily a seagrass specialist, some authors have reported *H. zosterae* from different types of habitat including mangrove areas (Odum et al. 1982), unattached algae (Tabb and Manning 1961), and inshore drifting vegetation (Hoese and Moore 1998).

Hippocampus zosterae appears to be most common in areas of high salinity (~30 ppt) (Alford and Grist 2005, Bruckner et al. 2005). Hoese and Moore (1998) report that *H. zosterae* is restricted to high salinity areas, but Powell et al. (2002) report that they have been collected in mesohaline, polyhaline, and euhaline waters (p. 15), and Strawn (1958) reported breeding following a low salinity period (~10 ppt). This species is more likely to be found in protected bays and lagoons in areas with low water flow and sediment with high organic content (Bruckner et al. 2005).

Strawn (1958) found that during winter at Cedar Key, FL, *H. zosterae* tend to concentrate in deeper water and in tide pools where vegetation is most abundant. Masonjones et al. (2010) report that it is likely that some portion of the Tampa Bay population migrate out of the shallow seagrass beds during the dry season.

There has been drastic decline of seagrass throughout the range of the dwarf seahorse due to a variety of factors, and its remaining seagrass habitat continues to be threatened by pollution, loss of wetlands and mangroves, oil and gas development, scarring from boat propellers, trawling, and global climate change.

LIFE HISTORY

Syngnathids are unique in that the males carry the young and give live birth. The seahorse marsupium functions like the mammalian uterus with embryos developing in a protected environment where they are provided with oxygen via a capillary network, with the osmolarity of the chamber adjusting as the pregnancy progresses, and waste products diffusing out through the father's blood stream. The eggs develop enveloped in epithelial tissue in compartmentalized pits in the wall of the pouch where they absorb yolk, and they may also be nourished by male-contributed inorganic compounds (Foster and Vincent 2004), though they may be nutritionally independent of the male (Azzarello 1991).

All seahorse species which have been studied in the wild appear to be monogamous within a single breeding cycle, with the male incubating eggs from only one female, though some seahorse species have been found to be polygamous across breeding cycles (Wilson et al. 2003, Foster and Vincent 2004). *Hippocampus zosterae* is thought to be a monogamous species, with the males making considerable investments in brooding the offspring (Masonjones 1997, Vincent and Giles 2003, Wilson et al. 2003). *Hippocampus zosterae* populations may have female-biased sex ratios (Strawn 1958), similar to female-biased sex ratios which have been reported for *H. erectus* and *H. abdominalis*, though this may be a bias of sampling method, and underwater populations may actually have equal numbers of males and females (Baum et al. 2003).

Masonjones and Lewis (1996) observed *H. zosterae* mating in the laboratory and report a monogamous system with four distinct courtship phases. During the first phase, one or two days prior to copulation, the male and female meet in the mornings and alternately display reciprocal side-to-side body quivering. During the next two phases, which take

place on the day of mating, the females point with their heads raised upward, and then the males point in response. During the final phase the male and female repeatedly rise in tandem until the female transfers her eggs into the male's brood pouch. During gestation, monogamy and pair bonding are reinforced by daily morning greeting rituals in which the female visits the male (Lourie et al. 2004). Across species, morning interactions continue throughout the pregnancy and likely help synchronize egg preparation with the end of male pregnancy so that the male can become impregnated again quickly upon giving birth (Foster and Vincent 2004). Male and female *H. zosteræ* pairs remain together and mate repeatedly over the course of the breeding season, with males often re-mating the same day they release their young (Masonjones and Lewis 1996, Baum et al. 2003). Monogamous partners eschew interactions with other seahorses, and the pair bond only terminates if the partner disappears (Vincent and Sadler 1995).

Even though male seahorses become pregnant, seahorses maintain conventional sex roles with males competing for access to females by wrestling and snapping at each other, and females carefully selecting their mates (Vincent 1994, Foster and Vincent 2004). Masonjones and Lewis (1996) found that male dwarf seahorses are more active than females in initiating courtship. Vincent (1994) reports that males which succeed in mating are more active and heavier than their rivals.

Masonjones and Lewis (1996) report *H. zosteræ* clutch size as 12.4 (+ 2.2) eggs (n = 9), with 3-16 young being born after a 10-day gestation period. Foster and Vincent (2004) report that *H. zosteræ* gestation ranges from 10-13 days. Pairs remate within 4-20 hours of the male giving birth. Strawn (1958) reports largest female clutch size of *H. zosteræ* as 69 eggs, and largest brood as 55 released young. Foster and Vincent (2004) suggest that the large clutch size reported by Strawn is likely erroneous because it is based on histology of the ovaries which can lead to an overestimate because of the continuous nature of egg production in the spiral ovaries of seahorses. Vincent (1990) reports a clutch size of five for *H. zosteræ*. Breder (1940) reports males birthing 6-8 young. Egg diameter is 1-3 mm, and length at birth is 7-9 mm (Foster and Vincent 2004). Young are fully formed miniature seahorses, and are precocial, receiving no parental care post-birth.

H. zosteræ carry two broods per month (Vari 1982), and the breeding season runs from February to October or November and appears to be correlated more so with day length than with water temperature (Strawn 1953, Vari 1982, Foster and Vincent 2004). Strawn (1958) reports that young *H. zosteræ* grow rapidly and reach maturity in three months, allowing the production of at least three generations annually.

The fitness of young seahorses may be influenced by the age and size of the parents and by whether they are born early or late during the breeding season. Lourie et al. (2004) report that across species, brood size increases with male height. Vincent and Giles (2003), however, found that in *H. whitei*, female mass was the key determinant of brood size, and that the number of young produced was not related to male size. Dzyuba et al. (2006) found that in *H. kuda*, offspring from older and larger parents had higher initial postnatal growth rates and higher survivorship than offspring of younger, smaller parents, suggesting that pouch physiology and size may influence the future fitness and survival

of offspring. They also found that older parents had significantly larger average brood sizes than younger parents. In a study of wild *H. whitei*, Vincent and Giles (2003) found that the number of young per brood and the size of young decreases over the breeding season.

H. zosterae commonly live for only one year, only rarely surviving past their first year (Strawn 1958, Alford and Grist 2005). Larger species are thought to live for three to five years (Foster and Vincent 2004). Natural mortality rates in seahorses are largely unknown, but are thought to be low (Vincent 1995b, Foster and Vincent 2004). Incidence of mortality appears to decrease after the first few weeks post birth (Job et al. 2006).

ECOLOGY

Hippocampus zosterae is a diurnal species (Froese and Pauly 2008), though some species of seahorses are nocturnal. Seahorses consume live mobile prey, relying on their camouflage to allow them to sit and wait to ambush prey such as small crustaceans, amphipods, fish fry, and invertebrates. Seahorses lack teeth and they do not have a differentiated stomach (Foster and Vincent 2004). When a prey item swims close to the seahorse's mouth, the seahorse rapidly intakes water through its snout, allowing it to consume any item small enough to fit through the snout (Foster and Vincent 2004). Tipton and Bell (1988) examined the diet of juvenile dwarf seahorses in Tampa Bay from April to October and found that harpacticoid copepods make up the majority of the diet. They concluded that *H. zosterae* consume sufficient invertebrate prey so as to affect the structure of the invertebrate community (Tipton and Bell 1988). Strawn (1958) suggests that dwarf seahorses are an important trophic link in seagrass communities.

Due to their camouflage capabilities and unpalatable bony plates and spines, it is assumed that sub-adult and adult seahorses have few natural predators (Lourie et al. 1999). Numerous species, however, have either been documented to have seahorses in their stomachs or have been reported to prey on seahorses including various invertebrates, numerous fishes including red snapper, dorado, rays, skates, tuna, dolphin fish, anglerfish, flatheads, sea urchins, blue cod, ling, red cod, sea perch, and trumpeter, loggerhead sea turtles, cormorants, penguins, and other water birds, and marine mammals (Foster and Vincent 2004, Kleiber et al. 2010). Observations of seahorses with shortened tails may also indicate that partial predation by crabs may threaten seahorses (Baum et al. 2003). Juveniles are thought to be more vulnerable to predation than adults (Foster and Vincent 2004).

Kleiber et al. (2010) reviewed 135 accounts of predation on seahorses and pipefishes and identified 82 predator species. They found that seahorses are generally depredated in low numbers and only form a high proportion of predator diets during seahorse population booms or die-offs, when individuals are concentrated in floating marine vegetation, or when juveniles are abundant during the breeding season. They suggest that seahorses may occur in the open ocean more than is commonly thought, using floating mats of vegetation.

Seahorses are considered to be feeble swimmers with low mobility, and are thought to disperse by clinging to rafts of drifting macroalgae, vegetation, or debris (Foster and Vincent 2004, Masonjones et al. 2010). Young seahorses are more likely to disperse than adults, and juvenile dispersal likely contributes to gene flow among populations, though the extent of juvenile dispersal is not known (Foster and Vincent 2004). Strong storm events may serve as a mechanism to cast seahorses adrift and aid dispersal over long distances (Foster and Vincent 2004). Woodall et al. (2009) identified a specimen of *H. erectus* taken from the eastern Atlantic Ocean, well outside its known range in the western Atlantic, and attribute the dispersal to either rafting on a floating object in the ocean current, transport via ballast water, or intentional release from an aquarium.

Most seahorse species are site-faithful, maintaining small home ranges, and living in highly structured communities, at least during the breeding season (Vari 1982, Masonjones and Lewis 1996, Vincent et al. 2004, Curtis and Vincent 2006). However, Masonjones et al. (2010) found that at least some fraction of the *H. zosterae* population in their study in Tampa Bay appeared to be mobile, with a low recapture rate and high rate of site turnover. The tiny *H. breviceps* also appears to be more mobile than has been reported for other seahorse species (Moreau and Vincent 2004). Seahorses do not appear to defend their home ranges, likely because they do not need to defend oviposition sites or young (Foster and Vincent 2004).

Across seahorse species, distribution is patchy and density tends to be low (Baum et al. 2003, Foster and Vincent 2004, Bruckner et al. 2005). Seahorse density shifts across site and season, and varies temporally, particularly with lunar phase (Baum et al. 2003, Masonjones et al. 2010). Serviss and Sauers (2003) report *H. zosterae* density in Sarasota Bay as 0.0 seahorses per 100 square meters in subtidal pools, seagrass beds, in channels, and along restored marsh edges, and as 0.6 seahorses per 100 square meters along the natural marsh edge. Powell et al. (2002) report *H. zosterae* mean density in north Florida Bay as 5.85 seahorses per 100 cubic meters. Thayer et al. (1999) report *H. zosterae* density in Florida Bay as less than 19 individuals per hectare. Masonjones et al. (2010) found *H. zosterae* density in Tampa Bay to be even across sites when separated across seasons, with an overall mean density of 0.08 animals per square meter, and a range of 0.02 animals per square meter (p. 124).

ABUNDANCE and TREND

Across species seahorses are generally found in low abundance which makes populations vulnerable to decline (Foster and Vincent 2004, Martin-Smith and Vincent 2005). In Florida, *H. zosterae* is common in many bays and estuaries, but is “not a numerically dominant community component” (Masterson 2008). Due to its wide range, *H. zosterae* is perceived to be an abundant species, though most survey data do not support this assumption. Zieman (1982) describes *H. zosterae* as “abundant in seagrass throughout south Florida,” but this generalization is contradicted by the survey data reported in the same study. Based on surveys reporting abundance as either rare, present, common, or abundant, *H. zosterae* is not reported as abundant in any survey (Zieman 1982). It is reported as common in South Biscayne Bay (Bader and Poessler 1971), as present in

Porpoise Lake (Hudson et al. 1970), and as rare in all other surveyed areas including North Biscayne Bay, Card Sound, Metecumbe Key, Whitewater Bay, Fakahatchee Bay, Marco Island, and Rookery Bay (Zieman 1982, p. 124).

Powell et al. (2002) identify *H. zosterae* as among the dominant species in North Florida Bay, though only a total of 15 *H. zosterae* were collected, and the species was detected at only 9 of 202 sites. They report that *H. zosterae* were found primarily at Bradley Key, and were absent to rare at Middle Ground, McCormick Creek/Crocodile Point, and East Creek. Serviss and Sauers (2003) caught only 16 *H. zosterae* out of a total of 315,208 organisms collected in 2002 using a 21.3-m standard seine in Sarasota Bay. The Florida Department of Environmental Protection (2010) report a total of 54 *H. zosterae* from sampling in Faka Union Bay, Fakahatchee Bay, and Pumpkin Bay, though sampling method and time interval are unclear.

Surveys which have been conducted in Florida Bay, the Florida Keys, and Tampa Bay indicate that *H. zosterae* is declining (Sheridan et al. 1997, Matheson et al. 1999, Thayer et al. 1999, Bruckner et al. 2005, Masonjones et al. 2010). During sampling in 1984-1985, Sogard et al. (1987) caught 398 *H. zosterae* in 828 throw trap samples in Florida Bay, indicating that the species may have been relatively abundant at this time. Alford and Grist (2005) report that *H. zosterae* was regarded as common before 1970, but that “numbers have steadily decreased over subsequent years” (p. 88). They attribute the decline to loss of seagrass habitat, and state that collectors now find it difficult and time-consuming to locate *H. zosterae* around the Florida Keys, and that the species has not been detected in Mississippi Sound since 1987. Surveys conducted in the Florida Keys in 2003 detected seahorses in only 8 of 20 surveyed healthy seagrass beds, and revealed that only 5 of the 20 beds had robust seahorse populations (>2 seahorses recovered with a modified pushnet per sampling event) (Bruckner et al. 2005, p. 94).

Several studies have shown quantitative declines of *H. zosterae* in Florida Bay. Sheridan et al. (1997) found that the abundance of *H. zosterae* in western Florida Bay declined significantly from 1984 to 1993 coincident with a die-off of turtlegrass (*Thalassia testudinum*) (p. 816). Thayer et al. (1999) compared seagrass densities between 1984-1985 and 1994-1996 in Florida Bay and found that seagrass densities (*T. testudinum*, *H. wrightii*, and *S. filiforme*) within most of the basin and channel strata sampled in 1994-1996 had decreased by as much as 100 percent. They documented a concomitant decrease in the proportion of fish fauna represented by canopy-dwelling seagrass inhabitants such as *H. zosterae* (p. 518, 527). Thayer et al. (2000) report that the mean density of *H. zosterae* in the Gulf Transition Subdivision in Florida Bay in 1996 was low compared to the mean density in 1984. Similarly, Matheson et al. (1999) report that the abundance of *H. zosterae* in Florida Bay showed a significant inter-decadal decline from the 1980s to the 1990s (Mann-Whitney U-tests, $p < 0.01$) (p. 546). During sampling in seagrass canopy on mud banks in Florida Bay during the 1984-1986 sampling period, 116 *H. zosterae* were collected, but only 61 were collected during the 1994-1996 sampling period (Table 3, p. 541) (Matheson et al. 1999). The frequency of occurrence fell from 88 percent to 56 percent over the decade (Matheson et al. 1999, Table 8, p. 546).

In Tampa Bay, Masonjones et al. (2010) “observed a substantial decrease in overall syngnathid population size” during the course of a two-year study (p. 130). The measured decrease in seahorse population size occurred in the same time frame as the construction of a marina adjacent to the research site, and in the larger context of loss of seagrass in Tampa Bay.

In the U.S. Gulf of Mexico west of Florida, *H. zosterae* is widely distributed but does not appear to be abundant in most areas (Bruckner et al. 2005). Numerous studies conducted in the northern Gulf have failed to detect any specimens of *H. zosterae* (Bruckner et al. 2005). *Hippocampus zosterae* appears in moderate numbers in at least some Texas waters (Bruckner et al. 2005). This species was once thought to be common in Corpus Christi Bay, but there are few current records from eastern Texas and western Louisiana, though this is potentially related to low sampling effort (Beck et al. 2000). In the mid-1990s, the Center for Coastal Studies (1996) identified *H. zosterae* as a “dominant component” of the warm season fish community utilizing turtlegrass habitats in Redfish Bay, Corpus Christi, though numbers of seahorses detected are not provided. In 2007 and 2008, sampling in Corpus Christi (Packery Channel, Laguna Madre) detected very low abundance of *H. zosterae* (Stunz and Reese 2008). The researchers caught two *H. zosterae* during each winter of the study, no *H. zosterae* were caught in spring 2007, two were caught in spring 2008, none were caught in summer 2007, two were caught in summer 2008, and one was caught in fall 2007 (Stunz and Reese 2008).

Data on the abundance and trend of *H. zosterae* outside of the United States are not readily available. Baum and Vincent (2005) report that seahorse populations may have declined substantially throughout many regions of Latin America in recent decades due primarily to incidental mortality and habitat damage caused by commercial trawling (p. 317). Shrimp trawl fishers in Mexico have noted drastically reduced catch of seahorses over time, with estimated declines of greater than 90 percent in catch per unit effort in the Gulf of Mexico (Baum and Vincent 2005, p. 316). Targeted seahorse fisheries in Mexico have also reported declines exceeding 50 percent (Baum and Vincent 2005, p. 316).

The dwarf seahorse was declining before the BP Deepwater Horizon catastrophe contaminated a significant portion of the species’ range. Due to direct mortality, loss of seagrass habitat, and ongoing contamination from the spill, the scientists at Project Seahorse have warned that the species could now face extinction.

CONSERVATION STATUS

The U.S. population segment of *H. zosterae* is ranked by the American Fisheries Society as Vulnerable due to rarity, restricted habitat, degradation of habitat, and loss of seagrass beds (Musick et al. 2000, p. 13). It is classified by the state of Florida as a Species of Greatest Conservation Need (FFWCC 2005). The dwarf seahorse is listed as a Species of Concern by the Commonwealth of Puerto Rico. The Nature Conservancy (TNC) identifies *H. zosterae* as an imperiled species, and it is on TNC’s Target List for the Northern Gulf of Mexico ecoregion (Beck et al. 2000). The Commission for Environmental Cooperation categorizes *H. zosterae* as an at-risk species (Wilkinson et al.

2009). The status of *H. zosterae* is not ranked (SNR) by NatureServe (2010), and is considered to be Data Deficient by the International Union for Conservation of Nature (IUCN 2003). It was formerly ranked as Vulnerable by the IUCN (1996) based on suspected past declines in occupancy, occurrence and habitat, as well as on potential levels of exploitation, but under the new ranking criteria, appropriate data were considered to be lacking. The entire genus *Hippocampus* is listed in Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), effective May 2004, meaning all species in the genus may become threatened with extinction if international trade is not closely controlled. Mexico classifies *H. zosterae* as a species subject to special protection in the NOM-059-SEMARNAT-2001 (CITES 2004).

None of the above conservation designations provide *H. zosterae* with essential regulatory protection. Listing under CITES Appendix II allows the species to be traded internationally, but requires that shipments include documentation verifying that the specimens were obtained legally. The dwarf seahorse is declining (Sheridan et al. 1997, Matheson et al. 1999, Thayer et al. 1999, Thayer et al. 2000, Alford and Grist 2005, Bruckner et al. 2005, Masonjones et al. 2010) and faces an array of threats including habitat loss and degradation, collection, and global climate change. The dwarf seahorse needs to be protected under the Endangered Species Act to ensure its continued survival.

***HIPPOCAMPUS ZOSTERAE* WARRANTS PROTECTION UNDER THE ESA**

Hippocampus zosterae is both declining and threatened. The seahorse is threatened by at least four of the five listing factors under the Act. There has been widespread loss and degradation of the seagrass habitat on which this species depends. There is no question that the BP Deepwater Horizon catastrophe degraded a significant portion of this species' range. Global climate poses another threat to the habitat of the dwarf seahorse.

Hippocampus zosterae is also threatened by collection and by numerous other factors including trawling, noise pollution, and life-history characteristics which make populations vulnerable to decline. There are no existing regulatory mechanisms which are adequate to protect this species. It is clear that *Hippocampus zosterae* may warrant listing under the Endangered Species Act, and that a status review is warranted for this species.

THREATS

PRESENT OR THREATENED DESTRUCTION, MODIFICATION OR CURTAILMENT OF HABITAT OR RANGE

Overview

Seahorses inhabit shallow coastal waters which are highly vulnerable to habitat degradation caused by various anthropogenic activities (Foster and Vincent 2004, p. 45). Seahorses are threatened by numerous forms of habitat degradation including dredging, pollution, siltation, and loss of vegetation (Vincent 1995a, p. 125). Several aspects of seahorse biology render them particularly susceptible to habitat degradation including

low mobility, high site fidelity, patchy spatial distribution, low fecundity, low population density, complex reproductive behavior, and monogamous mating system. For example, increases in turbidity hinder the ability of seahorses to find one another and to receive the visual cues required for courtship and mating (Masonjones et al. 2010, p. 119). Degraded habitat conditions have been correlated with extremely low seahorse densities (Marcus et al. 2007, p. 1483), impaired seahorse reproduction (Masonjones and Rose 2009), and seahorse population declines (Beck et al. 2000, Alford and Grist 2005, Masonjones et al. 2010, p. 118).

The dwarf seahorse in particular is susceptible to decline because the shallow seagrass beds it inhabits are themselves highly sensitive to degradation (IUCN 2010). Numerous factors are known to be degrading the seagrass habitat of *H. zosterae* including coastal development, tourism, recreation, oil and gas activities, dredging, shipping, loss of wetlands, trawling, coastal erosion, and global climate change (Beck et al. 2000, Florida Fish and Wildlife Conservation Commission 2005, Wilkinson et al. 2009).

Loss and Degradation of Seagrass Habitat

As a habitat specialist, the health of dwarf seahorse populations is completely dependent on the health of the seagrass beds on which the species depends for survival. Seagrass meadows are among the most threatened ecosystems on earth, with seagrass cover declining across the globe at an accelerating rate (Waycott et al. 2009, p. 12377). Seagrasses are sensitive to a variety of environmental stressors and are in global decline due to deteriorating water quality, physical damage from development, dredging, and boating, and rising temperatures (Hughes et al. 2009, p. 242, Masonjones et al. 2010, p. 118). Turbidity caused by dredging results in siltation of seagrass beds in addition to the actual destruction of the areas being dredged (FDEP 2004b). Sewage outfalls, industrial wastes, power plant outfalls, oil spills, and extreme high temperatures can all be lethal to seagrasses (FDEP 2009). Oil spills are particularly harmful to seagrasses, as even low concentrations of oil can greatly reduce the ability of seagrasses to photosynthesize (FDEP 2009, p. 20). Loss of seagrasses causes an even greater decline in water quality due to an increase in nutrients resulting from seagrass decomposition, an increase in turbidity due to loss of sediment stabilization, and loss of seagrass filtration (Masonjones et al. 2010).

There has been documented decline of seagrass meadows throughout the range of *H. zosterae*. FWS (2010) reports that since 1950 the state of Florida has experienced over a 50 percent decline of seagrasses (p. 70). The Florida Fish and Wildlife Conservation Commission (FFWCC 2005) reports that the status of submerged aquatic vegetation in the state is “poor and declining” (p. 332). The statewide threat rank for seagrass habitat is “very high.” The FFWCC (2005) lists numerous threats to seagrass habitat including water pollution from nutrient loading, sedimentation, chemicals and toxins, industrial spills, non-point source runoff, algal blooms, propeller scarring, boat groundings, boat wakes, coastal construction, dock construction, seagrass shading from docks, dredge and fill activities, channel modifications, climate variability, fishing pressure and fishing gear impacts, invasive species, loss of key predators and herbivores, withdrawal of ground and

surface water, hydrological modifications that disrupt natural salinity patterns, severe storm events, and pathogens (p. 336).

Stresses ranked as “very high threats” to seagrass include altered water quality, habitat destruction, altered species composition, and sedimentation. Sources of stress for the “very high threats” include coastal development, harmful algal blooms, inadequate stormwater management, and channel modification/shipping lanes. Stresses ranked as high threats to seagrass include nutrients, altered structure, erosion, altered hydrologic regime, and altered primary productivity (FFWCC 2005, p. 337). Sources of stress for the “high” threats include nutrient loads, incompatible industrial operations, dam operations, climate variability, surface and groundwater withdrawal, invasive plants and animals, roads, bridges, and causeways, shoreline hardening, incompatible fishing pressure, and disruption of longshore transport of sediments (p. 338).

In northwestern Florida, including Pensacola Bay, Choctawhatchee Bay, St. Andrew Bay, and Big Bend, seagrass meadows have declined due to beachfront alteration, dredge and fill activities, sewage and industrial waste discharges, hydrologic alterations to watersheds, and propeller scarring from boats (USGS 2004). Big Bend, Florida (the area east of St. Andrew Bay south to Tampa Bay) lost approximately 270,000 hectares of seagrass from 1984-1992, a 52 percent decline (USGS 2004, Waycott et al. 2009, Table S1). Pensacola Bay has also lost more than half of its historical seagrass meadows (Waycott et al. 2009).

Southwestern Florida, including Tampa Bay, Sarasota Bay, Greater Charlotte Harbor, Florida Bay, Naples Bay, Faka Union Bay, and Fakahatchee Bay, has lost seagrass due to rapid population growth, dredge and fill activities, nutrient loading, and propeller scarring (USGS 2004). Tampa Bay has lost more than 60 percent of its historical seagrass cover (Waycott et al. 2009). Naples Bay has experienced a 90 percent loss of seagrass since the 1950s (FDEP 2010, p. 18). Seagrasses in Naples Bay have declined due to altered freshwater input, increased nutrient levels, low water clarity, dredging for navigation and development, boat wakes, and high sediment loads (FDEP 2010, p. 18). From 1950-1987, Sarasota Bay lost nearly one-third of its seagrasses (Waycott et al. 2009). In Fakahatchee Bay in the early 1970s, there were three species of seagrasses, and nearly 60 percent of the bay harbored seagrasses. In 2005, only one seagrass species survived in the bay, and the percentage of the bay supporting seagrasses had declined (FDEP 2010, p. 21). Similarly, nearly a quarter of Faka Union Bay supported seagrasses of three species in the early 1970s, but in 2005 no seagrass remained in the bay (FDEP 2010, p. 21).

The decline of seagrasses in Florida Bay has been particularly well documented. Thayer et al. (1999) compared seagrass densities between 1984-1985 and 1994-1996 in Florida Bay, using the same stations and sampling techniques. They found that seagrass densities within most of the basin and channel strata sampled had decreased by as much as 100 percent. The Atlantic stratum declined by an average of 57 percent and the central stratum declined by an average of 80 percent. At individual stations, they documented declines up to 100 percent, and also documented shifts in dominance (p. 525). The authors found that the proportion of fish fauna represented by canopy-dwelling fish

species such as *H. zosterae* also decreased (p. 518). Hall et al. (1999) found that the biomass of *T. testudinum* decreased by 28 percent from 1984-1994 in Florida Bay, and that the abundance of the subordinate seagrass species *Halodule wrightii* and *Syringodium filiforme* decreased by more than 90 percent over the decade. Matheson et al. (1999) report that from 1987-1991, Florida Bay lost more than 4,000 hectares of *T. testudinum*, with an additional 23,000 ha showing signs of degradation (p. 535). They report that the density of *T. testudinum* declined by 22 percent from 1983-1994, based on measurements taken at 108 stations throughout the bay (p. 535). The die-off of seagrasses in Florida Bay has been attributed to multiple causes including turbidity, hypersalinity, hypoxia, sulfide toxicity, eutrophication, phytoplankton blooms, altered estuarine function, development, pathogens, and abnormally warm temperatures (Fourqurean and Robblee 1999, Matheson et al. 1999, Zieman et al. 1999).

Seagrass has also declined on the Atlantic Coast of Florida. The Mosquito Lagoon area has undergone one of the greatest losses of seagrass coverage in the state, with a 94 percent loss since 1943 (FDEP 2009b). Seagrasses in Mosquito Lagoon Aquatic Preserve are threatened by elevated levels of suspended solids which impede photosynthesis (FDEP 2009b, p. 27). In the Indian River Lagoon complex, Fletcher and Fletcher (1995) analyzed seagrass distribution and abundance from Ponce de Leon Inlet in Volusia County to Jupiter Inlet in Palm Beach County and found that from 1943-1992, the maximum depth of seagrass beds decreased by as much as 50 percent. They estimate that seagrass abundance declined by 11 percent from 1970-1992, and by 16 percent from 1986-1992 (p. 49). Waycott et al. (2009) report that seagrass cover in the Indian River Lagoon area declined by 25 percent from 1970-1992. There are also large areas of seagrass die-off in the Florida Keys (FDEP 2004b, p. 18). Seagrasses have declined in the Florida Keys National Marine Sanctuary due to nutrient loading and propeller scarring (Fourqurean 2009).

Loss of seagrass in Alabama and Mississippi has also been dramatic. Mobile Bay, Alabama lost more than 80 percent of its seagrass from 1981-2003, and Perdido Bay lost 75 percent (Waycott et al. 2009, Table S1). Mississippi Bays lost nearly 90 percent of seagrass cover from 1969-1992, and Mississippi Sound lost 50 percent of its seagrass from 1992-2003 (Ibid.). The U.S. Geological Survey (2004) identifies causes of seagrass loss in Alabama, Louisiana, and Mississippi as turbidity, dredging, erosion of offshore islands, recreational boat use, coastal development, wastewater effluent, and tropical storms.

Texas has also experienced drastic seagrass decline and concomitant decline of seagrass-dwelling organisms. Roberts and Hawkins (1999) report that shallow nearshore seagrass beds in southeastern Texas declined rapidly following the 1970s due to human activities at the coast (p. 242). Galveston Bay experienced a decline in submerged vascular vegetation of approximately 90 percent from 1956-1990 due to shoreline industrial and municipal development, excess inputs of pollutants and wastewater discharges, channelization, dredging projects, subsidence, and alterations in bay-water circulation dynamics (Pulich and White 1990, p. 1). The U.S. Geological Survey (2004) reports that more than 95 percent of historic seagrass cover has been lost from the Galveston Bay

region (p. 19). Corpus Christi has also undergone loss of seagrass, associated with eutrophication and harmful algal blooms (Breier et al. 2004). A decade long algal bloom in Laguna Madre/Baffin Bay caused a decline in seagrasses beginning in 1989 (An and Gardner 2000). The U.S. Geological Survey (2004) identifies numerous causes of seagrass decline in Texas including turbidity, dredging, Texas brown tide, nutrient loading, propeller scarring, shoreline development, and tropical storms.

Seagrass has also declined in Bermuda, with the Bermuda Platform experiencing a 23 percent decline in seagrass from 1997-2004 (Murdoch et al. 2007, Waycott et al. 2009, Table S1).

In some areas the plight of seagrass has modestly improved, but loss of seagrass continues to be a primary threat to the dwarf seahorse. In Tampa Bay for example, a reduction in nitrogen loads has improved water clarity and led to some seagrass recovery, yet overall Tampa Bay has lost 63 percent of its seagrass (Petition Table 1, Waycott et al. 2009, p. 12379 and Table S1). Seagrass cover at Mosquito Lagoon has increased by 23 percent since 1970 (Waycott et al. 2009), yet seagrass cover in the area is 94 percent less than it was in 1943 (FDEP 2009b). In Texas, seagrass cover has increased in Upper Laguna Madre, but decreased in Lower Laguna Madre; in Laguna Madre as a whole, seagrass has decreased by four percent since 1965 (USGS 2004, p. 15). Similarly, seagrass cover increased in East Matagorda Bay, but decreased in Matagorda Bay (Waycott et al. 2009, Table S1).

Because *H. zosterae* is entirely dependent on seagrass habitat, declines in seagrasses are reflective of declines in seahorse populations. Numerous authors have attributed declining *H. zosterae* populations to loss and degradation of seagrass habitat. The American Fisheries Society has recognized loss and degradation of seagrass beds as a primary threat to *H. zosterae* (Musick et al. 2000, p. 13). Alford and Grist (2005) attribute the decline of *H. zosterae* in Florida since 1970 to the reduction in the extent of seagrass beds. Beck et al. (2000) report that Mississippi Sound populations of *H. zosterae* have declined in conjunction with loss of seagrass beds. Sheridan et al. (1997) found that the abundance of *H. zosterae* in western Florida Bay declined significantly from 1984 to 1993 coincident with a die-off of turtlegrass (*Thalassia testudinum*) (p. 816). Likewise, Thayer et al. (1999) and Matheson et al. (1999) documented a decline in seagrass density and a decline in *H. zosterae* abundance in Florida Bay from the 1980s to the 1990s. Masonjones et al. (2010) “observed a substantial decrease in overall syngnathid population size” during the course of a two-year study which occurred in the same time frame as the construction of a marina adjacent to the research site, and in the larger context of loss of seagrass in Tampa Bay (p. 130).

Due to declining seagrass meadows, the dwarf seahorse had already experienced widespread habitat loss even before the April 2010 BP Deepwater Horizon oil spill. The aftermath of the spill and its effects on seagrass beds in the Gulf of Mexico could extirpate the dwarf seahorse throughout a significant portion of its range (Project Seahorse 2010).

Table 1. Documented Seagrass Loss in the Range of *H. zosterae*. Data from Waycott et al. 2009, Table S1.

Location	Rate of Change Per Year	Acreage First Year Measured	Recent Acreage	Percentage Change	Time Frame
FLORIDA					
Big Bend	-9.15	520,000	250,000	-52%	1984-1992
Florida Bay	-1.89	142,473	124,787	-12%	1987-1994
Greater Charlotte Harbor	-0.35	23,127	21,802	-6%	1982-1999
Indian River Lagoon	-1.30	17,227	12,935	-25%	1970-1992
Pensacola Bay	-2.31	3800	1814	-52%	1968-1992
Sarasota Bay	-0.56	4,886	3,476	-29%	1950-1987
St. Andrew Bay	-0.60	4706	3979	-15%	1964-1992
Tampa Bay	-0.78	30,970	11,466	-63%	1879-2006
ALABAMA					
Mobile Bay	-7.77	1,105	200	-82%	1981-2003
Perdido Bay	-2.18	475	120	-75%	1940-2003
MISSISSIPPI					
Mississippi Bays	-9.65	5,520	600	-89%	1969-1992
Mississippi Sound	-6.36	600	298	-50%	1992-2003
LOUISIANA					
Chandeleur Islands	-1.33	6,377	4,511	-29%	1969-1995
TEXAS					
Galveston Bay	-5.40	2,025	210	-90%	1956-1998
Lower Laguna Madre	-0.80	66,608	55,437	-17%	1965-1988
Upper Laguna Madre	+1.86	11,800	21,400	+81	1967-1999
Matagorda	-0.93	570	491	-14%	1971-1987
East Matagorda	+2.51	256	387	+51%	1971-1987
BERMUDA					
Bermuda Platform	-3.66	2,100	1,625	-23%	1997-2004

Oil and Gas Development

Pollution from oil and gas development is a primary threat to *H. zosterae* and its seagrass habitat. Pollution from numerous activities associated with oil and gas development threatens *H. zosterae* including offshore exploration, extraction, shipping, construction, refining, abandoned wells, leaking pipelines, and oil spills. Extensive oil and gas production takes place both in coastal waters and on the Outer Continental Shelf (OCS) in the Gulf of Mexico. The western and central Gulf of Mexico, including offshore Texas, Louisiana, Mississippi, and Alabama, is a major U.S. petroleum-producing area, accounting for 25 percent of U.S. oil production in 2007 (U.S. MMS 2009). The U.S. Minerals Management Service (MMS) (now known as the Bureau of Ocean Energy Management, Regulation, and Enforcement) predicts that by 2013 production in the Gulf will increase to 686 million barrels per year (MMS 2009). There are around 4,000 offshore oil and gas platforms and tens of thousands of miles of pipeline in the central and western Gulf of Mexico, where 90 percent of the country's offshore drilling takes place (Robertson 2010). Oil and gas development in the Gulf of Mexico has resulted in severe environmental impacts including oil spills, brine discharges, and heavy metal deposition from drilling muds and tailings (Wilkinson et al. 2009, p. 73).

In April 2010 the BP Deepwater Horizon oil spill released an estimated 205.8 million gallons (4.9 million barrels) of oil and 205,000 metric tons of methane into the Gulf of Mexico (Achenbach and Fahrenthold 2010, Graham et al. 2010). It is estimated that approximately 25 percent of the spilled oil was directly recovered or burned, leaving approximately 75 percent of the oil at sea (Graham et al. 2010). The spill resulted in more than 1,050 total linear miles of oiled shoreline, some of which were still oiled in January 2011 (NOAA 2011). Tar balls from the spill have washed up on beaches from Texas to the Atlantic Coast of Florida. The scientists at Project Seahorse (2010) caution that the BP oil spill could cause the extinction of *H. zosterae* due to direct mortality and to the devastation of the dwarf seahorse's seagrass habitat by the spill itself, by the dispersants used to breakdown the oil, and by the burning of oiled seagrass mats.

A significant portion of *H. zosterae*'s range is threatened by pollution from the spill, which covered vast areas in the Gulf. Due to pollution from the spill, NOAA closed large portions of the Gulf to fishing to ensure that seafood from the Gulf would not threaten consumers. The closures are based on computer modeling of where the oil is likely to be based on weather, satellite imagery, ocean buoy data, and ocean currents, and daily overflights to verify the extent of the oil. The closures began on May 2 and expanded throughout the summer with up to 88,522 square miles, nearly 37 percent of the federal waters in the Gulf, being closed to fishing (NOAA 2010c). Portions of the Gulf were still closed to fishing in November 2010 due to the discovery of tar balls in shrimp trawls. The closures extended east to Dry Tortugas in southwest Florida, just west of the Florida Keys, and to Panama City Beach in the Panhandle.

The dwarf seahorse is a habitat specialist that is completely dependent on healthy seagrass beds for survival, and the spill and its aftermath pose substantial risk to seagrass meadows in the Gulf (Whigham et al. 2010, p. 1). Oil pollution has severe impacts on

seagrasses which can persist for decades (Zieman 1982, p. 87, FDEP 2009b, p. 27). Impacts of oil and dispersants on seagrass range from complete mortality to sublethal stress and chronic impairment of seagrass metabolism and function (Hatcher and Larkum 1982, Thorhaug and Marcus 1987, Jackson et al. 1989, Ralph and Burchett 1998, Sandulli 1998, Peirano et al. 2005, Scarlett et al. 2005). For example, a 1986 oil spill in Panama trapped oil in sediments of intertidal beds of turtle grass, which killed the seagrass beds, and resulted in the erosion of the sediment down to the coralline rock layer (Shigenaka 2010, p. 43). Even low concentrations of oils and greases are known to significantly interfere with seagrass photosynthesis (FDEP 2009b). Seagrasses are also threatened by damage from vessels during spill response and cleanup efforts (Cosentino-Manning et al. 2010, p. 4). The burning of seagrass beds and the use of dispersants in oil spill clean-up attempts also harm seagrasses (Project Seahorse 2010). Dispersants may increase the toxicity of oil to seagrasses. Ibemesim and Bamidele (2008) found that seashore grasses treated with heavy crude oil and Corexit 9527 experienced 100 percent mortality, while plants treated with oil alone recovered (p. 875).

In addition to destroying and degrading its habitat, oil pollution threatens *H. zosterae* with direct mortality. Acute and chronic oil spills have a wide array of lethal and sublethal impacts on marine plants and animals, including immediate and long-term effects. Petroleum oil is a complex mixture of hundreds of different compounds, mostly hydrocarbons, with different levels of toxicity. Polycyclic aromatic hydrocarbons (PAHs) are among the most toxic oil components and have been documented to cause significant impacts on wildlife. Direct impacts to wildlife from exposure to oil include behavioral alteration, suppressed growth, induced or inhibited enzyme systems and other molecular effects, physiological responses, reduced immunity to disease and parasites, histopathological lesions and other cellular effects, tainted flesh, and chronic mortality (Holdway 2002). There is an increasing body of evidence that even at extremely low concentrations, oil is highly toxic to aquatic organisms (Barron et al. 2003, p. 658).

Graham et al. (2010) used carbon isotopic signals to trace the fate of the BP oil in warm, shallow shelf waters and found that fractions of the oil became suspended in the shallow water column where oil is being trophically transferred within the planktonic food web, which poses ongoing contamination risk to the dwarf seahorse. Seahorses are sensitive to contaminants, and are known to be threatened by oil pollution (Alcoforado Santos et al. 2010, p. 958). Using micronuclei tests and comet assays, Alcoforado Santos et al. (2010) found that exposure to diesel oil caused DNA damage in *H. reidi* seahorses following 96 hours of exposure to diluted water-soluble fractions. The frequency of cell death increased with the concentration of oil. Seahorses exposed to more concentrated dilutions of diesel experienced a higher frequency of cell death than those exposed to lower concentrations, but all concentrations caused DMA damage compared to controls (p. 958). Similarly, Negreiros et al. (2011) found that crude oil exposure caused DNA damage, hypertrophy, capillary dilation, hyperplasia, and epithelial damage in the seahorse *H. reidi* in the laboratory.

Exposure to crude oil adversely affects fish at all life stages (Carls et al. 1999, Bernanke and Kohler 2009). Early life stages of fish are particularly sensitive to the toxic effects of

PAHs which can cause larval deformation and death. Laboratory experiments found that PAHs from partially weathered crude oil at concentrations as low as 1 part per billion (ppb) are toxic to the developing eggs of Pacific herring (*Clupea pallasii*) and pink salmon (*Oncorhynchus gorbuscha*) (Carls et al. 1999, Heintz et al. 1999). Pacific herring embryos experienced mortality, malformations, genetic damage, decreased size, and inhibited swimming when exposed for 16 days at PAH concentrations of 0.7 ppb while concentrations of 0.4 ppb caused sublethal responses including edema and reduced growth (Carls et al. 1999). Crude oil from the Exxon Valdez spill is thought to have caused the elevated mortality of pink salmon eggs in oiled streams for at least four years after the spill (Peterson et al. 2003) and to have contributed the crash of Pacific herring populations which were exposed during the spawning season and which have yet to recover. Adult fish exposed to oil can suffer from reduced growth, enlarged liver, changes in heart and respiration rates, fin erosion, and reproductive impairment (Bermanke and Kohler 2009, USFWS 2010b). Additionally, fish are at risk from lethal coating of their gills with oil.

Oil pollution also threatens the dwarf seahorse by reducing and contaminating its invertebrate food sources. Oil affects virtually all invertebrate taxa (Suchanek 1993). It is toxic to bottom-dwelling, pelagic and intertidal invertebrates including the zooplankton on which *H. zosterae* feed (USFWS 2010b, Peterson et al. 1996, Table 1). Widespread mortality of marine invertebrates generally occurs in the immediate vicinity of oil spills due to chemical toxicity and smothering, and additional mortality can result when toxic components of oil are remobilized from sediments (Suchanek 1993, Peterson et al. 1996, Peterson et al. 2003). Sublethal effects to invertebrates from oil exposure include impairment of reproduction, growth, respiration, excretion, chemoreception, feeding, movements, stimulus response and disease resistance (Suchanek 1993). Due to these chronic impacts, invertebrate populations and community structure can take years to decades to recover after oil exposure (Suchanek 1993).

Seahorses, seahorse habitat, and seahorse prey are also threatened by the dispersants which were widely used in an effort to prevent the BP oil spill from reaching shore. As of August 6, approximately 1.84 million gallons of total dispersant had been applied in the Gulf—1.07 million gallons on the surface and 771,000 gallons sub-sea (Nelson 2010). Dispersants do not reduce the total amount of spilled oil, they merely change its fate in the environment. By promoting dispersion of oil into the water column, dispersants increase the potential exposure of water-column and benthic biota to spilled oil. Dispersants and dispersed oil have been shown to have significant negative impacts on marine life, including plankton and fish. Dispersants can bioaccumulate and persist in sediment and in the water column where they could impact larvae and other plankton for decades (Marine Conservation Biology Institute 2010).

Studies have shown that dispersants create a toxic environment for fish by releasing harmful oil break-down products into the water. Dispersed oil has been shown to be toxic to fish at all life stages, from eggs to larval fish to adults, according to numerous laboratory studies that have tested a variety of species (Khan and Payne 2005, Anderson et al. 2009). Monitoring data have indicated that the use of the Corexit dispersants killed

up to 25 percent of all organisms living 500 feet below the surface in areas where the dispersant was used (Farren and Blackburn 2010).

Dispersants release toxic break-down products from oil that, alone or in combination with oil droplets and dispersant chemicals, can make dispersed oil more harmful to marine life than untreated oil. Bhattacharyya et al. (2003) found that crude oil, diesel, and dispersants are all toxic to marsh microorganisms (*Chironomus tentans*, *Daphnia pulex*, and *Oryzias latipes*), and that cleaners and dispersants enhance the toxicity of the oil to the microorganisms (p. 205). Barron et al. (2003) found that when exposed to sunlight, oil treated with Corexit 9527 was significantly more toxic to early life stages of Pacific herring than oil alone. Mortality occurred more rapidly in the presence of dispersant, which increased the bioavailability of the toxic oil compounds (p. 656). Anderson et al. (2009) found that crude oil treated with the dispersant Corexit 9500 was more toxic to topsmelt (*Atherinops affinis*) embryos than untreated oil, causing reduced survival and cardiovascular and other abnormalities (p. 1058). McIntosh et al. (2010) found that early life stages of Atlantic herring (*Clupea harengus*), including free-swimming embryos, were sensitive to realistic exposures of weathered oil treated with dispersant, and conclude that early life stages of other fish species are also at risk from dispersed oil (p. 1166).

The use of dispersants underwater, as permitted by the Environmental Protection Agency in the Deepwater Horizon spill response, can result in the formation of massive deepwater oil plumes extending many miles from the spill site (NOAA 2010). The negative ecological effects of these plumes are unlikely to remain isolated to deep water habitats, and have the potential to harm the entire ecosystem from the bottom up. In addition, significant reductions in dissolved oxygen have been reported in the vicinity of underwater dispersed oil plumes, resulting from oil digestion by microbes (Winerman 2010). The northern Gulf of Mexico already experiences the annual formation of a hypoxic “dead zone”—a large area of oxygen-poor conditions that can barely support life—and the oxygen deprivation caused by the deep-sea oil plumes could increase the size of this zone (NOAA 2010b).

In addition to the ever-present threat of catastrophic spills, smaller-scale oil spills and leaks are commonplace in the Gulf. At least several spills occur annually due to blowouts, hurricanes, lax pipeline maintenance, tanker leaks and human error (Mufson 2010). More than 320 known spills involving offshore drilling have occurred in the Gulf since 1964, releasing more than 550,000 barrels of oil and drilling-related substances into the ocean (Robertson 2010). There are numerous examples of oil spills causing degradation of habitat in the Gulf. In 1967, 160,638 barrels of oil were spilled in the Gulf when an anchor tore a hole in a corroded pipeline operated by Humble Oil, a unit of Exxon. In 1969, 80,000 barrels were spilled when a Union Oil well blew out. In 1974, a Pennzoil pipeline was ruptured by a dragging anchor, spewing 19,833 barrels of oil into the Gulf. In 1979 the Ixtoc I exploratory well blew out in the western Gulf of Mexico and released an estimated 113 to 300 million gallons of oil into the western Gulf, oiling 150 miles of shoreline in Texas, and an unknown area of Mexican shoreline (Bowman 2010). Small remnants of Ixtoc tar mats persist to the present. In 1988 an anchor ruptured an

Amoco pipeline, spilling 15,576 barrels of oil. In 1990 a Shell pipeline leak dumped 14,423 barrels into the Gulf (Mufson 2010). In 2005, 16,302 barrels of oil were spilled into the Gulf due to damage caused by Hurricanes Katrina and Rita, which resulted in five spills of 1,000-2,000 barrels each, 125 smaller spills, and more than 250 “oil-related pollution incidents” (Bowman 2010, Mufson 2010). From 2006 through 2009, there were 33 spills of more than 50 barrels each in the Gulf (Mufson 2010).

The actual amount of oil polluting Gulf waters is underreported. There are several instances of official statistics understating the actual quantities of oil that have spilled into the Gulf, with reported quantities as low as one-tenth the actual amount of oil which spilled (Mufson 2010).

As well as inevitable spills, seepage from abandoned oil wells is an ongoing source of oil contamination. On March 23, 2011 emulsified oil washed onto Louisiana’s shores when mile-long slicks emanated from an abandoned well owned by Anglo-Suisse Offshore Partners that had been partially-plugged due to damage caused by Hurricane Katrina (Hammer 2011). There are approximately 27,000 abandoned wells in the Gulf which are not checked for leaks, and seepage from abandoned wells is common (Wray 2010). The Obama administration has ordered the oil industry to plug only 3,500 (13 percent) of these abandoned wells (Associated Press 2010).

There are natural oil seepages in the Gulf of Mexico, and the Gulf is accustomed to assimilating small, slow, chronic releases of oil. Chronic natural seepages, however, are vastly different than acute, catastrophic spills, which have significant long-term impacts. Persistent pollution from lingering, submerged oil and resultant behavioral changes could affect wildlife populations for years (Peterson et al. 2003). The Gulf of Mexico may also be becoming less resilient to oil pollution due the variety of environmental impacts the region now faces (Kumpf et al. 1999).

Pollution from oil transport also threatens the seahorse. Spills and leaks from oil tankers and other vessels are even more common than spills from oil platforms (FWS 2009). Up to 65 percent of imported crude oil is transported through the Gulf, totaling 1.7 billion barrels of imported crude oil in 1995, with approximately the same volume of petroleum products being exported through Gulf waters (FWS 2007). Spills and accidents involving oil-transport vessels and other vessels are common and threaten the seahorse throughout its range (Zieman 1982, FWS et al. 2004, Shields 2002, MSNBC 2010). In 1973 an oil tanker ran aground on a reef in Puerto Rico and 1.58 million gallons of oil was spilled, killing organisms in seagrass beds and sending oil into the mangrove forests (Hoff et al. 2010, p. 56). In 1979 two fully-loaded tankers collided in Galveston Bay releasing an estimated 2.6 million gallons of oil into the Gulf which travelled more than 200 miles and impacted marshes on Padre Island and Matagorda Bay. In 1980 another tanker accident released 5.1 million gallons of oil near Galveston. Also in 1980, a tanker ran aground in Louisiana and spilled 2.7 million gallons of crude into the Gulf. In 1993 three vessels collided in Tampa Bay spilling 336,000 gallons of fuel oil and coating seagrass beds and mangrove roots (Hoff et al. 2010). For more than a year after the Tampa spill, unrecovered submerged and buried oil chronically oiled beaches in the Tampa area

during storm events (Shigenaka 2010). Seven years after the spill, buried oil was remobilized during dredging operations and tarballs coated some shorelines (Shigenaka 2010). Tanker discharges also threaten the seahorse. Van Vleet and Pauly (1987) chemically analyzed tar found in or on stranded sea turtles collected along the Florida coast and throughout the Gulf of Mexico and found that the tar had originated from crude oil tanker discharges. Sunken vessels may also cause oil pollution, sometimes for decades (Hampton et al. 2003).

Oil pollution threatens the dwarf seahorse not only in the Gulf of Mexico but also in the Caribbean and on the Atlantic Coast of Florida. Though offshore drilling is not permitted in Florida, the Atlantic Coast remains threatened by pollution from oil transport, and by oil spills washing into the area on currents. In 2000, oil pollution from a spill of unknown origin washed up on Florida beaches from North Miami to Pompano, north of Fort Lauderdale, including submerged sticky oil mats mixed with seagrass (Shigenaka 2010, p. 79). Tarballs not associated with any particular oil spill are frequently documented in Florida (Carr 1987, Witherington 1994, 2002, Shigenaka 2010).

Offshore oil and gas development in Cuba poses another threat to the seahorse's habitat. Cuba has three producing offshore oil fields within 5 km of its north coast opposite Florida (Cervera 2004). Several countries have already purchased or are in the process of purchasing oil and gas leases off the coast of Cuba, some as close as 50 miles from Florida, including Russia, Norway, India, Venezuela, Vietnam, Brazil, and Spain (Padgett 2008, BBC 2009, Kramer 2010). Cuba does not have the underwater robots and spare drilling rigs required to contain a big spill, and the trade embargo could hinder assistance from U.S. companies if a spill were to occur (Kramer 2010). An oil spill on Cuba's north coast could reach the Florida Keys in three days (Ibid.)

The development of liquefied natural gas (LNG) facilities poses an additional threat. The proposed use of "open-loop," or once-through, systems for LNG re-gasification poses several threats to the seahorse due to pollution, large volume of water intake, generation of thermal plumes, discharge of treated water, and increased turbidity (Wilkinson et al. 2009, p. 73).

There is no question that the dwarf seahorse is threatened in a significant portion of its range by ongoing pollution from the 2010 BP oil spill, by the threat of future spills, and by commonplace smaller spills and leaks. Toxic subsurface oil from the BP spill will persist for decades, leading to chronic exposure and sublethal negative effects on the dwarf seahorse, its seagrass habitat, and its prey (Peterson et al. 2003). There is a growing body of literature which suggests that oil spills have the potential to negatively affect wildlife for much longer time frames than is commonly assumed (Esler et al. 2010). Esler et al. (2010), for example, found that harlequin ducks in Prince William Sound continue to be exposed to residual oil from the 1998 Exxon Valdez spill, 21 years after the spill.

As long as oil is drilled and transported in its habitat, oil pollution will continue to threaten the dwarf seahorse. Unfortunately, the seagrass beds which support the dwarf seahorse are threatened not only by oil pollution, but also by numerous additional threats.

Loss and Degradation of Mangrove Habitat

In some areas seagrass beds occur in close association with mangroves, with mangroves protecting seagrass beds by trapping sediments and stabilizing shorelines (Pauly and Ingles 1999, Hoff et al. 2010, p.16). Odum et al. (1982) report *H. zosterae* as occurring in mangrove areas in Florida, in association with seagrasses and algae in estuarine and oceanic bays (p. 116). To the extent that seagrass beds are negatively affected by the loss of mangroves, or that mangroves provide direct habitat value for the seahorse, loss and degradation of mangrove habitat also threatens *H. zosterae*.

Rapid population growth in south Florida has caused mangrove areas to be dredged and filled for development (FWS 2010, p. 41, FDEP 2010, p. 18). Though mangroves are now protected, illegal dredge and fill practices are ongoing (FDEP 2009). Before state laws were established to protect mangroves, they were commonly removed from shorelines because they blocked views and harbored mosquitoes (FDEP 2010). Unauthorized removal of mangroves continues to be pervasive in order to gain access to water and to improve views (FDEP 2009, p. 54). Clearing of vegetation causes erosion and colonization by invasive exotic plants, both of which diminish fisheries habitat value (FDEP 2009). Invasive Brazilian pepper plants, for example, are now a serious threat to mangrove communities and the organisms which depend on them (FWS 2010, p. 49). Mangroves have been replaced by seawalls and riprap, resulting in increased turbidity and reduced light penetration, which directly threatens the seagrasses on which *H. zosterae* depends.

Numerous areas of Florida have lost a substantial percentage of their original mangrove ecosystems. Naples Bay has lost 70 percent of its original mangrove fringe (FDEP 2010, p. 18). The lower east coast of Florida, Tampa Bay, the Florida Keys, and Marco Island have also suffered substantial mangrove loss (Odum et al. 1982, p. 80).

Water pollution is causing ongoing loss and degradation of mangrove ecosystems. Mangroves have aerial roots which are vulnerable to flooding and becoming coated with fine sediments, which can be lethal (Odum et al. 1982, p. 77). Mangroves are “unusually susceptible to herbicides” (Odum et al. 1982), and herbicides are widely used in agricultural and urban areas in Florida (FWS 2010). Water pollution from dredging, development, sewage effluent, oil spills, and other sources poses an ongoing threat to mangroves. Large hurricanes also cause extensive damage to mangroves (Odum et al. 1982). Hurricane Katrina scoured and damaged mangrove roots, transforming previous mangrove areas into open water (Klemas 2009, p. 1273). Rapid sea level rise resulting from global climate change also threatens low-lying mangrove communities and the species dependent on them (FWS 2010, p. 49).

Like seagrasses, mangroves are also threatened by oil pollution. Oil spills are “extremely harmful to mangroves” (Odum et al. 1982, p. 80). Petroleum and petroleum byproducts kill and injure mangroves in a variety of ways. Oil coats the roots, rhizomes, and pneumatophores and disrupts oxygen transport to underground roots. Petroleum is readily absorbed by lipophilic substances on the surfaces of mangroves, which leads to severe metabolic alterations. Damage from oil spills follows a predictable pattern which may require years to unfold, with tree death occurring weeks to years after the spill. Sublethal damage includes partial defoliation and necrosis of pneumatophores. Dispersants commonly used to combat oil spills are also toxic to vascular plants. Damage during clean-up from trampling and compaction also threatens mangroves (Odum et al. 1982). Tropical and subtropical mangroves seem especially vulnerable to oil spills, as was demonstrated along the Persian Gulf following the Gulf War and in Panama following a major spill in 1986 (Garrity and Levings 1993, Burns et al. 1994, Proffitt et al. 1995, Whigham et al. 2010). Mangroves grow in low-energy depositional areas, which also tend to be the sites where oil accumulates (Hoff et al. 2010, p. 38). Soil conditions in mangrove areas inhibit oil breakdown, and heavier oils can persist in mangrove sediment for decades after a spill (Hoff et al. 2010, p. 39).

FWS (2010) lists numerous and ongoing threats to mangroves, including: “altered hydrologic regime, habitat destruction, altered structure, altered water quality, altered weather regime and sea level rise, altered species composition, habitat disturbance, and habitat fragmentation predominantly from coastal development; roads, bridges, and causeways; harmful algal blooms; incompatible industrial operations; invasive plants; shoreline hardening; invasive animals; incompatible releases of water (including water quality, quantity, and timing); incompatible wildlife and fisheries management strategies; climate variability; parasites and pathogens; channel modification; incompatible aquaculture operations; and pollution and nutrient loading” (p. 70).

Declining Water Quality

Declining water quality due to a variety of factors threatens the near-shore and estuarine habitat on which *H. zosterae* depends for survival. The loss and degradation of estuarine wetlands in the range of *H. zosterae* is well documented. The National Coastal Condition Report II (EPA 2004) reports that the nation’s estuaries are declining and continue to be at risk, with 28 percent of estuarine waters considered to be impaired for aquatic life use. The report ranks 60 percent of the nation’s estuarine waters as moderately to highly degraded based on decreased dissolved oxygen content, increased nutrient and chlorophyll *a* concentrations, and decreased water clarity. Decreased water clarity threatens *H. zosterae* directly by interfering with the visual cues of mating behavior and by threatening seagrass habitat due to decreased photosynthesis. Increased nutrient input causes ecosystem changes which cause seagrass, which is dominant in oligotrophic conditions, to be replaced with faster-growing primary producers such as algae. As nutrient availability increases, macroalgae outcompetes seagrass for light. If nutrient availability continues to increase, microalgae outcompetes the macroalgae, and the microalgal blooms deprive seagrass of light, leading to loss of seagrass meadows (Fourqurean 2009, p. 3).

Algal blooms also deprive water of oxygen, forming hypoxic “dead zones” which threaten aquatic life. In the northern Gulf of Mexico, nutrient loading from the Mississippi River causes a dead zone each spring and summer which in some years extends west from the mouth of the Mississippi River past the Texas border (Lellis-Dibble et al. 2008). From 1985 to 1992, the region averaged 3,205 square miles of habitat affected by hypoxia, and from 1993 to 2001, it averaged 16,178 square miles (EPA 2004). Hypoxic conditions threaten *H. zosterae* with direct mortality, and also threaten the seagrass on which it depends for survival, as persistently low dissolved oxygen levels can be lethal to seagrasses (Zieman 1982, p. 86). Hypoxic conditions also exacerbate the physiological damage caused to seahorses by exposure to oil pollution (Negreiros et al. 2011).

Brown and red tide events further threaten *H. zosterae*. The large seagrass die-off and algal bloom in Florida Bay which began in the 1980s was coincident with declining water quality conditions throughout the Gulf of Mexico, including a brown tide event which affected seagrass beds in Laguna Madre, Texas, on the western edge of the Gulf (Fourqurean and Robblee 1999). Red tides pose another threat to *H. zosterae*. Red tides occur in the Gulf of Mexico almost every summer or fall, generally off the central and southwestern coasts of Florida. The Florida red tide organism (*Karenia brevis*) produces a toxin that can be fatal to marine animals. Generally red tide blooms persist for 3 to 5 months and affect hundreds of square miles, but occasionally blooms may continue for up to 18 months and may affect thousands of square miles (FWS 2010). In 2005, an extended and intense red tide impacted the west coast of Florida from Tampa to Fort Meyers (FWS 2010).

Toxins and contaminants also threaten *H. zosterae*. Levels of contaminants from non-point sources such as stormwater runoff and septic systems are increasing in the northern Gulf of Mexico (Beck et al. 2000). Anderson et al. (2005) and the National Park Service (NPS 1996) report that elevated metal concentrations, impacts from several Superfund sites, landfills, and septic systems all threaten aquatic habitats and species at Timucuan Ecological and Historic Preserve, where *H. zosterae* occurs (NPS 1996, p. 62). At John Pennekamp Coral Reef State Park, where *H. zosterae* occurs (FDEP 2004b, p. A 4-19), numerous dredged canals adjacent to the park trap pollutants and create chronic water quality problems. In addition, storm events can cause an overturn of the water column in the stagnant canals, which brings oxygen-poor waters and benthic pollutants to the surface, resulting in “massive fish kills” and reduced water quality within the park’s boundaries (FDEP 2004b, p. 25).

There is extensive evidence of mercury contamination in fish in South Florida ecosystems, with atmospheric mercury deposition having increased approximately fivefold since 1900 (FWS 2010). Excessive concentrations of mercury have been found in all of Florida’s coastal waters (FWS 2010).

Pesticides are widely used in agricultural and urban areas in South Florida with pesticide usage per acre ranking in the top five in the nation (FWS 2010). Since the late 1960s,

persistent organochlorine pesticides have been detected in fish in the Everglades. Even pesticides which are no longer in use, such as DDD, DDE, DDT, Dieldrin, and Heptachlor, are detected in surface waters throughout south Florida. Chlorinated chemicals, such as polychlorinated biphenyls (PCBs), dioxins, and furans also pose serious concerns to fish (FWS 2010, p. 62-64).

The loss of coastal and marine wetlands further threatens water quality in the range of *H. zosterae*. EPA (2004) ranks the overall national coastal habitat condition as poor, based on long-term wetland loss rates. Estuarine and marine wetlands are being lost at a rate of approximately 4,740 acres annually, and this rate is accelerating (EPA 2004). Stedman and Dahl (2008) report that from 1998 to 2004, 59,000 acres of coastal wetlands were lost annually. Since 1940, Louisiana alone has lost more than 1.22 million acres of coastal wetlands. Even the loss of freshwater wetlands in coastal areas negatively affects the habitat of *H. zosterae* because freshwater wetlands are important components of coastal and estuarine ecosystems, supplying nutrients and providing floodwater control (Lellis-Dibble et al. 2008, p. 46). Causes of wetland loss include coastal development, dredging, water control activities, commercial and recreational boat traffic, agriculture, waste disposal, shipping, oil and gas development, hurricanes, subsidence, and rising sea levels (EPA 2004). Hurricane Katrina destroyed nearly 100 square miles of marsh, transforming it into open water and resulting in permanent loss of large areas of wetland (Klemas 2009, p. 1273). Rising sea-levels are expected to cause Louisiana to lose an additional 448,000 wetland acres in the next 50 years (Lellis-Dibble et al. 2008). The current stresses on Gulf coastal wetlands from land subsidence (Richardson and Pahl 2006) and sea-level rise (Stumpf and Haines 1998) have been greatly exacerbated by pollution from the 2010 BP oil spill (Whigham et al. 2010).

Seagrasses and dwarf seahorses are also threatened by water pollution from shipping lanes (Zieman 1982, FDEP 2004b). A quarter of U.S. commercial shipping passes through the Straits of Florida (Wilkinson et al. 2009, p. 73). This vast amount of shipping traffic increases the risk of oil spills, and causes chronic oil pollution in small amounts (Zieman 1982, p. 88). A considerable amount of tar regularly washes up on Florida beaches, with the beaches of the Florida Keys among the most contaminated (Zieman 1982, p. 88).

Water pollution and other impacts from aquacultural operations pose a potential threat to *H. zosterae*. The shallow areas preferred for aquacultural operations are the same areas inhabited by *H. zosterae*. Areas within the boundaries of aquatic preserves are being opened to aquacultural activities; for example, there are new aquacultural leases in Terra Ceia Aquatic Preserve. Aquaculture can have direct impacts on aquatic vegetation, reduce water quality, and introduce non-native species (FDEP 2009, p. 60).

Other factors which degrade water quality in the range of the dwarf seahorse include pipeline installation, pier construction, fishing practices, chemical spills, and phosphate mining discharges (Zieman 1982, Lellis-Dibble et al. 2008, Wilkinson et al. 2009, FWS 2010).

Development and Human Population Growth

The Monterey Bay Aquarium (2011) cites coastal population pressure and the accompanying dredging, pollution and development as a primary threat to the habitat of *H. zosterae*. An estimated 53 percent of the current U.S. population lives in coastal counties, creating tremendous stress on coastal resources (Lellis-Dibble et al. 2008). Brady and Goebel (2002) attribute 66 percent of wetland loss in coastal counties to development. Zieman (1982) attributes the greatest amount of destruction of seagrasses in south Florida from 1950-1980 to dredging and filling for construction and navigation to support booming development (p. 84). The Florida Department of Environmental Protection (2010) reports that the state's southwest estuaries continue to be threatened by urban development, channelization and drainage, and altered hydrologic and salinity regimes which contribute to the loss of seagrass beds (p. ii). Even protected areas are threatened by water pollution resulting from development adjacent to their boundaries. For example, dredging outside the preserve poses an ongoing threat to seagrass habitat in Terra Ceia Aquatic Preserve, due to the transport of fine sediments (FDEP 2009a). Coral Reef State Park (FDEP 2004b) identifies rapid development adjacent to park boundaries as a major problem, due to the destruction of natural communities, habitat fragmentation, pollution from stormwater runoff and sewage effluent, spread of invasive species, and ongoing dredging of shallow areas for boat use (p. 24).

FWS (2010) reports that the “numerous threats and various challenges” faced by refuges in the J.N. “Ding” Darling National Wildlife Refuge Complex are all related to the rapid growth of the human population and development of the landscape, including hydrologic alterations, water quality degradation, and habitat loss (p. 41, 289). Concerning threats to the estuarine ecosystem in southern and southwestern Florida, the Service states:

“Coastal southwest Florida is one of the fastest urbanizing regions in the U.S. Rapid urban development that has already occurred has radically changed the character and ecology of coastal waters. Mangroves have been removed or cut back, red tide events cause public health warnings, seagrass areas have declined or been damaged, and groundwater pumping has reached its maximum limit” (FWS 2010, p. 41).

In addition to direct habitat loss, dredging, and pollution, upstream development threatens the health of estuaries due to water management activities. Reduced freshwater inflows interfere with temperature and salinity regimes, threatening seagrasses and seahorses (Wilkinson et al. 2009, p. 73). Reduced freshwater flows into Florida Bay have contributed to both seagrass die-off and to fish-kills (U.S. DOC et al. 2007, p. 17).

Improper wastewater treatment also threatens the habitat of *H. zosterae*. Since 1999 a number of beach health advisories have been posted throughout the Keys due to pathogens associated with poor wastewater treatment (FDEP 2004b, p. 25). Municipal wastewater discharges and aging septic systems are known to be negatively affecting aquatic habitats at Mosquito Lagoon Aquatic Preserve (FDEP 2009b, p. 59, 75).

Human population growth and the resultant development, recreation, and water quality degradation will continue to cause the loss of seagrass habitat into the future (FWS 2010).

Damage from Vessels

Boat recreation and tourism are significant threats to the seagrass beds on which *H. zosterae* depends for survival. Seagrasses are susceptible to scarring from boat propellers, anchors and trawls. Seagrasses typically grow linearly along the shoreline, and can be fragmented by dock construction and boat traffic, which inhibit vegetative recolonization (FDEP 2009b). Beck et al. (2000) report that nearly every shallow seagrass habitat in the northern Gulf of Mexico has been degraded by damage from boats. Likewise, Hall (2010) reports that nearly all of Florida's seagrass beds show damage caused by boat propellers and hull groundings. Damage from boat propellers leads to loss of primary productivity, increased sedimentation and nutrient loading, and light reduction, which decreases water quality and contributes to further loss of seagrass (Gudeman 2010). In St. Andrews Bay, more than 50 percent of seagrass beds have been scarred by boat propellers (Sargent et al. 1995). Vessel damage is a major source of habitat destruction in the Florida Keys (Hall 2010). The U.S. Department of Commerce (DOC) et al. (2007) report that boat propellers and large ships have caused severe to moderate damage to more than 30,000 acres of seagrasses in the Florida Keys National Marine Sanctuary (U.S. DOC et al. 2007, p. 17). Sargent et al. (1995) estimate that statewide, seven percent of Florida's nearshore seagrass has been scarred by propellers. They found that the greatest acreage of moderate and severe scarring occurred in areas with denser human populations and more registered boats including the Florida Keys (Monroe and Dade counties), Tampa Bay (Hillsborough, Manatee and Pinellas counties), Charlotte Harbor (Lee County), and the north Indian River Lagoon (Brevard and Volusia counties). Within Cockroach Bay Preserve State Park, there is an excessive amount of propeller scar damage (FDEP 2004, p. 12). At John Pennekamp Coral Reef State Park, boat groundings occur frequently due to excessive visitor use and heavy boat traffic, and the majority of the shallow seagrass flats adjacent to navigational channels are in poor condition due to propeller scars and turbidity plumes (FDEP 2004b, p. 18, 24). Propeller scarring and bank erosion caused by boat wakes tears seagrass from the substrate, which leaves sediment exposed and easily re-suspended into the water column, inhibiting seagrass recovery (FDEP 2004b, p. 10).

Seagrass recovery from vessel damage in subtropical seagrass systems can take years to decades (Hall 2010). Prop scars revegetate very slowly and may actually increase in size with time if they erode (FDEP 2004b, p. 18). Zieman (1982) reports that even small cuts from boat propellers can take two to five years or more to heal (p. 91). In low-energy areas which lack coarse sediment, vessel grounding sites and propeller scars can fill with muck and prolong recovery (FDEP 2009, p. 37). Degraded water quality and ongoing disturbance further hinder the ability of seagrass to recover (FDEP 2009b). Grablow (2008) compared methods of seagrass restoration and concluded that transplantation and filling of scarring were unsuccessful, and that efforts should be made to prevent further seagrass destruction.

Trawling

Shrimp trawl fisheries negatively impact aquatic habitats in the Gulf of Mexico and Latin America (Wilkinson et al. 2009, p. 73). Commercial trawling damages seahorse habitat and negatively affects seahorse populations (Watling and Norse 1998, Baum and Vincent 2005, p. 317). Trawling damages habitat directly by scraping, and indirectly by causing loss of filter-feeding shellfish species, the loss of which has system level effects on water clarity (Beck et al. 2000, p. 13). Bottom-fishing gear causes benthic habitat degradation by reducing habitat complexity and removing emergent epifauna, smoothing sedimentary bedforms, and removing structure-forming species such as corals and sponges (Baum et al. 2003, p. 729). Roller beam trawling redistributes macroalgae and seagrasses, which directly impacts seahorses (Meyer et al. 1999). Baum et al. (2003) estimate that seagrasses comprise more than half of the volume of catch for trawling operations. Roller beam trawls are assumed to have low impacts on seagrass habitats, but there are no data to support this assumption, as the effects of long-term repetitive trawling have not been tested (Baum et al. 2003). Curtis et al. (2007) found that even repeated use of light demersal fishing gear can cause habitat alteration (p. 479). In addition to damaging seagrass habitats, trawling can result in direct take of seahorses (Curtis et al. 2007, p. 469).

Global Climate Change

Global climate change directly threatens the seagrass habitat on which *H. zosterae* depends for survival (Twilley et al. 2001, FDEP 2009b, Veron et al. 2009, FWS 2010, Hoegh-Guldberg 2010). Rising sea levels due to climate change will result in declines in seagrass and mangrove communities (Twilley et al. 2001). Rising ocean temperatures, increasing ocean acidification, and coral bleaching are also expected to lead to domino effects that will contribute to the deterioration of seagrass and mangrove ecosystems (Veron et al. 2009). Because seagrasses are adapted to specific salinity regimes, they are threatened both by decreased freshwater input into bays and estuaries due to increasing drought, and by increased freshwater runoff from extreme precipitation events (Fourqurean and Robblee 1999, p. 345, FDEP 2009b, p. 23). Increased frequency and intensity of severe coastal weather events also threatens seagrasses. Hurricane Katrina, for example, uprooted and destroyed entire seagrass beds (Klemas 2009, p. 1273). Severe freezes can also be lethal to seagrasses and to mangroves (FDEP 2009b, p. 23). Though seagrasses have the natural ability to recover from hurricanes and severe freezes, reduced water quality and ongoing disturbance inhibit their recovery (FDEP 2009b, p. 27). Climate change will also likely lead to increased density and diversity of exotic and invasive species (FWS 2010, p. 44, Joel Fodrie et al. 2010). Climate change related stressors will also enhance the negative impacts of other stressors, exacerbating threats such as decreased water quality, altered water quantity and timing of flows, increased pollution, and increased algal blooms and red tide events (FWS 2010, p. 44).

In sum, the dwarf seahorse is threatened throughout its range by ongoing loss of seagrasses, declining water quality, and global climate change. Much of the seahorse's habitat had been lost even before the BP Deepwater Horizon catastrophe. The oil spill

and its aftermath could extirpate the seahorse in a significant portion of its range. Human population growth and increasing development, recreation, and pollution will continue to degrade the seagrasses on which the seahorse depends.

OVERUTILIZATION FOR COMMERCIAL, RECREATIONAL, SCIENTIFIC OR EDUCATIONAL PURPOSES

Overview

Globally, the commercial trade of seahorses is threatening wild populations with depletion (Vincent 1995a, p. 127). Numerous populations are collapsing due to exploitation (Vincent 1995a, p. 128). Seahorses are exploited for the aquarium trade, for use as trinkets and curiosities, and for use in traditional medicines and tonic foods (Lourie et al. 2004, Foster and Vincent 2005, p. 1045). As of 2001 nearly 80 countries were involved in the seahorse trade (Koldewey and Martin-Smith 2010, p. 133). Most seahorses in international trade are caught as bycatch in trawl fisheries, though there are also many targeted seahorse fisheries (Baum et al. 2003).

Seahorse populations are vulnerable to overexploitation due to life history factors such as sparse distribution, low population densities, low mobility, small home range sizes, slow re-colonization potential, highly structured social and reproductive behavior, and low rates of population increase (Foster and Vincent 2004, p. 50, Lourie et al. 2004, p. 6). Due to these traits, seahorse populations can decline rapidly. Estimated population declines of between 15 and 50 percent over five-year periods due to trade alone are common (Lourie et al. 2004, p. 3).

The dwarf seahorse is exploited for the aquarium trade, for use in curios (Bruckner et al. 2005, p. 98), and potentially for use in traditional medicine (Vincent 1995a, 1995b, Lourie et al. 2004). The dwarf seahorse is known to be exploited in Florida, and is potentially exploited in Mexico and the Caribbean as well (Bruckner et al. 2005). In Florida, harvest of live *H. zosterae* varies widely and has ranged from more than 2,000 to nearly 100,000 individuals per year (Table 2). The vast majority of *H. zosterae* are landed in southwest Florida, and a substantial number are also landed in the southeast Florida/Florida Keys region (Bruckner et al. 2005). Most of the Florida seahorse harvest is conducted by divers using nets to target seahorses or by fishers using trawls in a live, bait-shrimp fishery where seahorses are taken as bycatch. Some specimens are also harvested by seine or dredge. From 1990 to 2003, divers collected from 18 to 90 percent of the annual harvest of *Hippocampus zosterae*, and trawlers collected from 0 to 60 percent. Overall, divers collected approximately 90 percent of the *H. zosterae* harvested in the southeast Florida/Florida Keys region and slightly more than 50 percent of those harvested in southwest Florida. (Bruckner et al. 2005, p. 97).

The United States exports live and dried seahorses. The number of exported seahorses prior to 2004 is unknown because prior to this date, seahorses were coded as “tropical fish” by customs officials (Bruckner et al. 2005). From 1996 to 2003, the U.S. Fish and Wildlife Service recorded 33 records of seahorse export shipments, consisting of nearly

1,000 live animals and thousands of dried specimens for either the curio or medicinal trade. This value probably largely underestimates the U.S. trade volume because of the “tropical fish” coding prior to the listing of all seahorse species under the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) in 2004, which requires exports to be coded to the species level (Bruckner et al. 2005). From 1983-2000, Chinese Taipei alone reported approximately 500 kg of dried seahorse imports from the United States (Bruckner et al. 2005, p. 100). Seahorses are also exported live to Europe and other countries for the aquarium trade. Based on lack of record keeping and discrepancies between custom’s records among countries, much of the seahorse trade is unreported (Baum and Vincent 2005, p. 318).

The Caribbean and Mexico

The dwarf seahorse is also exported from the Caribbean and from Mexico. Seahorses are one of the most numerous fishes exported from the Caribbean (Bruckner 2005, p. 127). The Caribbean currently supplies a small percentage of the total global trade in marine ornamental species, but ornamental fisheries in the region represent an important emerging industry. The ornamental trade in the Caribbean increased substantially between 1992 and 2000, and is expected to continue to increase (Bruckner 2005). The take of seahorses for the aquarium trade is prohibited in the U.S. Virgin Islands and Puerto Rico.

Mexico is one of the leading exporters of dried seahorses (Baum and Vincent 2005), which threatens *H. zosterae* on the Gulf Coast of Mexico. Dried seahorses exported from Mexico, and most of those traded domestically, are caught incidentally in the country’s commercial shrimp trawl fisheries. It is only legal to trade incidentally caught seahorses in Mexico, though illegal targeting of seahorses still occurs (Baum and Vincent 2005). Official records from Hong Kong, mainland China, and the United States all show imports of dried seahorses purportedly originating in Mexico (Baum and Vincent 2005). U.S. customs records only documented imports of 80 live seahorses from Mexico to the United States in 2003, but this number is likely a gross underestimate of the actual trade, as wholesalers in Mexico admitted to interviewers that they exported much greater numbers (Baum and Vincent 2005). Live seahorses are also traded in substantial numbers within Mexico. In the year 2000, from 8,200 to 14,600 live seahorses were traded on the domestic market in Mexico.

Shrimp trawl fishers in Mexico have noted drastically reduced catch of seahorses over time, with estimated declines of greater than 90 percent in catch per unit effort in the Gulf of Mexico (Baum and Vincent 2005, p. 316). Targeted seahorse fisheries in Mexico have also reported declines exceeding 50 percent (Baum and Vincent 2005, p. 316).

Aquarium Trade

The dwarf seahorse is threatened by collection for display in aquariums. On average, approximately 49,000 dwarf seahorses are harvested from Florida each year for the aquarium trade (Table 2). The trade in marine ornamental fish began in the 1930s in Sri

Lanka, spread to Hawaii and the Philippines in the 1950s, and became a multi-million dollar industry by the 1970s with fisheries established throughout the tropical Atlantic, Pacific, and Indian Oceans (Bruckner 2005). Globally, the keeping of aquaria is one of the most popular hobbies, and is now purported to be the second most popular hobby in the United States (Adams et al. 2001, Job et al. 2006). The United States is the largest single market for aquarium fish, with large markets also existing in the U.K., Australia, and a growing number of other countries (Job et al. 2006). Approximately 10 percent of all aquaria are believed to be marine (Job et al. 2006).

Table 2. Florida Live Seahorse Harvest 1991-2003. Data from CITES 2002, Larkin and Adams 2003, Bruckner et al. 2005.

Year	Seahorses caught (all species)	<i>H. zosterae</i> caught	Value <i>H. zosterae</i>
1991	13,982	7,226	\$61,102
1992	83,715	76,706	\$46,109
1993	71,815	66,440	\$65,632
1994	110,948	98,779	\$69,079
1995	23,341	22,662	\$30,231
1996	19,037	~19,000	
1997	90,049	~85,000	
1998	16,977	~16,000	
1999		~61,000	
2000		~16,000	
2001		~61,000	
2002		~90,000	
2003		~28,000	
AVERAGE	48,426	~49,800	

Most *H. zosterae* are harvested from Florida (Larkin and Degner 2001). The live marine life collection industry in Florida has grown considerably since 1990, as evidenced by increases in the number of licensed collectors and in the volume and value of landings (Adams et al. 2001). During the 1990s, seahorse landings in Florida rose 184 percent, with seahorses becoming ranked the seventh most economically important ornamental fish group (Adams et al. 2001, Baum et al. 2003). The majority of seahorses are collected in south Florida, in Monroe and Dade counties, with some collection occurring on the Gulf Coast and on the Atlantic Coast north of Dade County (Adams et al. 2001). Seahorses are harvested in targeted seahorse trawl fisheries in shallow seagrass beds, and are also harvested incidentally from shrimp trawl fisheries (Wood 2001). Alford and Grist (2005) report that the dwarf seahorse was common before 1970, but that numbers have steadily decreased and the species is now difficult and time-consuming to harvest in some areas, and that it has disappeared entirely from other areas (p. 88).

Hippocampus zosterae is one of the most popular seahorses in the aquarium trade (Vincent 1996, Wood 2001). The dwarf seahorse has been a popular aquarium pet since the 1960s. In the 1960s, dwarf seahorses were marketed as “perfect pets,” and were sold with goldfish bowls (Alford and Grist 2005). As early as 1968, *H. zosterae* was ranked as an economically important aquarium fish (Idyll 1968). Today numerous internet

companies sell dwarf seahorses (Table 3). There are no records to indicate how many of these seahorses actually survive (Alford and Grist 2005, p. 88).

Table 3. Sources of Dwarf Seahorses for Sale on the Internet. Compiled by the Center for Biological Diversity.

Source	Price	Link
Seahorse Source	10.50	http://www.seahorsesource.com/zosteræ.html
Aqualand	5.63	http://www.seahorseworld.com/product_p/12dsmaf.htm
Ocean Rider	6.25	http://www.seahorse.com/shop/index.php?main_page=product_info&cPath=1&products_id=29
Inland Aquatics	14.99	http://www.inlandaquatics.com/prod/tr_fish.html
Seahorse Corral	14.00	http://seahorsecorral.net/cart/index.php?main_page=product_info&cPath=5&products_id=10&zenid=0da27a9340e40bfcea749f27747e6dc3
Simply Exotics (UK)	104.00	http://www.simplyseahorses.co.uk/index.php?option=com_content&view=article&id=5&Itemid=2
Tropical Fish Auction	bid	http://www.tropicalfishauction.com/Dwarf-Seahorse--Hippocandus-Zosteræ-Pair-Wild-Caught,name,106391,auction_id,auction_details
LiveBrineShrimp.com	12.00	http://www.mariculturetechnology.com/Livestock.htm
ORA	retail	http://www.orafarm.com/seahorses.html
AquaCorals	39.00	http://www.aquacorals.com/SHOPFish-Seahorses-Pipefish.htm
OceanPro Aquatics	8.00	www.oceanproaquatics.com/sample/Oceanpro%20Price%20List.xls

Most *H. zosteræ* enter the domestic U.S. market, though numerous dwarf seahorses are also exported. Wood (2001) reports that the dwarf seahorse is the second most commonly exported marine ornamental fish from Florida (p. 18). Koldewey and Martin-Smith (2010) report that wild-caught *H. zosteræ* are among the species traded in large numbers globally (p. 136). Though the number reported in the CITES trade database is known to be an underestimate due to the labeling of seahorses as “tropical fish” prior to 2004, the database reports that from 1997-2008, more than 2,100 *H. zosteræ* were internationally traded, including 1,572 wild-caught seahorses, zero confirmed captive-bred seahorses, and 584 dwarf seahorses from unknown sources (p. 137). There are four aquacultural operations that breed *H. zosteræ*, some of which use wild-caught broodstock (Koldewey and Martin-Smith 2010, p. 140).

Seahorse Curio Trade

Seahorses are collected to make curios, trinkets, and souvenirs such as jewelry, key chains, paper weights, crafts, and ornaments (Vincent 1995a, Grey et al. 2005, p. 413). Vincent (1995a) reports that the curio trade is substantial in Florida (p. 127). Bruckner et al. (2005) report that most of the seahorses harvested in Florida are sold dried in the curio market, but a substantial number are also sold live in the aquarium trade (p. 98). CITES (2002) also reports that most seahorses landed in Florida are sold as curios (p. 8). It is unclear what proportion of the seahorse species entering the curio trade is made up of the dwarf seahorse, but this species is known to be in the curio market (Espinosa pers. comm. cited in Bruckner et al. 2005, p. 69). As early as the mid-1950s, dwarf seahorses were preserved, dried, and sold to shell dealers for \$15.00 to \$25.00 per 1,000 (Strawn 1954).

The United States is also a major importer of seahorses for use in curios, with an estimated 1,000 curio importers and wholesalers (Grey et al. 2005, p. 414). Florida alone has up to 10,000 retail outlets for marine curios (Ibid.). Ninety-nine percent of seahorses imported into the United States for use in curios are wild-caught (Grey et al. 2005, p. 417). Nearly 30 percent of imported seahorses to the United States come from Mexico (Grey et al. 2005), and thus likely include *H. zosterae* from the Gulf of Mexico.

Traditional Chinese Medicine

Globally, seahorses are threatened by collection for use in traditional medicines and tonic foods. Dried seahorses are believed to be useful in the treatment of numerous conditions including asthma, arteriosclerosis, impotence, incontinence, thyroid disorders, back pain, skin disease, broken bones, open wounds, and as an aid in childbirth, a general tonic, and an aphrodisiac (Vincent 1995a, 1995b). Traditional Chinese Medicine (TCM) and its derivatives account for the largest consumption of seahorses, with the global trade exceeding 20 million dried seahorses annually for use in TCM (Foster and Vincent 2005, p. 1045). The demand for seahorses in Asia has increased significantly since the mid-1980s, causing a geographic expansion in seahorse collection (Vincent 1996). Ethnic Chinese communities around the world purchase dried seahorses with numerous countries importing and exporting seahorses for use in TCM (Vincent 1995a). Pre-packaged pharmaceuticals have become popular in TCM, and pose a new threat to small seahorse species and juvenile seahorses which were previously considered undesirable (Lourie et al. 2004). This dramatic growth in demand for prepackaged medicines poses an emerging threat to small species such as *H. zosterae* (Vincent 1995a, Lourie et al. 2004). As of 1995, prepackaged medicines already accounted for 30 percent of total seahorse use in China (Vincent 1995a). In 2000, Asian trade of dried seahorses exceeded 50 tons (Lourie et al. 2004). The dwarf seahorse is exported from the United States and Mexico, and could also be used domestically for TCM in both countries. Grey et al. (2005) report that nearly two-thirds of dried seahorses imported to the United States for commercial purposes are used in traditional medicines (p. 417).

Stock assessments do not exist for *H. zosterae*, and there is a growing demand for this species in the domestic and international market for display in aquaria (Adams et al. 2001), and for use in curios (Bruckner et al. 2005, Grey et al. 1995), and prepackaged traditional medicines (Lourie et al. 2004). The data which are available indicate that while demand for this species is increasing, populations are declining (Sheridan et al. 1997, Matheson et al. 1999, Thayer et al. 1999, Alford and Grist 2005, Baum and Vincent 2005, Bruckner et al. 2005, Masonjones et al. 2010). In conjunction with the widespread loss of seagrass habitat, and pollution from the BP Deepwater Horizon oil spill, collection poses a significant threat to dwarf seahorse survival.

INADEQUACY OF EXISTING REGULATORY MECHANISMS

There are no existing regulatory mechanisms at the international, federal, or state level which adequately protect *H. zosterae* or the seagrass habitat it needs to survive.

Conservation Designations are Inadequate

The dwarf seahorse has been designated as a species of conservation concern by numerous entities, but none of these designations afford the species with regulatory protection. It is ranked by the American Fisheries Society as vulnerable, by the Nature Conservancy as imperiled, by the Commission for Environmental Cooperation as at-risk, by the state of Florida as a Species of Greatest Conservation Need, by the Commonwealth of Puerto Rico as a Species of Concern, and by Mexico as a Species Subject to Special Protection. While these designations draw attention to the plight of *H. zosterae*, they do not provide regulatory protection.

Regulation of Trade is Inadequate

The entire genus *Hippocampus* was listed in Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) in November 2002, and implementation became effective in May 2004. All Appendix II-listed taxa can still be traded internationally, but shipments must include CITES documentation that verifies that the species were obtained legally and that international trade is not detrimental to the persistence of wild populations. CITES is implemented primarily by the U.S. Fish and Wildlife Service (FWS), though the Service works collaboratively with the National Marine Fisheries Service (NMFS) and the state of Florida.

The Fish and Wildlife Service is the primary agency involved in implementation of CITES provisions. Inspectors from the agency are stationed at 15 designated ports throughout the country, and process over 100,000 shipments each year (Bruckner et al. 2005). U.S. seahorse importers and exporters must purchase a standard wildlife import/export license each year, and pay fees to process each shipment they receive at a U.S. port. Inspectors from FWS review each shipment's paperwork, and make case by case decisions whether to perform visual inspections of the shipment itself. Approximately 25 percent of all wildlife shipments are inspected, and the inspections may cover the entire shipment or just selected specimens (Bruckner et al. 2005).

Listing under CITES does not effectively protect *H. zosterae* for several reasons. First, though *Hippocampus zosterae* is exported from the United States, most dwarf seahorses enter the domestic U.S. market and are thus not subject to CITES regulation (Foster and Vincent 2005, p. 1047). Second, not all exports can be inspected, and moreover, monitoring is dependent on traders' declarations (IUCN 2010). Third, CITES documentation cannot certify that trade in *H. zosterae* is not detrimental to its persistence because comprehensive population data are not available for the dwarf seahorse, and the data which are available indicate the species is declining. Widespread decline of seagrass habitat is also indicative of widespread decline of *H. zosterae*, because the species is a habitat specialist. The state of Florida has lost more than half of its seagrass, and seagrass decline has been similarly drastic in Alabama, Mississippi, Louisiana, and Texas (Waycott et al. 2009).

CITES listing is also not sufficient to protect *H. zosterae* in Mexico. Though CITES requires that all exported seahorses be obtained legally, this is not currently the case in Mexico, where most of the seahorse trade occurs on the black market (Baum and Vincent 2005, p. 316, 318). Mexico prohibits the intentional capture and trade of wild seahorses, permitting trade only in seahorses which are incidentally caught in non-selective fishing gear. Seahorse bycatch in trawl fisheries is a significant threat to seahorses in Mexico, and trade in incidentally caught seahorses is heavy. Seahorses also continue to be illegally targeted in Mexico for the aquarium trade (Ibid.). Moreover, fisheries regulations are often not enforced, with seasonal and area closures for trawl fisheries regularly violated (Baum and Vincent 2005, p. 316). Though Mexico has established marine reserves, the level of enforcement in the reserves is unknown (Ibid.). Existing regulatory mechanisms have not prevented the decline of seahorses in Mexico. Declines in incidentally caught seahorses in Mexico are estimated to exceed 75 percent (Baum and Vincent 2005, p. 317).

In addition to the implementation of CITES provisions, at the federal level, the only additional regulatory mechanism which appears to be relevant to the U.S. seahorse trade is the Magnuson-Stevens Fishery Conservation and Management Act. The Magnuson-Stevens Act is the primary law governing marine fisheries management in the United States. The Magnuson-Stevens Act was enacted to conserve fishery resources. It establishes Regional Management Councils which are charged with developing and implementing Fishery Management Plans to restore depleted stocks. Because the dwarf seahorse has not been designated as a depleted stock, the Magnuson-Stevens Act does not pertain to current seahorse trade. In 2005 the National Marine Fisheries Service issued a final rule implementing a comprehensive amendment for the Fishery Management Plan for the Caribbean Fishery Management Council to ensure the regional plan was fully compliant with the provisions of the Magnuson-Stevens Act. In the final rule, the dwarf seahorse was included in Appendix A as a species in the aquarium trade. Aquarium trade species are included in the act for data collection purposes only (NOAA 2005, 70 FR 62073).

The dwarf seahorse occurs in the Florida Keys National Marine Sanctuary, a federally protected area, but there are no regulatory mechanisms which prohibit fishing in the sanctuary. Targeted fishing for *H. zosterae* is permitted within the sanctuary, with gear prohibitions (NOAA 1997, p. 32176). Thus, occurrence in the national sanctuary does not protect the species from exploitation.

Almost all fisheries management in U.S. coastal waters is the responsibility of state governments. At the state level, there are no data to indicate that the dwarf seahorse is commercially collected in Alabama, Louisiana or Texas. In the state of Florida it is both targeted for the aquarium trade and taken as bycatch in trawl fisheries. In 1983 the Florida Legislature created the Marine Fisheries Commission to manage Florida's marine fisheries. The commercial harvest of seahorses in Florida is monitored by the Fisheries-Dependent Monitoring (FDM) program. Florida law (Chapters 370.021, .06 (2) (a) and Administrative Code 16R-5.002) requires a Saltwater Products License (SPL) to harvest

and sell any marine species. A Marine Life Endorsement (MLE) on the SPL has been required since 1992 to participate in the marine life collection industry, the taking of organisms to be sold alive. The dwarf seahorse is designated as a “restricted species,” and license holders must also have a Restricted Species Endorsement to collect seahorses (FL 68B-42.001).

Licensed wholesale dealers must maintain records of each sales transaction by filling out a Marine Fisheries Trip Ticket for each purchase of saltwater products from a fisherman. Retailers who produce their own products must also maintain records of the saltwater products that they produce for sale through their retail license. Trip tickets are used to quantify commercial landings (pounds and value) of fish and shellfish. Required information includes Saltwater Products License number, dealer’s license number, date of purchase, time fished, county landed, gear fished, number of sets, traps pulled, species code, size code (if species graded), amount of catch (usually in pounds), area fished, depth, unit price, and dollar value (Bruckner et al. 2005).

There is no seasonal closure for seahorses; the commercial season runs from October 1 – September 30. Commercial collectors are limited to 400 dwarf seahorses per day, per person or per vessel, whichever is less. Recreational collectors are limited to 5 dwarf seahorses per day, and no person collecting recreationally may possess more than 40 total seahorses anywhere at any time (FL 68B-42.005). Fishermen who collect seahorses as bycatch must have a Marine Life Bycatch Endorsement on their license. There does not appear to be a limit on the number of seahorses which can be caught as bycatch, but the annual landings value of all marine life bycatch must be less than \$5,000 (Florida Marine Fisheries Commission 2009).

Though the commercial collection of dwarf seahorses is regulated in Florida, there are no existing data which indicate that the fishery is sustainable. The data which are available indicate that the dwarf seahorse is declining (Sheridan et al. 1997, Matheson et al. 1999, Thayer et al. 1999, Alford and Grist 2005, Baum and Vincent 2005, Bruckner et al. 2005, Masonjones et al. 2010). Because the dwarf seahorse’s seagrass habitat is known to be declining, and because the BP Deepwater Horizon spill contaminated the species’ habitat in the Gulf of Mexico, collection now poses a major threat to the survival of the seahorse, and existing regulatory mechanisms are not adequate to ensure its survival.

Seagrass Protections are Inadequate

There are no existing regulatory mechanisms which adequately protect the seagrass habitat on which *H. zosterae* depends for survival. Due to declining water quality and physical damage from boats and trawls, seagrass has declined drastically throughout the seahorse’s range. The seahorse occurs in many national and state protected areas, but even within these areas seagrass is declining. Damage from boat propellers and large ships is a primary threat to seagrass, and this damage occurs even in protected areas. In the Florida Keys National Marine Sanctuary, vessels have caused severe to moderate damage to more than 30,000 acres of seagrasses (U.S. Department of Commerce et al. 2007, p. 17). Since the designation of the sanctuary in 1990, the condition of seagrasses

has not improved (Hoegh-Guldberg 2010, p. 81), and turbidity and macroalgal blooms pose ongoing threats to seagrass (p. 47). The dwarf seahorse occurs in several state parks and other state protected areas in Florida. Even within these areas, the seahorse is threatened by loss and degradation of seagrass due to development and resultant pollution from outside park boundaries (NPS 1996, p. 62, FDEP 2004b, p. A 4-19), and to damage from vessels (FDEP 2004, p. 12). Boats frequently illegally ground in seagrass habitats, and parks do not have the resources to manage the problem (FDEP 2004b, p. 17).

The Coastal Zone Management Act (CZMA) was passed by Congress in 1972 to preserve, protect, develop, and where possible, restore or enhance, the resources of the nation's coastal zone. The Act is administered by NOAA's Office of Ocean and Coastal Resource Management (OCRM) and is intended to balance economic development with environmental conservation. The law established a voluntary national program within the Department of Commerce to encourage coastal states to develop and implement coastal zone management plans. Theoretically the CZMA provides some degree of habitat protection for the dwarf seahorse, yet despite the law's intentions, seagrass habitat has declined severely throughout the species' range.

Protection from Oil Pollution is Inadequate

One of the most dire threats to the dwarf seahorse is oil pollution. The scientists at Project Seahorse (2010) are concerned that the BP Deepwater Horizon oil spill could cause the extinction of the species. There are no existing regulatory mechanisms which protect the dwarf seahorse from the threats posed to its survival and habitat by pollution from offshore oil drilling. The Oil Pollution Act of 1990 was passed to help prevent oil spills off the coast of the United States and to ensure clean up following spills. The law requires that companies have a plan to prevent spills and have a detailed containment and cleanup plan for oil spills. The Oil Pollution Act theoretically protects the habitat of the dwarf seahorse, but given that accidental oil spills are inevitable, this law cannot safeguard the dwarf seahorse from oil pollution. As discussed above in the threats section, oil spills are commonplace in the Gulf of Mexico, with more than 320 known spills involving offshore drilling having occurred since 1964 (Robertson 2010). The 2010 BP Deepwater Horizon spill dumped 4.9 million barrels of oil into the Gulf, exceeding worst case scenario planning. Cleanup efforts from the spill, including burning of seagrass mats and the use of 1.84 million gallons of dispersant, were focused on preventing the oil from reaching shore, and potentially magnified the impacts of the spill on the dwarf seahorse. Dispersants do not reduce the amount of oil in the water, they merely change its form. By promoting dispersion of oil into the water column, dispersants increase the potential exposure of water-column and benthic biota to spilled oil. Dispersants and dispersed oil have been shown to have significant negative impacts on marine life, including plankton and fish. Dispersants can bioaccumulate and persist in sediment and in the water column where they could impact larvae and other plankton for decades (Marine Conservation Biology Institute 2010). Though the Oil Pollution Act requires companies to have a cleanup plan in place, contingency plans are not focused on the protecting of seagrass-dwelling organisms such as the dwarf seahorse.

Regulation of Greenhouse Gases is Inadequate

There are no existing regulatory mechanisms which protect the seahorse from the threat posed to its habitat and survival by global climate change. Climate change and ocean acidification represent the most significant long-term threat to the future of biodiversity on the Gulf and Atlantic Coasts. Sea level rise, stronger and more frequent hurricanes, and higher storm surge all threaten the dwarf seahorse and its seagrass habitat (Knutson et al. 1998, Easterling et al. 2000, Scavia et al. 2002, Komar and Allan 2008, Saunders and Lea 2008). Changes in water temperature, salinity, and increasing ocean acidification further magnify the threat.

To avoid catastrophic climate change, greenhouse gas emissions must be reduced, but the United States has not enacted the regulatory mechanisms necessary to accomplish this. Leading scientists and numerous scientific studies have warned that current warming and the warming commitment “in the pipeline” already constitute “dangerous” climate change with regard to species and ecosystems (Warren 2006, Hansen et al. 2008, Lenton et al. 2008, Jones et al. 2009, Smith et al. 2009). The updated Intergovernmental Panel on Climate Change (IPCC) Reasons for Concern reflect that current warming is already at a point where significant risks to species and ecosystems are occurring, and that these risks will become “severe” at a $\sim 1^{\circ}\text{C}$ rise above preindustrial levels (Smith et al. 2009). The continuation of the current global emissions trajectory, which is tracking the most fossil-fuel intensive projection of the IPCC (Raupach et al. 2007, Richardson et al. 2009), would increase the Earth’s temperature by an average of 4°C by the end of the century (IPCC 2007).

Dr. James Hansen, Director of the NASA Goddard Institute for Space Studies, and colleagues concluded that the safe upper limit for atmospheric CO_2 needed to avoid ‘dangerous climate change’ is at most 350 parts per million (ppm) (Hansen et al. 2008). Hansen et al. (2008) found that our current CO_2 level has committed us to a dangerous warming commitment of $\sim 2^{\circ}\text{C}$ temperature rise still to come and is already resulting in dangerous changes. Hansen et al. (2008) concluded that the overall target of at most 350 ppm CO_2 must be pursued on a timescale of decades since paleoclimatic evidence and ongoing changes suggest that it would be dangerous to allow emissions to overshoot this target for an extended period of time:

“If humanity wishes to preserve a planet similar to that on which civilization developed and to which life on Earth is adapted, paleoclimate evidence and ongoing climate change suggest that CO_2 will need to be reduced from its current 385 ppm to at most 350 ppm, but likely less than that” (p. 217).

With atmospheric carbon dioxide at ~ 389 ppm and worldwide emissions continuing to increase by more than 2 ppm each year, rapid and substantial reductions are clearly needed immediately to protect the dwarf seahorse and countless other species, and there are currently no regulatory mechanisms to enact these reductions.

In sum, for the reasons discussed above, there are no regulatory mechanisms which adequately protect the dwarf seahorse from the threats it faces from habitat loss and degradation and commercial collection.

DISEASE AND PREDATION

Neither disease nor predation are known to present a population level threat to *H. zosterae* at this time. In conjunction with other threats however, either or both of these factors could become significant. For example, the threat of predation becomes more significant in conjunction with habitat degradation. Loss of seagrass contributes to altered predator-prey relationships and other important ecological interactions (Frazer 2010). Matheson et al. (1999) found that populations of benthic predators increased as the seagrass canopy used by *H. zosterae* decreased (p. 535).

At least 82 species are known to prey on seahorses, including various invertebrates, numerous fishes including red snapper, dorado, rays, skates, tuna, dolphin fish, anglerfish, flatheads, sea urchins, blue cod, ling, red cod, sea perch, and trumpeter, loggerhead sea turtles, cormorants, penguins, and other water birds, and marine mammals (Foster and Vincent 2004, Kleiber et al. 2010). Observations of seahorses with shortened tails may also indicate that partial predation by crabs may threaten seahorses (Baum et al. 2003). Predation pressure is likely higher on juvenile than adult seahorses, as adults have prominent bony plates and spines and excellent camouflage abilities (Lourie et al. 2004). Kleiber et al. (2010) reviewed 135 accounts of predation on seahorses and pipefishes and found that seahorses form a high proportion of predator diets during seahorse population booms or die-offs, when individuals are concentrated in floating marine vegetation, or when juveniles are abundant during the breeding season.

Little is known about the incidence of disease in wild populations of seahorses, but seahorses are known to suffer from a variety of diseases in captivity, many of which are fatal (Alcaide et al. 2001, Wooten and Waughman 2004). Vincent and Clifton-Hadley (1989) report an outbreak of microsporidan infection (*Glugea heraldi*) in a colony of seahorses that were caught in Florida Bay which killed 74 of 76 individuals. The seahorses were also diagnosed with a myxosporidan (*Sphaeromyxa* sp.) and an unidentified nematode infection, but the later two were not thought to be causative of the mortalities. Wild *Hippocampus zosterae* may also be subject to infection from various parasites including the isopod *Lironeca ovalis* and the haematozoan *Haemogreyarina bigemina*, and to ulcers and lesions (Masterson 2008). Martin-Smith and Vincent (2005) report a rapid and severe decline in populations of big-bellied seahorse (*H. abdominalis*) in Tasmania that were unrelated to fishing or trade. They hypothesize that the decline was potentially caused by disease or effects from invasive species.

OTHER NATURAL OR ANTHROPOGENIC FACTORS

Several other factors threaten the dwarf seahorse including life history parameters that make the species vulnerable to decline, noise, bycatch, unintentional and illegal fishing, hurricanes and tropical storms, invasive species, and unknown factors.

Life History Factors

Several life history factors make *H. zosteræ* vulnerable to extinction, especially in conjunction with threats such as habitat loss and overutilization. These factors include rarity, patchy spatial distribution, low abundance, low population density, low fecundity, site fidelity, complex reproductive behavior, and monogamous mating system (Musick et al. 2000, p. 13, Baum et al. 2003, p. 721, Martin-Smith and Vincent 2005, p. 534, Masonjones et al. 2010, p. 119).

Noise

Low frequency, moderate intensity noise from a variety of activities has significant deleterious effects on *H. zosteræ*. Low frequency boat motor noise negatively impacts the health, behavior, and reproductive success of dwarf seahorses (Masonjones and Babson 2003). Masonjones and Babson (2003) found that adult seahorses exposed to noisy conditions showed a significantly higher incidence of gas bladder disease, behavioral differences, and had significantly longer gestation lengths than controls. Fewer offspring were born to parents that were exposed to continuous noise, in addition to their offspring being smaller and having lower growth rates than control offspring. The authors report that their results suggest a strong effect of low frequency noise on both adult and juvenile *H. zosteræ*, indicating that this stressor may influence reproduction both in field populations and potentially in captive breeding programs.

Bycatch Mortality

In addition to targeted collection of *H. zosteræ* which is discussed in the Overutilization section above, the dwarf seahorse is threatened by being caught as bycatch in commercial trawling operations. Commercial trawling negatively impacts seahorses through habitat degradation, social disruption, and through displacement, injury, and mortality of incidentally caught individuals (Baum et al. 2003, p.721, Baum and Vincent 2005, p. 317, Curtis et al. 2007, p. 469).

Seahorses are likely greatly affected by nonselective fishing gear because intense trawling often covers seahorse habitat and because the life-history traits of seahorses, such as monogamy, obligatory paternal parental care, and relatively low fecundity, render them particularly vulnerable to overexploitation (Curtis et al. 2007). Sparse distribution and low mobility also inhibit the ability of seahorses to recolonized depleted areas (Baum et al. 2003, p. 721).

Baum et al. (2003) analyzed bycatch of the lined seahorse (*Hippocampus erectus*) in the bait-shrimp trawl fishery in Hernando Beach, Florida and found that approximately 72,000 seahorses were caught annually as bycatch in the fishery, with the number of lined seahorses caught per trawl ranging from zero to 16, and very few *H. zosteræ* being caught. It is unknown how many *H. zosteræ* are annually caught as bycatch, but even a low level of catch could have negative repercussions for the species due to life history

characteristics which limit its resiliency to overexploitation (Curtis et al. 2007). Baum et al. (2003) conclude that even species that comprise only a small portion of the total bycatch in a fishery may experience significant impacts of incidental harvest on their population size and structure (p. 721).

In a study of trawling-induced mortality, Baum et al. (2003) found that few seahorses died during towing or sorting, but that nearly 5 percent suffered injuries. Post-release mortality of seahorses caught as bycatch is unknown, but is likely high. Meyer et al. (1999 pers. comm. cited in Baum et al. 2003) found that only one of four dwarf seahorses caught as bycatch in a live-bait shrimp trawl fishery study in Tampa Bay remained alive in the seawater holding tank 36 hours after collection. Discarded seahorses are also likely subject to intense predation pressure from dolphins and schools of fish upon release (Meyer et al. 1999, Baum et al. 2003, p. 729). Even if seahorses survive after being returned to the water, they will experience deleterious effects from being removed from their home ranges and undergoing disturbance of pair bonds (Foster and Vincent 2004, p. 50).

While most studies have focused on heavy trawls, Curtis et al. (2007) found that even repeated use of light demersal fishing gears can influence the structure of non-target species under exploitation, both directly through fishing mortality and indirectly through habitat alteration (p. 479).

Foster and Vincent (2004) report that small seahorse species constitute a significant proportion of discards from most tropical shrimp fisheries. Baum and Vincent (2005) report that seahorse fishermen in Latin America have reported substantial declines in seahorse abundance attributable primarily to incidental catch in shrimp trawlers (p. 305). Most dried seahorses exported from Mexico, and most of those traded domestically, are caught incidentally in Mexico's commercial shrimp trawl fisheries (Baum and Vincent 2005, p.309). Eighty-eight percent of experienced shrimp trawl fishers on the Caribbean coast of Mexico who commented on changes in seahorse catch rates reported that catch rates had diminished over time (Baum and Vincent 2005, p. 316). Mean estimated catch per unit effort declines were greater than 75 percent in all regions on the Atlantic coast of Latin America. In the Gulf of Mexico, shrimp trawl fishers reported declines in catch per unit effort of greater than 90 percent (p. 316).

Unintentional and Illegal Fishing

The dwarf seahorse is potentially threatened by illegal fishing and by derelict fishing gear. Even protected areas in Florida are subject to illegal fishing activity. The Florida Department of Environmental Protection (2009) reports that Terra Ceia Aquatic Preserve has considerable illegal and unintentional fishing activity, including violations of prohibitions on net fishing and taking of undersized and/or protected animals (p. 56). Abandoned traps and nets also threaten aquatic species. FDEP (2009) reports that derelict crab traps are common within Terra Ceia Aquatic Preserve, continuing to entrap marine life after being abandoned. An additional threat is posed by discarded nets which become anchored to the substrate by barnacles and other organisms and pose an entanglement

hazard (p. 56). Removal of derelict lobster and stone crab traps is also a continual problem at John Pennekamp Coral Reef State Park (FDEP 2004b, p. 26).

Hurricanes and Tropical Storms

In addition to habitat damage, hurricanes and tropical storms pose physical threats to *H. zosterae*. The churning of water during hurricanes can wash seahorses from their territories into new areas which may not contain appropriate habitat (Breder 1962, p. 459). High winds and waves associated with storms also increase water turbidity by re-suspending bottom sediments, which can interfere with the visual cues seahorses use for reproduction and social structure (FDEP 2009b, p. 23).

Invasive Species

Invasive species potentially threaten the dwarf seahorse. Martin-Smith and Vincent (2005) documented significant declines in seahorse populations in Tasmania that appeared to be unrelated to habitat degradation or trade. They hypothesize that the declines were possibly related to interactions with invasive species due to physical or trophic modifications to the ecosystem (p. 533). Global climate change is expected to lead to increased density and diversity of exotic and invasive species, increasing the threat (FWS 2010, p. 44, Joel Fodrie et al. 2010).

Unknown Factors

Unknown factors may also be contributing to the decline of *H. zosterae*. Worldwide, many seahorse populations are declining for reasons that are not understood (Masonjones et al. 2010, p. 119).

CONCLUSION

The Endangered Species Act requires that NMFS promptly issue an initial finding as to whether this petition “presents substantial scientific or commercial information indicating that the petitioned action may be warranted.” 16 U.S.C. § 1533(b)(3)(A). There is no question that under the five listing factors of the Act, listing the dwarf seahorse may be warranted. The seahorse is threatened by loss or curtailment of habitat or range, overutilization, inadequacy of existing regulatory mechanisms, and various other factors. Available scientific information indicates that the seahorse is declining. The life history traits of the dwarf seahorse render the species vulnerable to decline. For the seahorse to have the best chance at recovery, it should be promptly protected under the Act and given the critical habitat it needs to survive.

REQUEST FOR CRITICAL HABITAT DESIGNATION

Petitioners urge NMFS to designate critical habitat for *Hippocampus zosterae* concurrently with its listing under the ESA because of the serious nature of the threats to the species. Critical habitat as defined by Section 3 of the ESA is:

(i) the specific areas within the geographical area occupied by a species, at the time it is listed in accordance with the provisions of section 1533 of this title, on which are found those physical or biological features (I) essential to the conservation of the species and (II) which may require special management considerations or protection; and (ii) the specific areas outside the geographical area occupied by the species at the time it is listed in accordance with the provisions of section 1533 of this title, upon a determination by the Secretary that such areas are essential for the conservation of the species. 16 U.S.C. § 1532(5).

Congress recognized that the protection of habitat is essential to the recovery of listed species, stating that: classifying a species as endangered or threatened is only the first step in insuring its survival. Of equal or more importance is the determination of the habitat necessary for that species' continued existence... If the protection of endangered and threatened species depends in large measure on the preservation of the 40 species' habitat, then the ultimate effectiveness of the Endangered Species Act will depend on the designation of critical habitat. H. Rep. No. 94-887 at 3 (1976).

Critical habitat is an effective and important component of the ESA, without which the dwarf seahorse's chance for recovery diminishes. Species with critical habitat are twice as likely to be recovering compared to species lacking designated habitat (Taylor *et al.* 2005). Petitioners request that the NMFS propose critical habitat for the dwarf seahorse concurrently with its proposed listing.

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