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Conservation and management of seahorses and other Syngnathidae

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This article analyses the pressures on seahorses and explores conservation responses. It focuses on seahorses (*Hippocampus* spp.) but also considers pipefishes and seadragons, especially where they can fill gaps in seahorse knowledge. The charisma of many syngnathids can make them good flagship species for threats and solutions in marine conservation. The article combines a synthesis of published literature with new data on the trade in seahorses for traditional medicine, aquarium display and curiosities. Most traded seahorses come from trawl by-catch, although seahorses are also targeted. The total extraction is large, tens of millions of animals annually, and unsustainable. A first review of the effect of habitat change on syngnathids raises many questions, while suggesting that some species may cope better than others. The combination of pressures means that many species of syngnathid are now included in the IUCN Red List of Threatened Species or national equivalents. In addition, seahorse exports from 175 countries are limited to sustainable levels under the Convention on International Trade in Endangered Species (CITES) of Wild Fauna and Flora. Possible conservation measures include marine protected areas, fisheries management, select aquaculture ventures, trade regulation, improved governance (particularly) and consumer engagement.

Key words: aquarium trade; by-catch; habitat degradation; threatened species; traditional Chinese medicine.

INTRODUCTION

The global conservation status of marine fishes has only really been formally assessed since 1996, when the first IUCN Red List workshop for marine fishes was held and the first list was proposed (Vincent & Hall, 1996). Even in 1996, the implied position that marine fishes were wildlife as well as economic commodities, and could be judged with the same criteria as elephants or orchids, provoked a storm of controversy and many demands that criteria be changed by raising the thresholds for listing (Vincent & Hall, 1996; Dulvy *et al.*, 2005). Such debate originated from historic perceptions that marine fishes were too abundant and too fecund to be threatened by exploitation (Huxley, 1883). These arguments are, however, anachronistic given mounting evidence that many marine fish populations are in trouble (Dulvy *et al.*, 2003; IUCN, 2010*a*).

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Overexploitation was the earliest driver of coastal ecosystem collapse, and remains the most significant threat to marine populations (Jackson *et al*., 2001). Despite large increases in global fishing effort, cumulative yields across all species and large marine ecosystems have declined by 13% since passing a maximum in 1994 (Worm *et al*., 2006). Large predatory fish biomass today is only *c*. 10% of pre-industrial levels (Myers & Worm, 2003). In a meta-analysis of 230 marine fish populations, there was a median reduction of 83% in breeding population size (Hutchings & Reynolds, 2004). In order to maintain growth in fisheries, exploitation has expanded geographically at about one degree of southward movement annually since the 1950s (Swartz *et al*., 2010). Other threats to marine populations include habitat destruction, introductions of exotic species and climate change, in that order (Jackson *et al*., 2001).

A wide array of individual and population characteristics renders some fish species particularly vulnerable to population depletion. Targeted fishing and pollution affect larger more than smaller marine fishes, while habitat loss or degradation disproportionally affects small marine fishes (Olden *et al*., 2007). Large body size, small range size and ecological specialization are linked with vulnerability of 61 marine species (Dulvy *et al*., 2003). A review of extinction risk in marine fishes found that large body size, late maturity, high trophic level, slow growth, demersal behaviour and long life span were correlated with vulnerability in different species (Reynolds *et al*., 2005). Other variables that increase a species' threat status include low reproductive effort, strong Allee effects (tendency for population to decline numerically when it gets below a certain threshold size or density), low adult mobility, nearshore horizontal distribution, narrow vertical depth range, high population patchiness within the species' range, high habitat specificity, high habitat vulnerability and rarity (Dye *et al*., 1994; Roberts & Hawkins, 1999).

Seahorses (*Hippocampus* Rafinesque 1810 spp.) and their relatives in the family Syngnathidae (pipefishes, seadragons and pipehorses) have life histories and behaviours that might make them vulnerable to population decline (Foster & Vincent, 2004). Seahorse taxonomy remains problematic (Lourie *et al*., 1999, 2004; Kuiter, 2001, 2009) but morphological, genetic and behavioural evidence suggests there are currently *c*. 48 species, once synonyms have been reconciled (S. Lourie, unpubl. data). In all seahorse species, the female transfers her eggs to the male, where they are fertilized (Boisseau, 1967; Vincent, 1990). The male then broods the developing embryos in a specialized abdominal pouch, providing them with nutrition, oxygen and a controlled environment (Linton & Soloff, 1964). After *c*. 10 days to 6 weeks, depending on species and water temperature (Lin *et al*., 2006), the male gives birth to relatively few, independent young (Vincent, 1990; Foster & Vincent, 2004) that then disperse in the plankton (Morgan, 2007). In many species, the same male and female mate repeatedly and exclusively (Vincent & Sadler, 1995; Kvarnemo *et al*., 2000, 2007; Foster & Vincent, 2004). Adults of many species (and especially mated males) maintain small home ranges (Foster & Vincent, 2004; Vincent *et al*., 2005) although they also occasionally engage in short-range migratory moves (Strawn, 1953; Boisseau, 1967; Hardy, 1978). Many of the other 300 or so species and 52 genera of pipefishes, pipehorses and seadragons in the family Syngnathidae (Kuiter, 2009) have similar life histories, although pipefishes differ across genera in the form of their brooding structure (Wilson *et al*., 2003; Monteiro *et al*., 2005). As well, some pipefish range much more widely than seahorses by, for example, undergoing seasonal migrations (Lazzari & Able, 1990).

Syngnathidae are important in ecological, economical, medicinal and cultural terms. They live in corals, seagrasses, macroalgae, mangroves, estuaries, lagoons and open bottom habitats and can be important predators on bottom-dwelling organisms (Tipton & Bell, 1988; Bologna, 2007). Subsistence fishers in some nations make a substantial portion of their annual income catching seahorses (Pajaro *et al*., 1998; Vincent *et al*., 2007) for use in ornamental display, curios and traditional medicine (Vincent, 1996; Parry-Jones & Vincent, 1998; May & Tomoda, 2002; Alves & Rosa, 2006; Qian *et al*., 2008; Vincent *et al*., 2011). They are often featured in art and stories and are attractive enough to draw considerable public support for their conservation and for the larger marine environment (Scales, 2009).

This article extracts and integrates available information on seahorse conservation and management. A recent review obtained literature on seahorse ecology and biogeography and surveyed available information on the seahorse trade (Scales, 2010). This paper largely takes up the story from there. It presents new data and analyses of seahorse fisheries and trade (and their effects), provides the first survey of how habitat change affects syngnathids and promotes a plan of conservation action. The paper focuses on seahorses, but other syngnathid species are discussed where knowledge on them complements or fills gaps in research on seahorses. The paper also identifies research and action needed to improve the capacity to assess and address conservation concerns.

CONSUMPTION AND TRADE

DATA SOURCES

Information on the consumption and trade in seahorses and other syngnathids was gathered from three sequential sets of data that each provide information on species, trade routes, source and consumer countries and total trade volumes of dried and live seahorses. Given the differences in how the three sets of data were collected and the inherent challenges in all three, their compatibility on key findings is notable.

The first investigation into the international trade in syngnathids involved extensive Asian trade field surveys in 1993 and 1995 (Vincent, 1996). The results from these first surveys indicated a large global trade in seahorses (Table S1) and aroused conservation concern that led to the establishment of Project Seahorse in 1996 (www.projectseahorse.org). This marine conservation group has executed most subsequent trade research in syngnathids and launched the first conservation projects for this taxon.

The next investigation into seahorse trade (with limited attention to other syngnathids) was conducted by Project Seahorse team members from 1998 to 2001. These second surveys covered a broader geographic scale than the first surveys and, consequently, uncovered an even broader trade in seahorses than found in the first surveys (Table S1). Detailed trade reports for each country and region outside Asia have been compiled into one report (Vincent *et al*., 2011), with a similar Asian overview to follow. Many of these reports (Asian and non-Asian) have also been summarized as primary papers (McPherson & Vincent, 2004; Baum & Vincent, 2005; Giles *et al*., 2006; Martin-Smith & Vincent, 2006; Perry *et al*., 2010).

Both sets of trade field surveys drew on (1) *in situ* interviews with participants in the trade (*e.g.* fishers, buyers, importers, exporters, and retailers) or those with knowledge of the trade (*e.g.* scientific researchers and non-governmental organisations) and (2) official data collected by government agencies detailing either the catch or trade of seahorses and pipefishes, primarily from Taiwan (from 1982) or Hong Kong Special Administrative Region, China (HKSAR, from 1998).

More information on seahorse trade became available once the genus *Hippocampus* was added to Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES: www.cites.org), and this forms the third set of data for this article. Since that listing was implemented in 2004, all nations signatory to CITES (currently 175 parties) have been required to submit export and re-export records for seahorses. Some have also chosen to submit import records (UNEP-WCMC, 2010). CITES import and export records for 2004 to 2008 (the latest available) were analysed for temporal and geographical trends in volumes and trade routes (Evanson *et al*., 2011). Re-exports were excluded from the analyses, to preclude double counting. The reliability of this analysis is subject to inherent challenges with the CITES database; *e.g.*, reporting was submitted by parties themselves, data might reflect permits or actual exports, units (individuals or kg) are often missing, (required) export and (voluntary) import records did not match, and species were often not (or were wrongly) identified.

USES

The first surveys revealed that seahorses and other syngnathids are used for traditional medicine (TM), aquarium display and curiosities. In conservation terms, it is irrelevant whether a seahorse taken from the wild is sold dried or live. That said, differences in the method of capture or in the preferred species or size may affect the overall conservation effect of removal and effective conservation actions.

The great majority of seahorses, *c*. 95% of those in trade, are sold for use in TM and particularly in traditional Chinese medicine (TCM; Tang, 1987). TCM has been codified for *c*. 2000 years (Zhu & Woerdenbag, 1995) and is accessed (at least to some extent) by many ethnic Chinese people, who together comprise perhaps one-quarter of the global population. Seahorses also play a role in *hanyak* and *kanpo*, the derivatives of TCM used in Korea and Japan, respectively, and in Indonesia's *jamu* TM (Vincent, 1996; Lee, 1998*b*). In TCM, seahorses are used (whole or powdered) in combination with other ingredients to treat ailments ranging from asthma and arteriosclerosis to impotence and incontinence (Vincent, 1996; Lee, 1998*a*, *b*; May & Tomoda, 2002; Qian *et al*., 2008). The most valuable seahorses in TCM are individuals that are large, pale and smooth (Vincent, 1996; Baum & Vincent, 2005; Perry *et al*., 2010). Seahorses that were sold bleached, a processing only known to occur in HKSAR, were sold at a premium price wherever they were available in Asia. The emergence of pre-packaged formulations of TM in boxes and bottles (Vincent, 1996) contrasts with individually tailored treatments usually offered in TM and might remove the bias for larger, paler, smoother seahorses.

The aquarium trade uses fewer seahorses than the dried trade, but can place heavy pressure on particular populations or species. The greatest demand in the live trade comes from the hobby market, but most public aquariums also display seahorses (Vincent, 1996). Complex seahorse husbandry requirements mean that they often fare very badly in captivity, although considerable advances have been made in the last decade (Vincent & Koldewey, 2006; Koldewey & Martin-Smith, 2010). Those animals that die in the live trade are often sold dried for TM or as curios (Vincent, 1996).

Little is known about the curio trade for seahorses but, globally, large numbers are certainly involved. In the U.S.A., for example, seahorses comprised the second most significant documented import of marine fish curios in terms of volume, at *c*. 65 000 individuals (valued at *c*. U.S. $10 000) year^{-1} (Grey *et al*., 2005). Similarly, large numbers of dried seahorses have been sold as curios in Portugal, among other countries (Vincent, 1996).

SPECIES

Species designations in trade records and studies should be viewed with caution, given the high potential for misidentification, especially of dried specimens, and the high frequency with which multiple species are traded and sold in the same shipment or case. About a quarter (23%) of all entries in CITES database were listed under the generic designation *Hippocampus* sp. (Evanson *et al*., 2011) and certain default species names are probably over-represented in the remaining CITES records (Evanson *et al*., 2011).

The first identification guide exclusively for seahorses (Lourie *et al*., 1999) allowed researchers executing the second surveys to identify 24 seahorse species in trade (of 32 species then recognized): 20 were sold dried, 17 for TM and 12 for curiosities, and 19 were sold live for the aquarium trade (Appendix I). CITES trade data included reports on more than half of all currently known seahorse species (28/48). Eighteen species were reportedly used in the dried trade, probably mostly for TM, while 27 seahorse species were traded live (Appendix I; UNEP-WCMC, 2010; Evanson *et al*., 2011).

At least 21 other syngnathid species are also traded, dried and alive. The CITES database covers only trade in seahorses, but first and second surveys revealed sales for TCM in all five species in the pipehorse genus, *Solegnathus* Swainson 1839, and in five pipefish species (Appendix I; Vincent, 1996; second surveys). Specimens of *Solegnathus* are particularly valuable in TCM (Martin-Smith *et al*., 2003; Martin-Smith & Vincent, 2006) where they are known as *hai long* (Vincent, 1996; Martin-Smith *et al*., 2003). This name translates as seadragon but the weedy seadragon *Phyllopteryx taeniolatus* (Lacépède 1804) and leafy seadragon *Phycodurus eques* (Günther 1865) are not used in TCM and are only traded as aquarium fishes (Vincent, 1996; second surveys). Other syngnathids were also sold as popular aquarium fishes, including at least 13 pipefish species (Appendix I; Vincent, 1996; second surveys).

FISHERIES

Current understanding of seahorse fisheries comes primarily from haphazard sampling during the first and second surveys, supplemented by a few sets of landing data and catch records (Meeuwig *et al*., 2006 for by-catch in trawls; Vincent *et al*., 2007 for target catch). The Queensland east coast otter-trawl fishery may maintain the only mandated tracking of syngnathid landings, requiring records of all by-catch of syngnathids because they are species of conservation interest in Queensland (Dodt, 2005). There is a great need for more systematic and enduring monitoring of syngnathid catches globally (including discards), both to improve assessments of species' and populations' conservation status and to validate trade records.

The vast majority of seahorses in trade (as much as 95%) comes from shrimp trawl by-catch, as do virtually all *Solegnathus* pipehorses (McPherson & Vincent, 2004; Baum & Vincent, 2005; Giles *et al*., 2006; Martin-Smith & Vincent, 2006; Perry *et al*., 2010). Many syngnathids are likely to be particularly vulnerable to capture in shrimp trawls because they live in the same habitats as shrimp and are demersal, slow swimming and much the same size as targeted shrimp. In some regions, seahorses and (particularly) *Solegnathus* pipehorses are valuable enough, and income-earning opportunities are few enough, to warrant extraction from the piles of by-catch or even perhaps to influence fishing patterns (Martin-Smith & Vincent, 2006). Where they are not worth extracting, seahorses are presumably discarded or processed into low-grade fishery by-products such as fishmeal along with the rest of the small species in the by-catch.

Indiscriminate gear tends to catch few seahorses haul^{-1}, but the cumulative total is large. For example, shrimp trawlers in Vietnam only caught a mean of about one or two seahorses vessel^{-1} day^{-1} in the late 1990s (Meeuwig *et al*., 2006), but that led to annual total catches of *c*. 6·5 t of seahorses, or *c*. 2·2 million animals (Giles *et al*., 2006). Other countries with significant seahorse by-catch in shrimp trawls include (in descending order of volume) the Philippines, Thailand, India, Malaysia, Indonesia and Mexico (Table I). Seahorses were also caught by trawlers in Florida, U.S.A. that obtain live shrimp for bait and haul the nets quickly enough to extract live seahorses; one study found that 72 000 were caught every year in just one small area near one port (Baum *et al*., 2003). Seahorses are also obtained in many other gear types, ranging from beach, shore and purse seines (McPherson & Vincent, 2004; Murugan *et al*., 2008) to crab pots (J. Selgrath, pers.comm.).

Most direct exploitation for syngnathids is by small-scale or subsistence fishers in developing countries (McPherson & Vincent, 2004; Perry *et al*., 2010), although some are taken by aquarium collectors in developed countries such as Australia or the U.S.A. (Larkin, 2003; Martin-Smith & Vincent, 2006). In Brazil, seahorses and pipefishes are mostly caught by hand in seas <7 m deep, generally by breath-hold divers (Rosa *et al*., 2006). In the central Philippines, fishers in Bohol swim at night

TABLE I. Estimates of annual seahorse by-catch by country. All estimates are from data up to 1999, except India (2001), Mexico (2000) and the Philippines (2001). More recent data are not available

Country	Annual seahorse by-catch (kg)	Annual seahorse by-catch (*n*)	Reference
India	9430	1.4×10^6	Salin *et al*. (2005)
Indonesia	350	170×10^3	A. Perry & A. C. J. Vincent (unpubl. data)
Malaysia	2900	912×10^3	Perry *et al*. (2010)
Mexico	170	72×10^3	Baum & Vincent (2005)
Philippines	5000–15 000	$2–6 \times 10^6$	M. Pajaro (unpubl. data)
Thailand	6600	2.1×10^6	Perry *et al*. (2010)
Vietnam	6500	2.2×10^6	Giles *et al*. (2006)

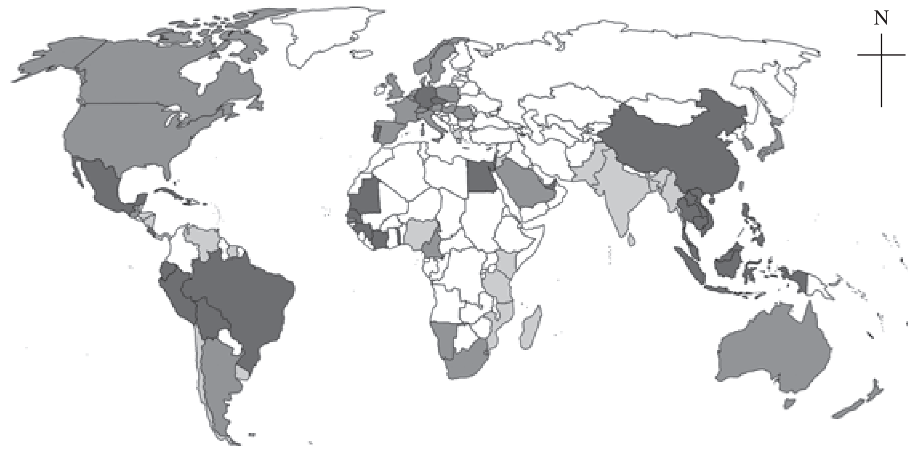

FIG. 1. Countries known to trade seahorses according to CITES (2004–2008) – net exporters (■) and net importers (■) – and historical surveys (1996–2001; ■). Countries not surveyed or not known to trade (□) are indicated.

under low-slung lanterns, towing their boats and free diving to collect the seahorses by hand (Martin-Smith *et al*., 2004), while those in northern Palawan employ small dip-nets from rafts to obtain seahorses in waist-deep water (Vincent, 1996). In India, divers target seahorses along with sea cucumbers *Holothuria* spp. and gastropods (*e.g. Murex* spp., *Turbinella pyrum*; Salin *et al*., 2005).

TRADE

The trade in seahorses (and some pipefish species) is global, covering all continents except Antarctica. The first surveys documented seahorse exchange among at least 32 countries (Table S1; Vincent, 1996). The subsequent, more thorough, set of surveys found that at least 72 countries were involved in the trade (Table S1; second surveys; Fig. 1). While some of this discrepancy is explained by increased survey effort, some of it represented genuine new engagement, particularly in Africa (McPherson & Vincent, 2004) and Latin America (Baum & Vincent, 2005). The most recent data reported by parties to CITES showed a total of 70 countries involved in the trade, with at least 46 exporters exchanging specimens with 45 importers (Table S1; UNEP-WCMC, 2010; Evanson *et al*., 2011).

Dried trade

In spite of an observed geographic expansion of the seahorse trade, Asian countries have remained the main sources and destinations for dried seahorses. All three data sets reveal that Thailand has been a dominant source of seahorses, by annual quantity, since 1996. The first and second surveys showed that India, Philippines and Vietnam were major exporters of seahorses (Vincent, 1996). In the second surveys, Mexico and Tanzania also appeared to be major exporters of seahorses. CITES data indicate that while Thailand continued to export many seahorses, mainland China was also now a major exporter. In these data, Guinea, though a small contributor, apparently exported more than Vietnam (Evanson *et al*., 2011). In the second surveys, most specimens appear to have been sourced from trawl by-catch, although both the Philippines and India had significant target fisheries. According to CITES data, India did not export seahorses (Evanson *et al*., 2011), in accordance with their national regulations banning catch and trade of seahorses (Indian Ministry of Environment and Forests, 1972, 2001). Exports from the Philippines were reported in 2004 and 2005 (Evanson *et al*., 2011) but then ceased, most likely as a 2004 national ban on seahorse capture began to take effect (Philippine Department of Agriculture, 1998).

The three data sets agree that the major consumers of dried seahorses were HKSAR, Taiwan and mainland China. While the first and second surveys also highlighted Singapore as a major consumer, CITES data did not show this. CITES data, however, unexpectedly found that Japan was a major seahorse importer, while the UK, USA, and Australia played smaller roles (Evanson *et al*., 2011).

The dried seahorse trade involved many tonnes of animals annually (Table II). Estimates across data sources were relatively convergent, especially given the great differences and uncertainties in data collection methods. The CITES data, however, did indicate lower volumes in some years than the two surveys suggest. There is no inherent reason to trust one data set over the other, formal CITES data often include gaps and discrepancies (World Conservation Monitoring Centre, 2004; Blundell & Masica, 2005); a comparative and critical study was undertaken to probe these differences (Evanson *et al*., 2011). Seahorses and pipehorses can be valuable commodities, with prices increasing along trade routes, to a maximum of about U.S. \$1200 kg^{-1} of large smooth, pale seahorses in Hong Kong (Vincent, 1996). The formal trade from HKSAR and Taiwan provide the only enduring data on import values, together reporting dried seahorse imports totalling U.S. \$1.7 million in 2000, for example (second surveys). CITES data did not include any information on price or value.

Live trade

Live seahorses sourced in Asia are usually exported to either North America or the European Union (E.U.). Both the first and second surveys reported the major suppliers of live seahorses to be the Philippines and Indonesia, with some of the latter routed through Singapore for re-export (Vincent, 1996; second surveys). The CITES data reported live seahorse exports from 27 countries, primarily from Vietnam, Sri Lanka and Indonesia (UNEP-WCMC, 2010; Evanson *et al*., 2011). All three data sets showed that the vast majority of live seahorses for the aquarium trade end up in the U.S.A. (particularly) or the EU. Only the first surveys indicated large exports of live seahorses to Australia, HKSAR, Japan and Taiwan (Vincent, 1996).

The live seahorse trade involves tens to hundreds of thousands of animals annually (Table II). As with the dried trade, CITES data showed lower volumes than the surveys suggested. The first surveys indicated that the Philippines and Indonesia alone supplied the aquarium trade with perhaps 500 000 and 100 000 seahorses $year^{-1}$, respectively, which is more than later estimates for the whole trade (Vincent, 1996). The lower numbers reported in the CITES data (Evanson *et al*., 2011) may be related to the Philippines' ban on seahorse capture and trade as of 2004.

The first and second survey data documented a live trade derived almost entirely from wild populations, whereas CITES trade records showed that tank-raised or cultured animals comprised 36% (2004) to 80% (2008) of seahorse live trade (Koldewey & Martin-Smith, 2010). Many cultured seahorses in global trade come from

TABLE II. Global estimates of annual dried and live seahorse trade as a mean with ranges representing estimated minimum and maximum values

	Dried			Live	
Source	Tonnes year^{-1}	Individuals year^{-1} (millions)	Conversion (individuals kg^{-1})	Individuals year^{-1} (millions)	Reference
First surveys (1996)	30–45 (Asia only; no mean reported)	9–36	300–800	610 000–1 000 000 (Asia only; no mean reported)	Vincent (1996)
Second surveys (1998–2001)	39–67 (mean 54)	14–23 (mean 19)	350	131 000–187 000 (mean 152 000)	Compilation (see text)
CITES (2004–2008)	16–49 (mean 27)	6–18 (mean 10)	370	20 000–170 000 (mean 120 000)	UNEP-WCMC (2010); Evanson *et al*. (2011)

breeding operations that raise non-native species. According to CITES data, *Hippocampus reidi* Ginsburg 1933 (native to the Caribbean) constituted 14–37% of all exported live seahorses globally in 2004–2008. *Hippocampus kuda* Bleeker 1852 constituted 55–72% of reported exports starting in 2006, dominating trade in all subsequent years. Of the live *H. reidi* trade between 2004–2008, a mean of 93% (range 88–96%) was captive bred, and a mean of 98% of these were from Sri Lanka (UNEP-WCMC, 2010; Evanson *et al*., 2011). By raising an exotic species, culture operations in Sri Lanka were able to prove that their animals were captive bred and thus simplify the process of exporting under CITES regulations. Culturing exotic species does, however, raise many environmental concerns, not least to do with the effect that escapes might have on wild native species (Weir & Grant, 2005; Vincent & Koldewey, 2006).

The live trade in seahorses was not particularly valuable as a whole, especially when compared to the dried trade (Vincent, 1996; second surveys) but provided important income to local fishers and traders (Pajaro *et al*., 1998; Vincent *et al*., 2007). Both first and second surveys showed that fishers were usually paid more per seahorse in the live trade than in the dried trade. The second surveys found that prices for wild seahorses in retail outlets ranged from U.S. $1–80 per seahorse. By comparison, cultured seahorses (that would eat dried or frozen food) were trading online for U.S. $60–$950 per fish in November 2010 (http://www.simplyseahorses.co.uk/; http://www.seahorse.com/).

EFFECTS OF FISHERIES

Direct or indirect fishing can affect seahorse individuals, populations and species in a variety of ways. For example, field sampling showed that trawls that obtained *Hippocampus erectus* Perry 1810 in Florida had the potential to (1) injure or kill individuals, (2) disrupt social structure by selectively capturing females, (3) reduce reproduction by disrupting pair bonds, (4) affect cohorts differentially and (5) damage habitat by removing seagrasses (Baum *et al*., 2003).

Seahorses (and other syngnathids) may be overexploited for the following reasons: (1) there is a high and persistent demand for seahorses, such that all animals extracted from the wild can find a market; (2) there is effectively no cost associated with catching seahorses in the non-selective gear that extracts so many; (3) the morphology of seahorses allows them to be dried and stored easily, giving fishers and buyers operating far from export centres the option of gradually accumulating marketable numbers of seahorses; (4) opportunity costs for the subsistence fishers that commonly target seahorses are low, given that the such fishers have few other sources of income; (5) with many luxury items, such as seahorses, the value per unit tends to rise as the target species becomes rarer (Courchamp *et al*., 2006). All these factors render biological overfishing more likely, because the fishery will seem economically viable even as the resource heads towards collapse (Sadovy & Vincent, 2002).

DECLINING NUMBERS

Inferences about numeric changes in syngnathid populations must often be drawn from trade surveys (Vincent, 1996). Most direct estimates of syngnathid abundance and density *in situ* were often mere snapshots with no temporal coverage (Rosa *et al*., 2007; Barrows *et al*., 2009), many of them buried in reports on fish community assemblages (Laffaille *et al*., 2000; Jones & West, 2005). The few studies that did track temporal change and identify declines took place in unfished areas (Power & Attrill, 2003; Martin-Smith & Vincent, 2005; Rosa *et al*., 2007; Barrows *et al*., 2009; Masonjones *et al*., 2010).

Fisheries and trade analyses, including interviews, become the default options for inferring populations and fisheries. Fishery landings have seldom been directly assessed, with most fisheries data being derived from interviews with fishers. One exception comes from the central Philippines, where catch landings were documented in one community for 4 years. This study revealed strong seasonal differences in CPUE, which was low compared to historical rates, along with decreases in the proportion of brooding males and increases in the mean size of caught seahorses over time (Vincent *et al*., 2007).

The use of narrative accounts from fishers and others associated with seahorse trade can help to create a longer perspective than would be available from rare biological data alone (Turvey *et al*., 2010). For example, interview data helped to recreate the probable effect of Vietnamese bottom trawling on seahorse catches whereas formal records only date from 1996 (Meeuwig *et al*., 2006). Trade surveys do, however, commonly suffer from limitations, particularly with respect to the difficulty in estimating effort. Also, the inferred severity of the population decline (and therefore extinction risk) hinges on assumptions about the accuracy of fisher or trader recall (O'Donnell *et al*., 2010); a quantitative study showed that interviews may exaggerate early fishing rates and capture less variability than logbooks (O'Donnell *et al*., 2010).

In a series of trade surveys around the world, declines in seahorse landings were reported by the majority of fishers in all regions surveyed. Of 598 fishers interviewed worldwide who commented on changes in seahorse catch (by-catch and targeted), 403 (67%) reported declines (second surveys; Table III). The following are explicit examples documented by Project Seahorse, to complement Table III, with the caveat that effort data were of variable quality: (1) in the central Philippines, lantern fishers (who caught seahorses as part of a multispecies fishery; all 21 interviewed) reported declines in catch per unit effort of 75–93% in the three decades to 1994 (O'Donnell *et al*., 2010); (2) most fishers in Vietnam who responded ($n = 122$ of 143) reported that seahorse catches had declined over the previous 2 to 5 years, with most of those estimating a 30–60% decline (Giles *et al*., 2006). Buyers also reported decreases in seahorse availability ($n = 21$ of 27) in most regions; (3) fishers in both Malaysia and Thailand described substantial declines in seahorse by-catch quantity per vessel ($n = 37$ of 52 and 30 of 37, respectively). Given that the number of trawl vessels was either stable (Malaysia) or had fallen in recent years (Thailand), lower catches per fisher probably indicated that seahorse populations in these areas were declining (Perry *et al*., 2010); (4) on the Pacific and Atlantic Ocean coasts of Latin America, declines in seahorse by-catch rates were reported by almost all who commented ($n = 88$ of 115) and were estimated to have exceeded 75% (Baum & Vincent, 2005). Declines were proportionately less commonly cited in areas with lower exploitation.

TABLE III. Trends in seahorse catches over time as reported by fishers, arranged alphabetically by country. The time frame given by fishers for these changes was highly variable (from 3 to 30 years) (B. Giles, unpubl. data)

Country	Number of fishers surveyed	Reported change in status of seahorse populations					Citing decrease (%)
		Increase	Decrease	No change	Do not know	No answer	
Brazil	29		25			4	86
Costa Rica	3		3				100
Ecuador	27		15			12	56
Guatemala	7		5			2	71
Honduras	13		9			4	69
India	160	23	80	57			50
Malaysia	52	1	37	14			71
Mexico (Caribbean)	29		21	3	5		72
Mexico (Pacific)	21		18			3	86
Nicaragua	8		6			2	75
Panama	14		7			7	50
Peru	9		5			4	56
Philippines	23		18			3	78
Thailand	53		43	10			81
Vietnam	163	7	122	14	20		75
Total	598	31	403	98	25	41	72

DEPENSATORY EFFECTS

Allee effects may contribute to population decline in seahorses (Vincent & Sadler, 1995; Foster & Vincent, 2004). In the many monogamous seahorse species, a widowed seahorse generally takes some time to re-pair (Vincent & Sadler, 1995), especially at low population densities where potential new mates are rare. Re-mating potential is also reduced by the adult fidelity to small home ranges and slow swimming speeds that limit immigration (Foster & Vincent, 2004; Vincent *et al*., 2005). Even when they do form, new pairs may have smaller broods than established pairs (Vincent, 1994; Vincent *et al*., 1994).

Allee effects might be more pronounced where fishing affects one sex disproportionately, particularly in the vast majority of species that have faithful pair bonds or serial monogamy. Live-bait trawling in Florida landed more *H. erectus* females than expected (catches were 58% female), perhaps because of spatial segregation (Baum *et al*., 2003). Greater catches of reproductively active males in shallower areas suggests that males may spend most of their time inshore of the trawled area (Baum *et al*., 2003). Weedy seadragons (*P. taeniolatus*) provide another example of where fishery landings might be biased by sex and reproductive state, because pregnant males hide in sheltered kelp patches more than other seadragons (Sanchez-Camara *et al*., 2006). In such cases of spatial segregation, localized habitat damage might also be expected to affect one sex more than the other.

Allee effects arising from size-selective fishing may also be problematic. Larger seahorses may be disproportionately extracted because of their greater value in the dried seahorse trade: some traders assert that large seahorses from target fisheries became increasingly difficult to obtain, such that smaller animals comprised more of their stock (McPherson & Vincent, 2004). Size selectivity will be costly since larger seahorses produce more young (Nguyen & Do, 1998) and young with greater postnatal growth rates and higher chances of survival (Dzyuba *et al*., 2006). In some pipefishes (*e.g. Syngnathus typhle* L. 1758), larger females produced more and larger eggs that gave rise to higher quality offspring (Ahnesjö, 1996). The size class most affected by fishing will also depend on spatial segregation: for example, breath-hold fishers in the Philippines tend to fish outside the *Sargassum* spp. habitats where more of the smaller, juvenile seahorses are found (Morgan & Vincent, 2007). Again, localized habitat damage may cause size-biased effects on species where size classes are somewhat segregated in space.

POPULATION CHANGES IN THE ABSENCE OF FISHING

Changes in population abundance of syngnathids cannot always be attributed to fishing and may represent fluctuations driven by episodic recruitment patterns arising from other factors. Two of the rare long-term monitoring studies on seahorses have been conducted by Project Seahorse. In the first, *Hippocampus abdominalis* Lesson 1827 in the Derwent Estuary, Tasmania, Australia, declined 79 to 98% over the period 2001–2004 in the absence of any fishing pressure or evident change in estuarine physicochemical conditions (Martin-Smith & Vincent, 2005). Putative explanations included interactions with invasive species, disease or reproductive limitation through allee effects (Martin-Smith & Vincent, 2005), but anecdotal reports indicate that numbers have subsequently increased again (K. Martin-Smith, pers. comm.). In the second Project Seahorse long-term study, *Hippocampus guttulatus* Cuvier 1829 in Ria Formosa, southern Portugal also declined considerably in number (94% over 6–8 years relative to Curtis & Vincent, 2005; I. Caldwell, unpubl. data). There was no known direct fishing pressure, but the lagoon system was subject to anthropogenic habitat stressors known to alter benthic fauna assemblages, including pollution, eutrophication and aquaculture (Gamito, 2008). As in the case of *H. abdominalis*, however, ongoing research is identifying a relative increase in the number of *H. guttulatus* (M. Correia, unpubl. data). Population sizes of *Hippocampus capensis* Boulenger 1900, a species endemic to just three lagoons in South Africa, can certainly also fluctuate considerably. Two lagoons exhibited declines of 80 and 100% respectively in just 1 year (Lockyear *et al*., 2006), after which monitoring ceased. Likewise, a 2 year study in Tampa Bay, Florida, found that unfished populations of *Hippocampus zosterae* Jordan & Gilbert 1882 increased while populations of *Syngnathus scovelli* Evermann & Kendall 1896 and mean size of individuals of both species declined (Masonjones *et al*., 2010).

EFFECTS OF HABITAT CHANGE

This first review of relationship between syngnathids and habitat change explores the limited available data for the effects of (1) physical damage to habitats, (2) chemical pollutants, eutrophication or other changes in water quality, (3) noise pollution, (4) invasive species and (5) climate change.

Seahorses and other syngnathids live in what are considered to be some of the world's most threatened marine habitats: seagrasses (Fonseca *et al.*, 1998; Orth *et al.*, 2006; Waycott *et al.*, 2009), mangroves (Valiela *et al.*, 2001; FAO, 2003; Polidoro *et al.*, 2010), coral reefs (Wilkinson, 2008), estuaries (Blaber *et al.*, 2000; Lotze *et al.*, 2006; Deinet *et al.*, 2010) and macroalgae (Steneck *et al.*, 2002; Airoldi & Beck, 2007). Syngnathids often also live among anthropogenic features, such as aquaculture pens, moorings, swimming (shark) nets and jetty pylons (Clynick, 2008; Harasti *et al.*, 2010) to an extent that these may make them important habitats for some, or even many, species.

PHYSICAL STRUCTURE

Syngnathids can certainly be affected (lethally or sublethally) by physical degradation and destruction of their habitats. In Malaysia, *H. kuda* numbers declined as an extensive port development around the Pulai Estuary, destroying large tracts of seagrass meadow (McKenzie *et al.*, 2006–2010; Sabri, 2009). In Florida, populations of *H. zosterae* and *S. scovelli* in Tampa Bay declined during demolition and construction phases of two nearby marina projects that damaged seagrass habitats (Masonjones *et al.*, 2010). In the Philippines, coral reefs that had been degraded by blast and poison fishing had very low densities of *Hippocampus comes* Cantor 1849 (Marcus *et al.*, 2007). A fourth example points to the need to assess habitat damage carefully: the eelgrass in damaged vegetation was in sufficiently good condition that *Stigmatopora nigra* Kaup 1856 preferred the vegetation over more open areas (Connolly, 1994). Some species (*e.g. Syngnathus fuscus* Storer 1839) also adjust foraging tactics to match varying habitat complexity (Ryer, 1988), while variation in structural complexity had little effect on the foraging success of *H. erectus* (James & Heck, 1994). Habitat alteration may still have consequences, however, with the example that *H. erectus* grew more rapidly in tanks with macroalgae than in those without (Xu *et al.*, 2010).

WATER QUALITY

At least some syngnathids may respond poorly to chemical pollutants, eutrophication or other changes in water quality and associated visibility. It is unclear whether seahorses have any particular sensitivity to environmental change (indicator species) but they are certainly vulnerable to stress, often manifested as disease (Koldewey & Martin-Smith, 2010). Two *Syngnathus* species [*Syngnathus floridae* (Jordan & Gilbert 1882) and *S. scovelli*] were significantly less abundant in polluted seagrass beds in the northeast Gulf of Mexico than in recovering ones (Livingston, 1984). Temporal trends in the mean annual population size of Nilsson's pipefish *Syngnathus rostellatus* (Nilsson 1855), sampled from the Thames Estuary in the U.K., correlate with changes in water treatment practice and reductions in organic pollution (Power & Attrill, 2003).

Some effects of reduced water quality arise from lower light levels or hypoxia associated with eutrophication or pollution. For example, lower light levels reduced prey capture rates by the seahorse *H. erectus* (James & Heck, 1994). This has broader implications as visual acuity has been recognized as important for prey capture in other syngnathids, *e.g. S. typhle* and *Syngnathus abaster* Risso 1827 (Mouillot *et al.*, 2007). In experimental conditions, male *S. typhle* (a polygynandrous species) in limited visibility spent less time assessing potential mates and no longer preferentially mated with large females (Sundin *et al*., 2010). Hypoxic conditions arising from excessive fertilizer use in Chesapeake Bay, U.S.A., led to reductions in feeding by northern and dusky pipefishes (*S. fuscus* and *S. floridae*, respectively) that were predicted to affect health, growth and reproduction (Ripley & Foran, 2007). That said, both these pipefish species were able to tolerate very low dissolved oxygen concentrations, below levels lethal to most commercially important species (Ripley & Foran, 2007). Such tolerance accords with observations that seahorses are often found in habitats with suboptimal water quality, in and around busy marinas, for example (A. Vincent, pers. obs.). Indeed, hypoxic conditions negatively affected embryonic growth but not embryonic survival in the brood pouch of a male *S. typhle* (Braga Gonçalves, 2010).

NOISE

Noise pollution may affect syngnathid populations, not least because the low mobility of these fishes means they have particularly limited ability to escape noise. Seahorses exposed to loud noise in aquariums for 1 month demonstrated physiological, chronic stress responses with reduced mass and body condition (Anderson, 2009) and spent less time stationary, an indication of irritation (Anderson *et al*., 2011). Reproductive and feeding rates had not, however, declined by the end of the experiment (Anderson, 2009). In the wild, settlement-stage larval syngnathids certainly distinguish among noises, preferring high-frequency over low-frequency trap noise on coral reefs (Simpson *et al*., 2008). The implications of such distinctions have not, however, been assessed.

INVASIVE SPECIES

The few available studies about the effects of invasive species on syngnathids relate to exotic vegetation. The double-ended pipefish *Syngnathoides biaculeatus* (Bloch 1785) in Moreton Bay, Queensland, Australia, preferred seagrass over an invasive alga (*Caulerpa taxifolia*) but also preferred both over unvegetated habitat (Burfeind *et al*., 2009). In estuaries in New South Wales, Australia, beds of invasive *C. taxifolia* had fewer syngnathid species and fewer syngnathids than adjacent seagrass beds, and sometimes no syngnathids at all (York *et al*., 2006). On the other hand, densities of *S. scovelli* did not differ between an invasive macrophyte (*Myriophyllum spicatum*) and native macrophytes in Lake Pontchartrain estuary, Louisiana, U.S.A., with the pipefish merely preferring vegetated to unvegetated areas (Duffy & Baltz, 1998). One hypothesis for large declines in *H. abdominalis* in Tasmania, Australia, was ecosystem change associated with the arrival of two exotic invertebrates: the northern Pacific seastar *Asterias amurensis* and the European green crab *Carcinus maenas* (Ross *et al*., 2004; Martin-Smith & Vincent, 2005).

CLIMATE CHANGE

Climate change is expected to negatively affect inshore marine habitats and their fauna, including syngnathids, through changes in, for example, temperature, rainfall patterns, atmospheric CO_2, community composition, oceanographic patterns, status of coastal habitats and storm action (Parry *et al*., 2007). For example, predicted levels of CO_2 are expected to lead to mass bleaching and ocean acidification of coral reefs (Veron *et al*., 2009), presumably with adverse consequences on reef-associated syngnathids. As one example of potential threats from climate change, a combination of flooding and high littoral water temperatures of up to 32° C resulted in the death of at least 3×10^3 *H. capensis* and several hundred *Syngnathus acus* L. 1758 in the marginal areas of the Swartvlei Estuary in South Africa (Russell, 1994). Also, populations of *S. scovelli* in the Thames Estuary in the U.K. decreased in response to drought-induced increases in water temperature (Power & Attrill, 2003). Although *H. erectus* and *Hippocampus whitei* Bleeker 1855 both grew more rapidly at warmer water temperatures, there are presumably limits on syngnathids' capacity to cope (*H. whitei*: Wong & Benzie, 2003; *H. erectus*: Lin *et al*., 2008).

POSITIVE RESPONSES TO CHANGE

Habitat change may actually sometimes favour increases in certain syngnathid populations. For example, habitat damage caused by seine fishing in the Ria Formosa lagoon system, Portugal, benefited *Hippocampus hippocampus* (L. 1758) while creating problems for the sympatric *H. guttulatus*: the former prefers the simpler habitats that emerge from fishing while the latter occupied the more complex habitats that are damaged by seining (Curtis & Vincent, 2005; Curtis *et al*., 2007). Also, there were more *S. scovelli* in boat-scarred than in reference seagrass beds in Charlotte Harbor, Florida (Bell *et al*., 2002). A vast increase in the numbers of pipefish *Entelurus aequoreus* (L. 1758) in the north-east Atlantic Ocean and North Sea after 2003 was variably attributed to (1) introduction of an alien habitat-forming seaweed (*Sargassum muticum*) that created habitat with more food and shelter for the pipefish, (2) increases in sea surface temperatures, (3) fishery removal of putative competitors and (4) expansions in the range of their prey items (Kirby *et al*., 2006; Lindley *et al*., 2006; Fleischer *et al*., 2007; Harris *et al*., 2007). That said, anecdotal reports suggest a considerable drop in numbers of *E. aequoreus* by 2010 (I. Ahnesjö, pers. comm.; R. George, pers. comm.; M. Gollock, pers. comm.; S. Wanless, pers. comm.).

ARTIFICIAL HABITAT

Anthropogenic habitat has potential value for syngnathids to a level that needs to be investigated. For example, the numbers of *H. abdominalis* and *H. whitei* declined significantly upon replacement of a swimming net in Australia, with recovery of the latter population taking >15 months (Harasti *et al*., 2010). In Tasmania, Australia, antipredator nets at Atlantic salmon *Salmo salar* L. 1758 farms can have large numbers of seahorses on them (Marshall, 2004), which die when the nets are removed from the water to be treated against fouling (K. Martin-Smith, pers. comm.). The real conservation effect of removing nets will depend on whether they actually enhance seahorse numbers or merely act as fish attraction devices, spatially concentrating the same number of seahorses.

CONSERVATION STATUS

Many species of seahorse are now included in the IUCN Red List of Threatened Species, which serves to warn of serious concern but does not itself impose any restrictions (Appendix I; www.iucnredlist.org). Globally, *H. capensis* is not currently in trade but is on the IUCN Red List, as Endangered because of its restricted and fragmented distribution and habitat decline. Seven others (all found in trade) are listed as Vulnerable, based on observed, estimated, inferred or suspected population declines of 30% over a 10 year period. These declines are attributed to changes in area of occupancy, extent of occurrence, habitat area or quality and levels of exploitation. Most seahorse species (29, of which one is possibly a synonym) are listed as Data Deficient, because there is inadequate information to make an assessment of their risk of extinction based on their distribution and population status. The IUCN Red List also includes 16 pipefishes, with *Syngnathus watermeyeri Smith* 1963 Critically Endangered, due to restricted range, continued decline in habitat quality, and the absence of mature individuals, and *Microphis insularis* (Hora 1925) Vulnerable, because of a restricted geographic range. Most other syngnathids, including newly described seahorse species, have not been assessed.

Seahorses are also found on lists of threatened species on regional, national or local levels but it is important to note that many of these lists do not use IUCN Red List criteria, even where the status names are the same. Again, most of these lists simply warn that a species faces trouble without necessarily supporting or protecting them in any direct manner. In addition, *H. hippocampus* and *H. guttulatus* are listed on the OSPAR Commission List of Threatened and Declining Species and Habitats for the Northeast Atlantic (OSPAR, 2008). Nationally, at least 15 countries or management regions around the world recognize seahorses as facing considerable threats (Appendix I). As one example, Brazil recently produced its first list of aquatic invertebrates and fish species that are endangered, overexploited or threatened by exploitation (MMA, 2004), a listing that requires the implementation of a recovery plan for a number of species, including seahorses.

CONSERVATION ACTION

Conservation of seahorses, as of all other species, depends on interdisciplinary approaches. Biological responses will simply not be enough in light of the multiple layers of socio-economic and political pressure bearing down on seahorse populations (Vincent, 2008). Seahorses will only flourish in healthy marine communities and ecosystems, but these depend on wise choices by fishers and their societies with support from many levels of government and global policies.

At least three clear anthropogenic pressures must be addressed: overexploitation in target fisheries (*e.g. H. comes* in the Philippines; Vincent & Pajaro, 1997), incidental take in non-selective gear (*e.g. H. erectus* in the Gulf of Mexico; Baum *et al.*, 2003) and threats from habitat degradation (*e.g.* Endangered *H. capensis*; Lockyear, 2000). Goals may take the form either of trends (a defined percentage increase) or of targets (a defined number of individuals) and should conserve the evolutionary potential of the species, not just a viable population size (Redding & Mooers, 2006; Isaac *et al.*, 2007). Regular monitoring of index (or sentinel) populations, fisheries and trades may be the most pragmatic way to assess conservation action. Action to protect seahorses, however, will also depend on discerning key variables and processes that are lacking for even the best-studied populations, such as density dependence, recruitment dynamics and effective population sizes.

SEAHORSE POPULATIONS

Seahorse populations will benefit from regulations and practices that protect the animals themselves. A total of eight countries plus the E.U. have some form of national monitoring or management for both dried and live syngnathids (Table S2). A further 14 countries have legislation specifically referring to live syngnathids. Restrictions range from complete bans on seahorse capture and exports to requirement for export permits implemented independent of CITES regulations or other trade monitoring measures (Table S2).

Better information about basic life-history variables relevant to management is needed for virtually all syngnathid species, particularly to reduce the number of Data Deficient seahorse species on the IUCN Red List and to assist CITES parties in setting export controls. The global scale of this research and conservation enterprise means that co-ordinated contributions from volunteers, such as educational institutions, public aquariums, citizen groups and dive clubs, would be an indispensable asset. The Seahorse Trust in the U.K. (www.theseahorsetrust.org) and Dragon Search in Australia (now www.reefwatch.asn.au/dsIntroduction.html) provide two helpful starting points. Genetic techniques may also be useful in estimating levels of genetic variation, gene flow, effective population sizes, local breeding population size and evolutionarily significant management units (Mobley *et al*., 2011).

ECOSYSTEM PROTECTION

Protection and restoration initiatives are underway for habitats where seahorses are found, including seagrasses, mangroves, coral reefs, estuaries and macroalgae. To focus conservation and management effort, population dynamics of seahorses need to be modelled in the context of habitat availability (Levin & Stunz, 2005). Take the case of *H. comes* in the Philippines. If the juvenile phase was identified as the most vulnerable, then conservation efforts would need to prioritize the macroalgae that the young inhabit (Morgan & Vincent, 2007). On the other hand, should the adult stage be most vulnerable, then the priority would be protection in areas of great habitat richness, where large reproducing adults are found (Morgan & Vincent, 2007). Among the many global initiatives to support marine habitats, Save Our Seahorses (founded in 2004) is distinctive in focusing on syngnathid habitat and monitoring; this non-profit group is committed to saving seahorses and pipefishes in the Pulai Estuary in Malaysia, partly by surveying seagrass beds (www.sosmalaysia.org). In a case that emphasizes the potential value of anthropogenic habitat to syngnathids, local authorities in Sydney, Australia, have agreed to manage growth on swimming nets in a way that minimizes effects on local seahorse populations (Harasti *et al*., 2010). A syngnathid conservation team in Brazil is hoping to benefit from fishers' skill at identifying microhabitats that hold higher densities of seahorses (Rosa *et al*., 2005).

It is unclear how much marine-protected areas (MPA) support seahorses, although they should be useful for seahorse communities and habitats. A large number of scientists, stakeholders and policy-makers support MPAs (1) as a means of improving fish species richness, number and size within the designated area (Leslie, 2005; Lundquist & Granek, 2005) and (2) in theory, of enhancing fish populations in nearby waters (Rowley, 1994; Gerber *et al.*, 2003; Mumby & Steneck, 2008). Project Seahorse has used seahorses effectively as flagship species to help generate 34 small, community-enforced and managed no-take MPAs in the central Philippines. Ironically, although these MPAs have been beneficial for many fish species in the community, seahorses themselves are actually among the cryptic families with more fishes outside the reserves (Samoilys *et al.*, 2007), possibly because of the recovery of predatory species within reserves. Syngnathids in these MPAs did, however, tend to be larger than in neighbouring waters, presumably with consequent reproductive advantages (Yasué *et al.*, in press). In contrast, syngnathids were indeed judged to be good flagship species for MPAs in New South Wales, Australia, leading to the argument that locating MPAs to support syngnathid species would reputedly benefit other fishes more than randomly selecting MPA locations (Shokri *et al.*, 2009).

FISHERIES MANAGEMENT

Fisheries management measures are available to support syngnathid populations, although the dearth of life-history and fisheries knowledge limits their specificity. One response is to adopt a process of adaptive management, in which tools and approaches are explicitly experiments, directly involve stakeholders, and are revised in light of new information. An iterative process of consultation involving diverse stakeholders (fishers, traders, consumers, conservationists, aquarists, national and international policy groups) produced a clear preference for no-take MPAs and minimum size limits to help improve sustainability of seahorse fisheries (Martin-Smith *et al.*, 2004). There was also strong support for marine tenure and moderate support for sex-selective fishing (leaving pregnant males), with the latter likely to be feasible, thanks to fisher acceptance and the ease by which pregnant males can be identified (Martin-Smith *et al.*, 2004). Setting quotas or limiting the number of fishers was considered difficult, given the limited information on seahorse population sizes and their intrinsic rate of increase, as well as the socio-economic realities of subsistence fishing.

Controls on indiscriminate fishing gear, in time and space, could limit incidental capture of seahorses. MPAs are one obvious management measure but other spatial or temporal controls might also help seahorses. For example, if reproductively active animals were concentrated in particular areas (Baum *et al.*, 2003), then trawling might be redirected away from these regions during important breeding periods. Likewise, if subadult seahorses were found in shallower water than adult seahorses (Dauwe, 1992; Perante *et al.*, 1998), then elimination of trawling activities from shallow zones might reduce the risk of recruitment overfishing.

Modification of trawl gear and methods is unlikely to do much to support wild populations of syngnathids. For example, a by-catch reduction device in a glass eel *Anguilla anguilla* (L. 1758) fishery was fully effective for all species and individuals >40 mm total length, except for *S. abaster* (Lopez & Gisbert, 2009). Setting gear for shorter time periods could allow retrieval of live seahorses, as in a Florida live-bait trawl fishery where nets are brought up every 30 to 60 min (Baum *et al.*, 2003), but even there the seahorses might suffer from physical injury, habitat damage, removal from home ranges and disturbance of pair bonds (Davis, 2002; Baum *et al.*, 2003). There may also be later mortality; only one-quarter of *H. zosterae* were alive 36 h after extraction from 5 min experimental trawls (Baum *et al.*, 2003; D. Meyer, pers. comm.), although *S. scovelli* had higher survival rates in this context (Meyer *et al.*, 1999).

CAPTIVE BREEDING AND AQUACULTURE

Numerous captive breeding and aquaculture ventures have emerged for seahorses and, to a lesser extent, pipefishes (Kaiser *et al.*, 1997; Payne *et al.*, 1998; Partridge *et al.*, 2004; Koldewey & Martin-Smith, 2010). There are many concerns about how aquaculture practices in general can damage marine environments and affect wild populations of the farmed species through disease transmission or escapes (Naylor *et al.*, 2000). Most seahorse aquaculture involves small-scale operations in developed countries that sell live animals for the home aquarium market (Koldewey & Martin-Smith, 2010). At least 13 species are in commercial culture or under research for their culture potential (Koldewey & Martin-Smith, 2010). Prior to the 1990s, seahorse aquaculture was plagued by problems with disease and feeding. In the late 1990s and early 2000s, however, there was considerable expansion in the number and size of aquaculture operations and the number of species in culture (Koldewey & Martin-Smith, 2010). The improvement in seahorse aquaculture has been reflected in an increasing contribution of captive-bred seahorses to the aquarium trade. In contrast, none of the many large-scale aquacultures developed to supply the traditional medicine market or as a livelihood venture has yet proven commercially viable (Koldewey & Martin-Smith, 2010).

Proposals to release captive-reared seahorses often emerge in discussions on seahorse conservation and usually ignore a key area of conservation policy. The IUCN Re-introduction Specialist Group is, however, clear that the pressures leading to population declines must have been alleviated and the effects on host populations evaluated before any release can be contemplated (IUCN, 1998). Release of captive-bred seahorses, even in well-intentioned restocking (*e.g.* restocking of *H. hippocampus* along sites on the Canary Islands' coasts; Domínguez & Ferrer, 2009), might well convey disease, alter genetics and disrupt social and spatial behaviour of wild conspecifics or congenerics (as with other marine species; Naylor *et al.*, 2000).

TRADE MANAGEMENT AND CITES

Trade management, particularly under CITES, has opened some promising avenues for seahorse conservation. In 2002, CITES added the entire genus *Hippocampus* to Appendix II of the convention, thus requiring all exports to be sustainably and legally sourced (CITES, 2004). Since the listing came into effect in May 2004, all parties to CITES have to ensure that their exports are not detrimental to wild populations of seahorses and to document their exports on a central global database (UNEP-WCMC, 2010). This CITES action, based largely on Project Seahorse data and programmes, is globally significant as the first listing for marine fishes of commercial importance in decades, followed by Appendix II listings for sharks and humphead wrasse *Cheilinus undulatus* Rüppell 1835. The listing also sets an agenda for seahorse conservation work, as parties seek advice and input to meet their obligations for sustainable exports.

The technical Animals Committee for CITES suggested that parties might meet their obligations for sustainable exports by adopting a minimum size limit of 10 cm height for all seahorses in international trade (CITES, 2004). The recommendation came from the wide-ranging previously mentioned Project Seahorse consultation with interested groups (Martin-Smith *et al*., 2004). The 10 cm universal minimum size limit is a compromise to ensure that breeding occurs in most species before the seahorses recruit to the fishery, while still leaving seahorses in trade (Foster & Vincent, 2005). The limit allowed 15 species to begin reproducing before being recruited into international trade, while another 16 species were (1) essentially not in international trade, (2) safeguarded under domestic legislation, (3) partly protected by this size limit or (4) geographically isolated so that they could be managed individually. That left only one large species (*Hippocampus kelloggi* Jordan & Snyder 1901) completely unsupported by the 10 cm limit.

Parties will certainly need to devise other management measures to complement or replace the minimum size limit in trade. It does not solve problems of incidental capture in non-selective gear or the emerging challenge of managing trade in pre-packaged TCM, in which seahorses have been ground into just one of many ingredients in a capsule or pill. Two CITES workshops, one specifically on seahorses and one on general non-detriment findings, have produced guidelines for exporting parties: these include the possibility of quotas and the option of spatial management (Bruckner *et al*., 2005; CITES, 2009). Brazil, for example, set export quotas that have been progressively reduced since 2002 (Rosa *et al*., 2006). In improving their export controls, CITES Management and Scientific Authorities are now able to access an emerging one-stop web-based resource that Project Seahorse has designed for their needs (www.hippocampusinfo.org). Parties can also draw on molecular forensics, using genetic techniques to identify seahorses to species and, in some cases, rough geographic origin; such an approach could help in verifying trade documentation and also in developing more effective, locally specific conservation measures (Sanders *et al*., 2008; Saarman *et al*., 2010).

SOCIAL DEVELOPMENT AND GOVERNANCE

Fishers' compliance and support are essential to any conservation action for exploited species. Much of the local *in situ* seahorse conservation activity to date has been executed in collaboration with subsistence fishing communities on Danajon Bank, a double barrier reef in the central Philippines (Vincent & Pajaro, 1997). Project Seahorse and its partner in the Philippines, the Project Seahorse Foundation for Marine Conservation, have worked closely with small-scale fishers who depended heavily on catching seahorses (to sell live and dried) for substantial portions of their annual income. Apart from assisting communities to establish MPAs (Samoilys *et al*., 2007; Yasué *et al*., 2010), the teams have helped build citizens' groups responsible for MPA management, create an alliance of small-scale fishing families and develop local government capacity for marine resource management (www.projectseahorse.org).

Many TCM traders and researchers have been generous with their engagement in marine conservation. The demand for dried syngnathids in TCM will continue, both because mainland China favours a dual system of health care (Hesketh & Zhu, 1997; Xu & Yang, 2009) and because seahorses and pipefishes are considered to have special properties that would be difficult to replace completely (C.-H. Tsang, pers. comm.). Even should culture facilities begin to produce seahorses for the TCM trade, it is far from clear that these animals would be accepted *in lieu* of wild animals, which are usually perceived as more potent in TCM (Moreau *et al*., 1998). There is, however, encouraging movement in the TCM trade. For example, the Hong Kong Chinese Medicine Merchants Association issued a voluntary code of conduct, before CITES placed seahorses on Appendix II, that asked members to avoid small (<10 cm) or pregnant seahorses or seahorses caught during their reproductive season (C.-H. Tsang, pers. comm.). In order to help move the TCM trade in marine species towards sustainability, Project Seahorse established an advisory committee in Hong Kong comprising representatives of TCM traders, government, public institutions, higher education and conservation organizations.

Demand for live seahorses will remain high, given the appeal of seahorses and the scale of the global marine ornamental industry (Wabnitz *et al*., 2003). Public aquariums have hitherto contributed to seahorse conservation through public education, with seahorse exhibits (usually developed with Project Seahorse) reaching *c*. 10 million visitors annually, and through limited research and field projects. They have, however, long been considering strategies to ensure the sustainability of their collections (Thoney *et al*., 2003; Koldewey & Zimmerman, 2007). There is new impetus for such improvements, with the World Association of Zoos and Aquariums actively encouraging sustainable acquisitions as best practice (Penning *et al*., 2009). The Marine Aquarium Council certification scheme was intended to improve sustainability of sourcing for ornamental fishes, through voluntary codes of conduct and an allied certification scheme (www.aquariumcouncil.org). Although this programme is currently in a hiatus, its standards and associated certification process remain relevant and viable.

As a precept, good governance would go a long way to securing populations of seahorses and other wildlife. CITES provides one example of how governments can foster or engender compliance or insist on enforcement. Most national and regional governments also have existing laws that would support syngnathids specifically or marine conservation more generally, if enforced (Table S2). Many of the regulations in place specifically to protect seahorses are, however, ignored. For example, seahorses were illegally targeted in Mexico for the live trade (Baum & Vincent, 2005) and many seahorses have historically been exported from India, despite these animals officially enjoying full protection (Table S2). Other legislation that affects seahorses is also flouted: for example, many countries already ban trawling in coastal waters, at least at certain times and places, but few enforce these regulations. Legislation is also ineffectual in other ways, as when seahorses are legally exported but at volumes that exceed set quotas (*e.g.* Indonesia; Nijman, 2010).

Ultimately, of course, the future of seahorses depends on human population pressure and consumption. The very impoverished communities that fish seahorses often seek additional income-earning opportunities, particularly as fisheries become difficult and unreliable (pers. obs.). There is, however, little evidence thus far to connect so-called alternative or supplementary livelihoods to direct conservation gains for seahorses or, indeed, for other species (Salafsky *et al*., 2001). Indeed, small-scale fishers often seek a diversity of livelihood options and will embrace new opportunities without relinquishing fishing (Sievanen *et al*., 2005). At a global level, demand for marine life is likely to grow, given that the world may have up to 10·6 billion people by 2050, an increase of *c*. 60% from now (United Nations Department of Economic and Social Affairs, Population Division, 2006).

CONCLUSION

The issues facing syngnathids, including overexploitation, by-catch and habitat degradation, are major concerns in marine conservation today. To make progress, there is a need to engage the relevant stakeholders and establish how to assess, protect and manage these fishes simply and effectively. The fact that the conservation status of so many seahorse species is Data Deficient demonstrates the challenges in conducting robust conservation assessments. The situation for most other syngnathid species has not even been evaluated. The shortfall in knowledge can partly be attributed to their life-history characteristics: seahorses are very highly cryptic and are often found naturally at low densities (Foster & Vincent, 2004). In addition, their taxonomy continues to be unresolved, resulting in many identification challenges and points of discussion and disagreement (Teske *et al.*, 2007; Foster & Gomon, 2010).

This review seeks to generate more research on and support for these fishes. There is a desperate need for empirical data, for long-term monitoring of wild seahorse populations and for fisheries documentation and port surveys. Exploitation, either direct or indirect, is a significant driver of population declines in seahorses. In most cases, a life-history strategy of mate fidelity, small home ranges and high parental care make them particularly vulnerable to fishing pressure (Foster & Vincent, 2004). Population fluctuations (increases and decreases) have been observed for a number of syngnathids in the absence of fishing pressure, but are not understood. While human changes to the environment undoubtedly have an influence, underlying natural cycles may also be affecting fish populations.

Given that perfect conservation action for syngnathids is impossible, the responsibility is to move ahead with imperfect action, rather than taking no action at all (Johannes, 1998). In developing options for management, oceans need better protection in MPAs, as highlighted in the recent Convention of Biodiversity Conference of the parties (http://www.cbd.int/nagoya/outcomes/). Seahorses may generate support for such MPAs, which should benefit a wide range of marine species and habitats. As is the case for most species, however, MPAs must be complemented with additional conservation strategies that engage many different levels of society. For example, CITES provides a powerful tool to move international trade towards sustainability. If CITES is implemented effectively by all signatory countries, then many conservation issues will have been solved for seahorses and this will set an important precedent for other marine fishes. In order to meet their obligations, CITES Management Authorities need to be equipped with information and tools to support wild populations and to make decisions on exports. Developing simple, web-based tools to involve broader interest groups in monitoring syngnathid populations will help improve knowledge about their distribution and status and raise awareness of conservation issues. Strong partnerships are critical for success, whether they be with TCM traders and consumers, public aquariums, marine resource managers, communities or policy makers in range countries.

Seahorses and their syngnathid relatives can serve as flagship species for a wide range of conservation issues and solutions. They are exploited by small-scale fishers who have often run out of options. They represent thousands of small species in trawls that will never be excluded by technology. They act as attractive representatives for their shallow seas habitats. They have repeatedly been instrumental in engaging user communities, resource managers, policy-makers and the public in creative approaches to marine conservation. As three examples from Project Seahorse, (1) an alliance of *c*. 1000 small-scale fishing families was mobilized in the central Philippines to implement MPAs and reduce illegal fishing, (2) an unusual multi-stakeholder advisory council was established in Hong Kong to increase sustainability in the TCM trade and (3) seahorses were the first marine fishes of commercial importance to be listed on a CITES appendix in decades, thereby setting precedent in deploying a new international instrument. In addition, seahorses should benefit from measures such as being added to the IUCN Red List, minimum size limits, no-take MPAs and trade regulation. In general, syngnathids are seen as non-controversial animals on the political and fisheries agenda, such that policies and legislation can be introduced that would be significantly more challenging for many other marine fish species. For example, syngnathids were the first marine fishes to be brought under Australia's Wildlife Protection Act (now superseded by the Environmental Protection and Biodiversity Conservation Act; Australian Department of Environment and Heritage, 1999), thus opening the door to other native marine fish species.

A key challenge now is to use the seahorse case study to help make CITES work for marine fishes. How can the global trade in seahorses be moved to sustainable levels? And, in so doing, how can a precedent be set for other marine fishes, such that there is a growing acceptance that marine fishes are indeed wildlife as well as economic commodities? Reconciling that duality for sustainable fisheries and trade is central to the pursuit of secure wild populations. Given all the controversy, it is important to remember that a CITES Appendix II listing only requires parties to ensure that they do not damage wild populations, a fairly obvious constraint, really. Such an insistence on sustainability will need to become standard for all fisheries if species are to be assigned an improved conservation status on the IUCN Red List that they joined amidst such controversy (Vincent & Hall, 1996; Dulvy *et al*., 2005).

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SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article:

TABLE S1. The regions and countries engaged in local and international trade of live and dried seahorses, as indicated with an 'x'. Data were collected between 1996 and 2008 from three data sets: the first surveys (1996), second surveys (1998–2001) and CITES (2004–2008). Data were obtained from on-site surveys and interviews, correspondence with researchers, official government customs and trade records or CITES records (UNEP-WCMC, 2010). CITES data were based on both import (consumer country) and export (source) records. Countries that only re-export seahorses are not included.

TABLE S2. Reported legislation that pertains to live or dried syngnathid extraction and trade. Species names are indexed in the table and given in the footnotes. The information under Legislation is the summary of often complicated statutes; see references for full wording. Year refers to the most recent date the information was confirmed.

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References

Ahnesjö, I. (1996). Apparent resource competition among embryos in the brood pouch of a male pipefish. *Behavioral Ecology and Sociobiology* **38,** 167–172.

Airoldi, L. & Beck, M. W. (2007). Loss, status and trends for coastal marine habitats of Europe. *Oceanography and Marine Biology: An Annual Review* **45,** 345–405.

Alves, R. R. N. & Rosa, I. L. (2006). From cnidarians to mammals: the use of animals as remedies in fishing communities in NE Brazil. *Journal of Ethnopharmacology* **107,** 259–276.

Anderson, P. (2009). The functions of sound production in the lined seahorse, *Hippocampus erectus*, and effects of loud ambient noise on its behavior and physiology in captive environments. PhD Thesis, University of Florida, Gainesville, FL, USA.

Anderson, P. A., Berzins, I. K., Fogarty, F., Hamlin, H. J. & Guillette, L. J. Jr. (2011). Sound, stress, and seahorses: the consequences of a noisy environment to animal health. *Aquaculture* **311,** 129–138.

Barrows, A. P. W., Martin-Smith, K. M. & Baine, M. S. P. (2009). Population variables and life-history characteristics of the alligator pipefish *Syngnathoides biaculeatus*, in Papua New Guinea. *Journal of Fish Biology* **74,** 806–819.

Baum, J. K. & Vincent, A. C. J. (2005). Magnitude and inferred impacts of the seahorse trade in Latin America. *Environmental Conservation* **32,** 305–319.

Baum, J. K., Meeuwig, J. J. & Vincent, A. C. J. (2003). Bycatch of lined seahorses (*Hippocampus erectus*) in a Gulf of Mexico shrimp trawl fishery. *Fishery Bulletin* **101,** 721–731.

Bell, S. S., Hall, M. O., Soffian, S. & Madley, K. (2002). Assessing the impact of boat propeller scars on fish and shrimp utilizing seagrass beds. *Ecological Applications* **12,** 206–217.

Blaber, S. J. M., Cyrus, D. P., Albaret, J. J., Ching, C. V., Day, J. W., Elliott, M., Fonseca, M. S., Hoss, D. E., Orensanz, J., Potter, I. C. & Silvert, W. (2000). Effects of fishing on the structure and functioning of estuarine and nearshore ecosystems. *ICES Journal of Marine Science* **57,** 590–602.

Blundell, A. G. & Masica, M. B. (2005). Discrepancies in reported levels of international wildlife trade. *Conservation Biology* **19,** 2020–2025.

Boisseau, J. (1967). Les regulations hormonales de l'incubation chez un vertebre male: recherches sur la reproduction de l'hippocampe. PhD Thesis, Universite de Bordeaux, Bordeaux, France.

Bologna, P. A. X. (2007). Impact of differential predation potential on eelgrass (*Zostera marina*) faunal community structure. *Aquatic Ecology* **41,** 221–229.

Braga Gonçalves, I. (2010). Egg size evolution and paternal care in pipefishes. PhD Thesis, University of Gothenburg, Gothenburg, Sweden.

Bruckner, A. W., Field, J. D. & Daves, N. (Eds) (2005). The Proceedings of the International Workshop on CITES Implementation for Seahorse Conservation and Trade. *NOAA Technical Memorandum* **NMFS-OPR-27**.

Burfeind, D. D., Tibbetts, I. R. & Udy, J. W. (2009). Habitat preference of three common fishes for seagrass, *Caulerpa taxifolia*, and unvegetated substrate in Moreton Bay, Australia. *Environmental Biology of Fishes* **84,** 317–322.

Clynick, B. G. (2008). Harbour swimming nets: a novel habitat for seahorses. *Aquatic Conservation: Marine and Freshwater Ecosystems* **18,** 483–492.

Connolly, R. M. (1994). A comparison of fish assemblages from seagrass and unvegetated areas of a southern Australian estuary (South Australia). *Australian Journal of Marine and Freshwater Research* **45,** 1033–1044.

Courchamp, F., Angulo, E., Rivalan, P., Hall, R. J., Signoret, L., Bull, L. & Meinard, Y. (2006). Rarity value and species extinction: the anthropogenic Allee effect. *PLoS Biology* **4,** 2405–2410.

Curtis, J. M. R. & Vincent, A. C. J. (2005). Distribution of sympatric seahorse species along a gradient of habitat complexity in a seagrass-dominated community. *Marine Ecology Progress Series* **291,** 81–91.

Curtis, J. M. R., Ribeiro, J., Erzini, K. & Vincent, A. C. J. (2007). A conservation trade-off? Interspecific differences in seahorse responses to experimental changes in fishing effort. *Aquatic Conservation: Marine and Freshwater Ecosystems* **17,** 468–484.

Dauwe, B. (1992). Ecology of the seahorse *Hippocampus reidi* on the coral reefs of Bonaire (NA): habitat use, reproduction and interspecific interactions. [Ecologie van het zeepaardje *Hippocampus reidi* (Syngnathidae) op het koraalrif van Bonaire (NA): Habitatgebruik, reproductie en interspecifieke interacties]. MSc Thesis, Rijksuniversiteit Groningen, The Netherlands.

Davis, M. W. (2002). Key principles for understanding fish bycatch discard mortality. *Canadian Journal of Fisheries and Aquatic Sciences* **59,** 1834–1843.

Dodt, N. (2005). *Fisheries Long Term Monitoring Program. Syngnathids in the East Coast Trawl Fishery: A Review and Trawl Survey.* Brisbane: Department of Primary Industries and Fisheries.

Domínguez, L. M. & Ferrer, F. O. (2009). Aquaculture and marine biodiversity boost: case examples from the Canary Islands. *Water Resources Management* **97,** 97–102.

Duffy, K. C. & Baltz, D. M. (1998). Comparison of fish assemblages associated with native and exotic submerged macrophytes in the Lake Pontchartrain estuary, USA. *Journal of Experimental Marine Biology and Ecology* **223,** 199–221.

Dulvy, N. K., Sadovy, Y. & Reynolds, J. D. (2003). Extinction vulnerability in marine populations. *Fish and Fisheries* **4,** 25–64.

Dulvy, N. K., Jennings, S., Goodwin, N. B., Grant, A. & Reynolds, J. D. (2005). Comparison of threat and exploitation status in North East Atlantic marine populations. *Journal of Applied Ecology* **42,** 883–891.

Dye, A. H., Branch, G. M., Castilla, J. C. & Bennett, B. A. (1994). Biological options for the management of the exploitation of intertidal and subtidal resources. In *Rocky Shores: Exploitation in Chile and South Africa* (Siegfried, W. R., ed.), pp. 131–154. Berlin: Springer-Verlag.

Dzyuba, B., Van Look, K. J. W., Cliffe, A., Koldewey, H. J. & Holt, W. V. (2006). Effect of parental age and associated size on fecundity, growth and survival in the yellow seahorse *Hippocampus kuda*. *Journal of Experimental Biology* **209,** 3055–3061.

Evanson, M., Foster, S. J. & Vincent, A. C. J. (2011). Tracking the international trade of seahorses (*Hippocampus* species) – the importance of CITES. *Fisheries Centre Research Reports 19(2)*. Vancouver, BC: Fisheries Centre, University of British Columbia.

FAO (2003). Status and trends in mangrove area extent worldwide. In *Forest Resources Assessment Working Paper 63* (Wilkie, M. L. & Fortuna, S., eds). Rome: Forest Resources Division, FAO.

Fleischer, D., Schaber, M. & Piepenburg, D. (2007). Atlantic snake pipefish (*Entelurus aequoreus*) extends its northward distribution range to Svalbard (Arctic Ocean). *Polar Biology* **30,** 1359–1362.

Fonseca, M. S., Kenworthy, W. J. & Thayer, G. W. (1998). *Guidelines for the Conservation and Restoration of Seagrasses in the United States and Adjacent Waters.* Silver Spring, MD: NOAA Coastal Ocean Office.

Foster, R. & Gomon, M. F. (2010). A new seahorse (Teleostei: Syngnathidae: *Hippocampus*) from south-western Australia. *Zootaxa* **2613,** 61–68.

Foster, S. J. & Vincent, A. C. J. (2004). Life history and ecology of seahorses: implications for conservation and management. *Journal of Fish Biology* **65,** 1–61.

Foster, S. J. & Vincent, A. C. J. (2005). Enhancing sustainability of the international trade in seahorses with a single minimum size limit. *Conservation Biology* **19,** 1044–1050.

Gamito, S. (2008). Three main stressors acting on the Ria Formosa lagoonal system (Southern Portugal): physical stress, organic matter pollution and the land-ocean gradient. *Estuarine, Coastal and Shelf Science* **77,** 710–720.

Gerber, L. R., Botsford, L. W., Hastings, A., Possingham, H. P., Gaines, S. D., Palumbi, S. R. & Andelman, S. (2003). Population models for marine reserve design: a retrospective and prospective synthesis. *Ecological Applications* **13,** 47–64.

Giles, B. G., Truong, S. K., Do, H. H. & Vincent, A. C. J. (2006). The catch and trade of seahorses in Vietnam. *Biodiversity and Conservation* **15,** 2497–2513.

Grey, M., Blais, A.-M. & Vincent, A. C. J. (2005). Magnitude and trends of marine fish curio imports to the USA. *Oryx* **39,** 413–420.

Harasti, D., Glasby, T. M. & Martin-Smith, K. M. (2010). Striking a balance between retaining populations of protected seahorses and maintaining swimming nets. *Aquatic Conservation: Marine and Freshwater Ecosystems* **20,** 159–166.

Hardy, J. D. (1978). Development of fishes of the mid-Atlantic bight an atlas of egg, larval and juvenile stages: Volume II Anguillidae through Syngnathidae. *U.S. Fish and Wildlife Service, Biological Services Program* **FWS/OBS-78/12**.

Harris, M. P., Beare, D., Toresen, R., Nøttestad, L., Kloppmann, M., Dörner, H., Peach, K., Rushton, D. R. A., Foster-Smith, J. & Wanless, S. (2007). A major increase in snake pipefish (*Entelurus aequoreus*) in northern European seas since 2003: potential implications for seabird breeding success. *Marine Biology* **151,** 973–983.

Hesketh, T. & Zhu, W. X. (1997). Health in China: traditional Chinese medicine: one country, two systems. *BMJ* **315,** 115–117.

Hutchings, J. A. & Reynolds, J. D. (2004). Marine fish population collapses: consequences for recovery and extinction risk. *Bioscience* **54,** 297–309.

Huxley, T. (1883). Inaugural address. *International Fisheries Exhibition Literature* **4,** 1–19.

Institute for Science and Technology of Vietnam (2007). *Vietnam Red Book.* Hanoi: Science and Technics Publishing House.

Isaac, N. J. B., Turvey, S. T., Collen, B., Waterman, C. & Baillie, J. E. M. (2007). Mammals on the EDGE: conservation priorities based on threat and phylogeny. *PLoS ONE* **2,** 296.

IUCN (1998). *Guidelines for Re-introduction.* Gland: IUCN/SSC Re-introduction Specialist Group.

Jackson, J. B. C., Kirby, M. X., Berger, W. H., Bjorndal, K. A., Botsford, L. W., Bourque, B. J., Bradbury, R. H., Cooke, R., Erlandson, J. & Estes, J. A. (2001). Historical overfishing and the recent collapse of coastal ecosystems. *Science* **293,** 629–638.

James, P. L. & Heck, K. L. (1994). The effects of habitat complexity and light intensity on ambush predation within a simulated seagrass habitat. *Journal of Experimental Marine Biology and Ecology* **176,** 187–200.

Jardas, I., Pallaoro, A. & Vrgoc, N. (2007). *Croatian Red List: Saltwater Fish*. Split: Institute of Oceanography and Fisheries.

Johannes, R. E. (1998). The case for data-less marine resource management: examples from tropical nearshore finfisheries. *Trends in Ecology & Evolution* **13,** 243–246.

Jones, M. V. & West, R. J. (2005). Spatial and temporal variability of seagrass fishes in intermittently closed and open coastal lakes in southeastern Australia. *Estuarine, Coastal and Shelf Science* **64,** 277–288.
Kaiser, H., Britz, P., Endemann, F., Haschick, R., Jones, C. L. W., Koranteng, B., Kruger, D. P., Lockyear, J. F., Oellermann, L. K., Olivier, A. P., Rouhani, Q. & Hecht, T. (1997). Development of technology for ornamental fish aquaculture in South Africa. *South African Journal of Science* **93,** 351–354.
Kirby, R. R., Johns, D. G. & Lindley, J. A. (2006). Fathers in hot water: rising sea temperatures and a Northeastern Atlantic pipefish baby boom. *Biology Letters* **2,** 597–600.
Koldewey, H. J. & Martin-Smith, K. M. (2010). A global review of seahorse aquaculture. *Aquaculture* **302,** 131–152.
Koldewey, H. & Zimmerman, B. (2007). Sustainable acquisition in aquariums. *EAZA News* **60,** 30–32.
Kuiter, R. H. (2001). Revision of the Australian seahorses of the genus *Hippocampus* (Syngnathiformes: Syngnathidae) with descriptions of nine new species. *Records of the Australian Museum* **53,** 293–340.
Kuiter, R. H. (2009). *Seahorses and Their Relatives.* Seaford, Victoria: Aquatic Photographics.
Kvarnemo, C., Moore, G. I., Jones, A. G., Nelson, W. S. & Avise, J. C. (2000). Monogamous pair bonds and mate switching in the Western Australian seahorse *Hippocampus subelongatus*. *Journal of Evolutionary Biology* **13,** 882–888.
Kvarnemo, C., Moore, G. I. & Jones, A. G. (2007). Sexually selected females in the monogamous Western Australian seahorse. *Proceedings of the Royal Society B* **274,** 521–525.
Laffaille, P., Feunteun, E. & Lefeuvre, J. C. (2000). Composition of fish communities in a European macrotidal salt marsh (the Mont Saint-Micheal Bay, France). *Estuarine, Coastal and Shelf Science* **51,** 429–438.
Larkin, S. L. (2003). The U.S. wholesale marine ornamental market: trade, landings, and market options. In *Marine Ornamental Species: Collection, Conservation and Culture* (Cato, J. C. & Brown, C. L., eds), pp. 77–93. Ames, IA: Iowa State Press.
Lazzari, M. & Able, K. (1990). Northern pipefish, *Syngnathus fuscus*, occurrences over the Mid-Atlantic Bight continental shelf: evidence of seasonal migration. *Environmental Biology of Fishes* **27,** 177–185.
Lee, J. (1998*a*). TCM use of marine medicinals: content and context. In *The Management and Culture of Marine Species used in Traditional Medicines* (Moreau, M.-A., Hall, H. J. & Vincent, A. C. J., eds), pp. 81–83. Cebu City: Project Seahorse.
Lee, Y.-J. (1998*b*). The state of seahorses as herbs in Korean oriental medicine. In *The Management and Culture of Marine Species Used in Traditional Medicines* (Moreau, M.-A., Hall, H. J. & Vincent, A. C. J., eds), pp. 92–94. Cebu City: Project Seahorse.
Leslie, H. M. (2005). A synthesis of marine conservation planning approaches. *Conservation Biology* **19,** 1701–1713.
Levin, P. S. & Stunz, G. W. (2005). Habitat triage for exploited fishes: can we identify essential "Essential Fish Habitat?" *Estuarine, Coastal and Shelf Science* **64,** 70–78.
Lin, Q., Lu, J. Y., Gao, Y. L., Shen, L., Cai, J. & Luo, J. N. (2006). The effect of temperature on gonad, embryonic development and survival rate of juvenile seahorses, *Hippocampus kuda* Bleeker. *Aquaculture* **254,** 701–713.
Lin, Q., Lin, J. & Zhang, D. (2008). Breeding and juvenile culture of the lined seahorse, *Hippocampus erectus* Perry, 1810. *Aquaculture* **277,** 287–292.
Linton, J. R. & Soloff, B. L. (1964). The physiology of the brood pouch of the male sea horse *Hippocampus erectus*. *Bulletin of Marine Science of the Gulf and Caribbean* **14,** 45–61.
Livingston, R. J. (1984). Trophic response of fishes to habitat variability in coastal seagrass systems. *Ecology* **65,** 1258–1275.
Lockyear, J. F., Hecht, T., Kaiser, H. & Teske, P. R. (2006). The distribution and abundance of the endangered Knysna seahorse *Hippocampus capensis* (Pisces: Syngnathidae) in South African estuaries. *African Journal of Aquatic Science* **31,** 275–283.
Lopez, M. A. & Gisbert, E. (2009). Evaluation of a by-catch reduction device for glass eel fishing traps. *Fisheries Management and Ecology* **16,** 438–447.

Lotze, H. K., Lenihan, H. S., Bourque, B. J., Bradbury, R. H., Cooke, R. G., Kay, M. C., Kidwell, S. M., Kirby, M. X., Peterson, C. H. & Jackson, J. B. C. (2006). Depletion, degradation, and recovery potential of estuaries and coastal seas. *Science* **312,** 1806–1809.

Lourie, S. A., Vincent, A. C. J. & Hall, H. J. (1999). *Seahorses: An Identification Guide to the World's Species and Their Conservation*. London: Project Seahorse.

Lourie, S. A., Foster, S. J., Cooper, E. W. T. & Vincent, A. C. J. (2004). *A Guide to the Identification of Seahorses.* Washington, DC: University of British Columbia and World Wildlife Fund.

Lundquist, C. J. & Granek, E. F. (2005). Strategies for successful marine conservation: integrating socioeconomic, political, and scientific factors. *Conservation Biology* **19,** 1771–1778.

Marcus, J. E., Samoilys, M. A., Meeuwig, J. J., Villongco, Z. A. D. & Vincent, A. C. J. (2007). Benthic status of near-shore fishing grounds in the central Philippines and associated seahorse densities. *Marine Pollution Bulletin* **54,** 1483–1494.

Marshall, D. (2004). The interactions of the big-bellied seahorse with artificial structures. Honours Thesis, University of Tasmania, Australia.

Martin-Smith, K. M. & Vincent, A. C. J. (2005). Seahorse declines in the Derwent estuary, Tasmania in the absence of fishing pressure. *Biological Conservation* **123,** 533–545.

Martin-Smith, K. M. & Vincent, A. C. J. (2006). Exploitation and trade in Australian seahorses, pipehorses, sea dragons and pipefishes (Family Syngnathidae). *Oryx* **40,** 141–151.

Martin-Smith, K. M., Lam, T. F. & Lee, S. K. (2003). Trade in pipehorses *Solegnathus* spp. for traditional medicine in Hong Kong. *Traffic Bulletin* **19,** 139–148.

Martin-Smith, K. M., Samoilys, M. A., Meeuwig, J. J. & Vincent, A. C. J. (2004). Collaborative development of management options for an artisanal fishery for seahorses in the central Philippines. *Ocean & Coastal Management* **47,** 165–193.

Masonjones, H. D., Rose, E., McRae, L. B. & Dixson, D. L. (2010). An examination of the population dynamics of syngnathid fishes within Tampa Bay, Florida, USA. *Current Zoology* **56,** 118–133.

May, B. & Tomoda, T. (2002). Seahorses in the *Ben cao gang mu* and in contemporary Chinese medicine. *Journal of the Australian Chinese Medicine Education and Research Council* **7,** 1–12.

McPherson, J. M. & Vincent, A. C. J. (2004). Assessing East African trade in seahorse species as a basis for conservation under international controls. *Aquatic Conservation: Marine and Freshwater Ecosystems* **14,** 521–538.

Meeuwig, J. J., Do, H. H., Troung, S. K., Job, S. D. & Vincent, A. C. J. (2006). Quantifying non-target seahorse fisheries in central Vietnam. *Fisheries Research* **81,** 149–157.

Meyer, D. L., Fonseca, M. S., Murphey, P. L., McMichael, R. H. Jr., Byerly, M. M., LaCroix, M. W., Whitfield, P. E. & Thayer, G. W. (1999). Effects of live-bait shrimp trawling on seagrass beds and fish bycatch in Tampa Bay, Florida. *Fishery Bulletin* **97,** 193–199.

Mobley, K. B., Small, C. M. & Jones, A. G. (2011). The genetics and genomics of Syngnathidae: pipefishes, seahorses and seadragons. *Journal of Fish Biology* **78**, 1624–1646. doi:10.1111/j.1095-8649.2011.02967.x

Monteiro, N. M., Almada, V. C. & Vieira, M. N. (2005). Implications of different brood pouch structures in syngnathid reproduction. *Journal of the Marine Biological Association of the United Kingdom* **85,** 1235–1241.

Moreau, M.-A., Hall, H. J. & Vincent, A. C. J. (1998). *Proceedings of the First International Workshop on the Management and Culture of Marine Species Used in Traditional Medicines.* Cebu City: Project Seahorse.

Morgan, S. K. (2007). The ontogenetic ecology and conservation of exploited seahorses: Chapter 2. Planktonic dispersal of tropical seahorses (*Hippocampus spinosissimus* Weber, 1913 and *Hippocampus comes* Cantor, 1850). PhD Thesis, McGill University, Montreal, QU, Canada.

Morgan, S. K. & Vincent, A. C. J. (2007). The ontogeny of habitat associations in the tropical tiger tail seahorse *Hippocampus comes* Cantor, 1850. *Journal of Fish Biology* **71,** 701–724.

Mouillot, D., Dumay, O. & Tomasini, J. A. (2007). Limiting similarity, niche filtering and functional diversity in coastal lagoon fish communities. *Estuarine, Coastal and Shelf Science* **71,** 443–456.

Mumby, P. J. & Steneck, R. S. (2008). Coral reef management and conservation in light of rapidly evolving ecological paradigms. *Trends in Ecology & Evolution* **23,** 555–563.

Murugan, A., Dhanya, S., Rajagopal, S. & Balasubramanian, T. (2008). Seahorses and pipefishes of the Tamil Nadu coast. *Current Science* **95,** 253–260.

Myers, R. A. & Worm, B. (2003). Rapid worldwide depletion of predatory fish communities. *Nature* **423,** 280–283.

Naylor, R. L., Goldburg, R. J., Primavera, J. H., Kautsky, N., Beveridge, M. C. M., Clay, J., Folke, C., Lubchenco, J., Mooney, H. & Troell, M. (2000). Effect of aquaculture on world fish supplies. *Nature* **405,** 1017–1024.

Nguyen, V. L. & Do, H. H. (1998). Biological parameters of two exploited seahorse species in a Vietnamese fishery. In *Third International Conference on the Marine Biology of the South China Sea* (Morton, B., ed.), pp. 449–464. Aberdeen: Hong Kong University Press.

Nijman, V. (2010). An overview of international wildlife trade from Southeast Asia. *Biodiversity and Conservation* **19,** 1101–1114.

O'Donnell, K. P., Pajaro, M. G. & Vincent, A. C. J. (2010). How does the accuracy of fisher knowledge affect seahorse conservation status? *Animal Conservation* **13,** 526–533.

Olden, J. D., Hogan, Z. S. & Vander Zanden, M. J. (2007). Small fish, big fish, red fish, blue fish: size biased extinction risk of the world's freshwater and marine fishes. *Global Ecology and Biogeography* **16,** 694–701.

Orth, R. J., Carruthers, T. J. B., Dennison, W. C., Duarte, C. M., Fourqurean, J. W., Heck, K. L. Jr., Hughes, A. R., Kendrick, G. A., Kenworthy, W. J., Olyarnik, S., Short, F. T., Waycott, M. & Williams, S. L. (2006). A global crisis for seagrass ecosystems. *Bioscience* **56,** 987–996.

Pajaro, M. G., Vincent, A. C. J., Buhat, D. & Perante, N. (1998). The role of seahorse fishers in conservation and management. In *Proceedings of the First International Symposium Marine Conservation* (Lunn, K. E., Boehm, J. R., Hall, H. J. & Vincent, A. C. J., eds), pp. 118–126. Chicago, IL: Hong Kong Marine Conservation Society and John G. Shedd Aquarium.

Parry-Jones, R. & Vincent, A. (1998). Can we tame wild medicine? *New Scientist* **157,** 26–29.

Parry, M. L., Canziani, O. F., Palutikof, J. P. Linden, P. J. V. D. & Hanson, C. E. (2007). Technical summary. Climate change 2007: impacts, adaptation and vulnerability. In *Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change* (Parry, M. L., Canziani, O. F., Palutikof, J. P., Linden, P. J. V. D. & Hanson, C. E., eds), pp. 23–78. Cambridge: Cambridge University Press.

Partridge, C., Cazalas, C., Rozelle, J., Hemming, J. & Boettcher, A. (2004). Small-scale captive breeding of a euryhaline pipefish. *World Aquaculture* **35,** 51–54.

Payne, M. F., Rippingale, R. J. & Longmore, R. B. (1998). Growth and survival of juvenile pipefish (*Stigmatopora argus*) fed live copepods with high and low HUFA content. *Aquaculture* **167,** 237–245.

Penning, M., Reid, G. M., Koldewey, H., Dick, G., Andrews, B., Arai, K., Garratt, P., Gendron, S., Lange, J., Tanner, K., Tonge, S., Sande, P. V. D., Warmolts, D. & Gibson, C. (Eds) (2009). *Turning the Tide: A Global Aquarium Strategy for Conservation and Sustainability.* Bern: World Association of Zoos and Aquariums.

Perante, N. C., Vincent, A. C. J. & Pajaro, M. G. (1998). Demographics of the seahorse *Hippocampus comes* in the central Philippines. In *Third International Conference on the Marine Biology of the South China Sea* (Morton, B., ed.), pp. 439–448. Aberdeen: Hong Kong University Press.

Perry, A. L., Lunn, K. E. & Vincent, A. C. J. (2010). Fisheries, large-scale trade, and conservation of seahorses in Malaysia and Thailand. *Aquatic Conservation: Marine and Freshwater Ecosystems* **20,** 464–475.

Philippine Department of Agriculture (1998). The Philippine Fisheries Code of 1998: Republic Act No. 8550.

Polidoro, B. A., Carpenter, K. E., Collins, L., Duke, N. C., Ellison, A. M., Ellison, J. C., Farnsworth, E. J., Fernando, E. S., Kathiresan, K., Koedam, N. E., Livingstone, S. R., Miyagi, T., Moore, G. E., Ngoc Nam, V., Ong, J. E., Primavera, J. H., Salmo, S. G. III, Sanciangco, J. C., Sukardjo, S., Wang, Y. & Yong, J. W. H. (2010). The loss of species: mangrove extinction risk and geographic areas of global concern. *PLoS ONE* **5,** e10095.

Power, M. & Attrill, M. J. (2003). Long-term trends in the estuarine abundance of Nilsson's pipefish (*Syngnathus rostellatus* Nilsson). *Estuarine, Coastal and Shelf Science* **57,** 325–333.

Qian, Z. J., Ryu, B., Kim, M. M. & Kim, S. K. (2008). Free radical and reactive oxygen species scavenging activities of the extracts from seahorse, *Hippocampus kuda* Bleeker. *Biotechnology and Bioprocess Engineering* **13,** 705–715.

Redding, D. W. & Mooers, A. Ø. (2006). Incorporating evolutionary measures into conservation prioritization. *Conservation Biology* **20,** 1670–1678.

Reynolds, J. D., Dulvy, N. K., Goodwin, N. B. & Hutchings, J. A. (2005). Biology of extinction risk in marine fishes. *Proceedings of the Royal Society B* **272,** 2337–2344.

Ripley, J. L. & Foran, C. M. (2007). Influence of estuarine hypoxia on feeding and sound production by two sympatric pipefish species (Syngnathidae). *Marine Environmental Research* **63,** 350–367.

Roberts, C. M. & Hawkins, J. P. (1999). Extinction risk in the sea. *Trends in Ecology & Evolution* **14,** 241–246.

Rosa, I., Alves, R. R. N., Bonifácio, K. M., Mourão, J. S., Osório, F. M., Oliveira, T. P. R. & Nottingham, M. C. (2005). Fishers knowledge and seahorse conservation in Brazil. *Journal of Ethnobiology and Ethnomedicine* **1,** 12–26.

Rosa, I. L., Sampaio, C. L. S. & Barros, A. T. (2006). Collaborative monitoring of the ornamental trade of seahorses and pipefishes (Teleostei: Syngnathidae) in Brazil: Bahia State as a case study. *Neotropical Ichthyology* **4,** 247–252.

Rosa, I. L., Oliveira, T. P. R., Castro, A. L. C., Moraes, L. E. D. S., Xavier, J. H. A., Nottingham, M. C., Dias, T. L. P., Bruto-Costa, L. V., Araújo, M. E., Birolo, A. B., Mai, A. C. G. & Monteiro-Neto, C. (2007). Population characteristics, space use and habitat associations of the seahorse *Hippocampus reidi* (Teleostei: Syngnathidae). *Neotropical Ichthyology* **5,** 405–414.

Ross, D. J., Johnson, C. R., Hewitt, C. L. & Ruiz, G. M. (2004). Interaction and impacts of two introduced species on a soft-sediment marine assemblage in SE Tasmania. *Marine Biology* **144,** 747–756.

Rowley, R. J. (1994). Marine reserves in fisheries management. *Aquatic Conservation: Marine and Freshwater Ecosystems* **4,** 233–254.

Russell, I. A. (1994). Mass mortality of marine and estuarine fish in the Swartvlei and Wilderness Lake Systems, Southern Cape. *South African Journal of Aquatic Science* **20,** 93–96.

Ryer, C. H. (1988). Pipefish foraging: effects of fish size, prey size and altered habitat complexity. *Marine Ecology Progress Series* **48,** 37–45.

Saarman, N. P., Louie, K. D. & Hamilton, H. (2010). Genetic differentiation across eastern Pacific oceanographic barriers in the threatened seahorse *Hippocampus ingens*. *Conservation Genetics* **11,** 1189–2000.

Sabri, S. B. (2009). Impact of water quality on seagrass meadow at Tanjung Kupang. MEng Thesis, Universiti Teknologi Malaysia, Johor, Malaysia.

Sadovy, Y. J. & Vincent, A. C. J. (2002). Ecological issues and the trades in live reef fishes. In *Coral Reef Fishes* (Sale, P., ed.), pp. 391–420. San Diego, CA: Academic Press.

Salafsky, N., Cauley, H., Balachander, G., Cordes, B., Parks, J., Margoluis, C., Bhatt, S., Encarnacion, C., Russell, D. & Margoluis, R. (2001). A systematic test of an enterprise strategy for community-based biodiversity conservation. *Conservation Biology* **15,** 1585–1595.

Salin, K. R., Yohannan, T. M. & Mohanakumaran Nair, C. (2005). Fisheries and trade of seahorses, *Hippocampus* spp., in southern India. *Fisheries Management and Ecology* **12,** 269–273.

Samoilys, M. A., Martin-Smith, K. M., Giles, B. G., Cabrera, B., Anticamara, J. A., Brunio, E. O. & Vincent, A. C. J. (2007). Effectiveness of five small Philippines' coral reef reserves for fish populations depends on site-specific factors, particularly enforcement history. *Biological Conservation* **136,** 584–601.

Sanchez-Camara, J., Booth, D. J., Murdoch, J., Watts, D. & Turon, X. (2006). Density, habitat use and behaviour of the weedy seadragon *Phyllopteryx taeniolatus* (Teleostei: Syngnathidae) around Sydney, New South Wales, Australia. *Marine and Freshwater Research* **57,** 737–745.

Sanders, J. G., Cribbs, J. E., Fienberg, H. G., Hulburd, G. C., Katz, L. S. & Palumbi, S. R. (2008). The tip of the tail: molecular identification of seahorses for sale in apothecary shops and curio stores in California. *Conservation Genetics* **9,** 65–71.

Scales, H. (2009). *Poseidon's Steed: The Story of Seahorses, From Myth to Reality.* New York, NY: Penguin Group Inc.

Scales, H. (2010). Advances in the ecology, biogeography and conservation of seahorses (genus *Hippocampus*). *Progress in Physical Geography* **34,** 443–458.

Shokri, M. R., Gladstone, W. & Jelbart, J. (2009). The effectiveness of seahorses and pipefish (Pisces: Syngnathidae) as a flagship group to evaluate the conservation value of estuarine seagrass beds. *Aquatic Conservation: Marine and Freshwater Ecosystems* **19,** 588–595.

Sievanen, L., Crawford, B., Pollnac, R. & Lowe, C. (2005). Weeding through assumptions of livelihood approaches in ICM: seaweed farming in the Philippines and Indonesia. *Ocean and Coastal Management* **48,** 297–313.

Simpson, S. D., Meekan, M. G., Jeffs, A., Montgomery, J. C. & McCauley, R. D. (2008). Settlement-stage coral reef fish prefer the higher-frequency invertebrate-generated audible component of reef noise. *Animal Behaviour* **75,** 1861–1868.

Steneck, R. S., Graham, M. H., Bourque, B. J., Corbett, D., Erlandson, J. M., Estes, J. A. & Tegner, M. J. (2002). Kelp forest ecosystems: biodiversity, stability, resilience and future. *Environmental Conservation* **29,** 436–459.

Strawn, K. (1953). A study of the dwarf seahorse, *Hippocampus regulus* Ginsburg at Cedar Key, Florida. MSc Thesis, University of Florida, Gainesville, FL, USA.

Sundin, J., Berglund, A. & Rosenqvist, G. (2010). Turbidity hampers mate choice in a pipefish. *Ethology* **116,** 713–721.

Swartz, W., Sala, E., Tracey, S., Watson, R. & Pauly, D. (2010). The spatial expansion and ecological footprint of fisheries (1950 to Present). *PLoS ONE* **5,** 5995–6000.

Tang, W. C. (1987). Chinese medicinal materials from the sea. *Abstracts of Chinese Medicines* **1,** 571–600.

Teske, P. R., Lourie, S. A., Matthee, C. A. & Green, D. M. (2007). *Hippocampus queenslandicus* Horne, 2001 – a new seahorse species or yet another synonym? *Australian Journal of Zoology* **55,** 139–145.

Thoney, D. A., Warmolts, D. I. & Andrews, C. (2003). Acquisition of fishes and aquatic invertebrates for zoological collections. Is there a future? *Zoo Biology* **22,** 519–527.

Tipton, K. & Bell, S. S. (1988). Foraging patterns of two syngnathid fishes: importance of harpacticoid copepods. *Marine Ecology Progress Series* **47,** 31–43.

Turvey, S. T., Barrett, L. A., Hao, Y. J., Zhang, L., Zhang, X. Q., Wang, X. Y., Huang, Y. D., Zhou, K. Y., Hart, T. & Wang, D. (2010). Rapidly shifting baselines in Yangtze fishing communities and local memory of extinct species. *Conservation Biology* **24,** 778–787.

Valiela, I., Bowen, J. L. & York, J. K. (2001). Mangrove forests: one of the world's threatened major tropical environments. *Bioscience* **51,** 807–815.

Veron, J. E. N., Hoegh-Guldberg, O., Lenton, T. M., Lough, J. M., Obura, D. O., Pearce-Kelly, P., Sheppard, C. R. C., Spalding, M., Stafford-Smith, M. G. & Rogers, A. D. (2009). The coral reef crisis: the critical importance of $<$ 350 ppm CO_2. *Marine Pollution Bulletin* **58,** 1428–1436.

Vincent, A. C. J. (1990). Reproductive ecology of seahorses. PhD Thesis, Cambridge University, Cambridge, UK.

Vincent, A. C. J. (1994). Operational sex ratios in seahorses. *Behaviour* **128,** 153–167.

Vincent, A. C. J. (1996). *The International Trade in Seahorses.* Cambridge: TRAFFIC International.

Vincent, A. C. J. (2008). Reconciling fisheries with conservation on coral reefs: the world as an onion. In *Reconciling Fisheries With Conservation: Proceedings of the Fourth*

World Fisheries Congress (Nielsen, J. L., Dodson, J. J., Friedland, K., Hamon, T. R., Musick, J. & Verspoor, E., eds), pp. 1435–1467. Bethesda, MD: American Fisheries Society.

Vincent, A. C. J. & Hall, H. J. (1996). The threatened status of marine fishes. *Trends in Ecology & Evolution* **11,** 360–361.

Vincent, A. C. J. & Koldewey, H. J. (2006). An uncertain future for seahorse aquaculture in conservation and economic contexts. In *Proceedings of the Regional Technical Consultation on Stock Enhancement of Species Under International Concern. Southeast Asian Fisheries Development Center* (Primavera, J. H., ed.), pp. 71–84. Iloilo: SEAFDEC/ADQ.

Vincent, A. C. J. & Pajaro, M. G. (1997). Community-based management for a sustainable seahorse fishery. In *The 2nd World Fisheries Congress* (Hancock, D. A., Smith, D. C., Grant, A. & Beumer, J. P., eds), pp. 761–766. Brisbane: CSIRO Publishing.

Vincent, A. C. J. & Sadler, L. M. (1995). Faithful pair bonds in wild seahorses, *Hippocampus whitei*. *Animal Behaviour* **50,** 1557–1569.

Vincent, A., Ahnesjö, I. & Berglund, A. (1994). Operational sex ratios and behavioural sex differences in a pipefish population. *Behavioral Ecology and Sociobiology* **34,** 435–442.

Vincent, A. C. J., Evans, K. L. & Marsden, A. D. (2005). Home range behaviour of the monogamous Australian seahorse, *Hippocampus whitei*. *Environmental Biology of Fishes* **72,** 1–12.

Vincent, A. C. J., Meeuwig, J. J., Pajaro, M. G. & Perante, N. C. (2007). Characterizing a small-scale, data-poor, artisanal fishery: seahorses in the central Philippines. *Fisheries Research* **86,** 207–215.

Vincent, A. C. J., Giles, B. G., Czembor, C. A. & Foster, S. J. (2011). Trade in seahorses and other syngnathids in non-Asian countries (1998–2001). *Fisheries Centre Research Reports 19(1)*. Vancouver, BC: Fisheries Centre, University of British Columbia.

Wabnitz, C., Taylor, M., Green, E. & Razak, T. (2003). *From Ocean to Aquarium: The Global Trade in Marine Ornamental Species.* Cambridge: UNEP-WCMC.

Waycott, M., Duarte, C. M., Carruthers, T. J. B., Orth, R. J., Dennison, W. C., Olyarnik, S., Calladine, A., Fourqurean, J. W., Heck, K. L. Jr., Hughes, A. R., Kendrick, G. A., Kenworthy, W. J., Short, F. T. & Williams, S. L. (2009). Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Proceedings of the National Academy of Sciences* **106,** 12377–12381.

Weir, L. K. & Grant, J. W. A. (2005). Effects of aquaculture on wild fish populations: a synthesis of data. *Environmental Reviews* **13,** 145–168.

Wilkinson, C. (2008). *Status of Coral Reefs of the World.* Townsville: Global Coral Reef Monitoring Network Reef and Rainforest Research Centre.

Wilson, A. B., Ahnesjö, I., Vincent, A. C. J. & Meyer, A. (2003). The dynamics of male brooding, mating patterns, and sex roles in pipefishes and seahorses (family Syngnathidae). *Evolution* **57,** 1374–1386.

Wong, J. M. & Benzie, J. A. H. (2003). The effects of temperature, *Artemia* enrichment, stocking density and light on the growth of juvenile seahorses, *Hippocampus whitei* (Bleeker, 1855), from Australia. *Aquaculture* **228,** 107–121.

World Conservation Monitoring Centre (2004). *A Guide to Interpreting Outputs From the CITES Trade Database Version 6.0.* Cambridge: World Conservation Monitoring Centre.

Worm, B., Barbier, E. B., Beaumont, N., Duffy, J. E., Folke, C., Halpern, B. S., Jackson, J. B. C., Lotze, H. K., Micheli, F., Palumbi, S. R., Sala, E., Selkoe, K. A., Stachowicz, J. J. & Watson, R. (2006). Impacts of biodiversity loss on ocean ecosystem services. *Science* **314,** 787–790.

Xu, J. & Yang, Y. (2009). Traditional Chinese medicine in the Chinese health care system. *Health Policy* **90,** 133–139.

Xu, Y. J., Lin, J. D. & Chen, S. (2010). Polyculture of the lined seahorse, *Hippocampus erectus* Perry, 1810 with two species of macroalgae in aquaria. *Acta Oceanologica Sinica* **29,** 26–32.

Yasué, M., Kaufman, L. & Vincent, A. C. J. (2010). Assessing ecological changes in and around marine reserves using community perceptions and biological surveys. *Aquatic Conservation: Marine and Freshwater Ecosystems* **20,** 407–418.

Yasué, M., Nellas, A. & Vincent, A. C. J. (in press). Seahorses helped drive creation of marine protected areas, so what did these protected areas do for the seahorse? *Environmental Conservation.*

York, P. H., Booth, D. J., Glasby, T. M. & Pease, B. C. (2006). Fish assemblages in habitats dominated by *Caulerpa taxifolia* and native seagrasses in south-eastern Australia. *Marine Ecology Progress Series* **312,** 223–234.

Zhu, Y. P. & Woerdenbag, H. J. (1995). Traditional Chinese herbal medicine. *Pharmacy World and Science* **17,** 103–112.

Electronic References

Australian Department of Environment and Heritage (1999). *Environment Protection and Biodiversity Protection Act*. Available at http://www.comlaw.gov.au/Details/C2010C00742/ (last accessed 22 March 2011).

BOC (2001). Decreto 151, de 23 de julio, por el que se crea el catálogo de especies amenazadas de Canarias. *Boletín Oficial de Canarias* **97,** 11101–11111. Available at http://www2.gobiernodecanarias.org/agricultura/pesca/primeraventa/autonomica/boc-a-2010-112-3310.pdf/ (last accessed 22 March 2011).

CITES (2004). *Notification to the Parties No. 2004/033. Concerning Trade in Seahorses. Implementation of Decision 12.54*. Available at http://www.cites.org/eng/notif/2004/033.pdf/ (last accessed 22 March 2011).

CITES (2009). *International Expert Workshop on Non-detriment Findings*. Available at http://www.cites.org/eng/com/AC/24/E24-09.pdf/ (last accessed 22 March 2011).

Davison, G., Ng, P. & Chew, H. H. (2008). *National Red Data Book Singapore*. Available at http://www.nparks.gov.sg/cms/index.php?option=com_content&view=article&id=143&Itemid=128/ (last accessed 22 March 2011).

Deinet, S., McRae, L., de Palma, A., Manley, R., Loh, J. & Collen, B. A. (2010). *The Living Planet Index for Global Estuarine Systems: Report to WWF Netherlands.* London: Institute of Zoology, Zoological Society of London. Available at http://www.zsl.org/science/research-projects/indicators-assessments/estuaries,1397,AR.html/ (last accessed 4 April 2011).

HELCOM (2007). HELCOM Red List of threatened and declining species of lampreys and fish of the Baltic Sea. *Baltic Sea Environmental Proceedings,* **109**. Available at http://www.helcom.fi/stc/files/Publications/Proceedings/bsep109.pdf/ (last accessed 22 March 2011).

Hitchmough, R., Bull, L. & Cromarty, P. (2005). *New Zealand Threat Classification System Lists.* Wellington: Science and Technical Publishing, Department of Conservation. Available at http://conservation.govt.nz/upload/documents/science-and-technical/sap236.pdf/ (last accessed 22 March 2011).

Indian Ministry of Environment and Forests (1972). *The Indian Wildlife Protection Act*. Available at http://envfor.nic.in/legis/wildlife/wildlife1.html/ (last accessed 22 March 2011).

Indian Ministry of Environment and Forests (2001). *Notification of Amendment*. Available at http://www.envfor.nic.in/legis/wildlife/so665%28e%29.htm/ (last accessed 22 March 2011).

IUCN (2010*a*). *IUCN Red List of Threatened Species*. *Version 2010.4*. Available at http://www.iucnredlist.org/ (last accessed 22 March 2011).

IUCN (2010*b*). *National Red Lists*. Available at http://www.nationalredlist.org/site.aspx?pageid=117/ (last accessed 22 March 2011).

Lindley, J. A., Kirby, R. R., Johns, D. G. & Reid, P. C. (2006). Exceptional abundance of the snake pipefish (*Entelurus aequoreus*) in the north-eastern Atlantic Ocean. *International Council for the Exploration of the Sea*. *CM 2006(C:06)*. Available at http://www.ices.dk/products/CMdocs/2006/C/C0606.pdf/ (last accessed 4 April 2011).

List of Threatened Animals of Rio de Janeiro and Sao Paulo States, B. (2008). *Apêndice II – Peixes Marinhos Ameaçados*. Available at http://www.estadao.com.br/ext/especiais/2008/09/II_Peixes_Marinhos.pdf/ and http://www.cetesb.sp.gov.br/licenciamentoo/legislacao/estadual/decretos/2010_Dec_Est_56031.pdf/ (last accessed 22 March 2011).

Lockyear, J. (2000). *Hippocampus capensis*. *IUCN 2010. IUCN Red List of Threatened Species, Version 2010. 4.* Available at www.iucnredlist.org (downloaded 4 April 2011).

McKenzie, L., Yoshida, R. & Coles, R. (2006–2010). *Seagrass-Watch*. Available at www.seagrasswatch.org (last accessed 22 March 2011).

Mejia, L. S. & Acero, A. (2002). *Libro Rojo de peces marinos de Colombia: Orden taxonómico y riesgo de extinción.* Santa Maria: INVEMAR, Instituto de Ciencias Naturales – Universidad Nacional de Colombia, Ministerio del Medio Ambiente. Available at http://www.invemar.org.co/archivo.jsp?id=90&red=true/ (last accessed 22 March 2011).

MMA (2004). Lista Nacional das Espécies de Invertebrados Aquáticos e Peixes Sobreexplotadas ou Ameaçadas de Sobreexplotação. *Instrução Normativa n° 05, de 21 de Maio de 2004. Diário Oficial da União. 28 de Maio de 2004 (2)*, pp. 136–142. Available at http://www.conservation.org.br/arquivos/Anexo2-Lista%20Subrexplotados%20Aquaticos.pdf/ (last accessed 22 March 2011).

OSPAR (2008). *OSPAR List of Threatened and/or Declining Species and Habitats (Reference Number: 2008-6). OSPAR Convention for the Protection of the Marine Environment of the North-East Atlantic*. Available at http://www.ospar.org/content/content.asp?menu=00120000000014_000000_000000/ (last accessed 22 March 2011).

Project Seahorse (2002). *Hippocampus kelloggi*. *IUCN 2010. IUCN Red List of Threatened Species. Version 2010.4*. Available at http://www.iucnredlist.org/ (downloaded on 3 November 2010).

Project Seahorse (2003). *Hippocampus guttulatus*. In *IUCN 2010. IUCN Red List of Threatened Species. Version 2010.4*. Available at http://www.iucnredlist.org/ (downloaded on 3 November 2010).

Rodríguez, J. P. & Rojas-Suárez, F. (2008). *Libro Rojo de la Fauna Venezolana, tercera edición*. Caracas: Provita and Shell Venezuela. Available at http://parquesdecaracas.com/index.php?option=com_content&task=view&id=45&Itemid=2/ (last accessed 30 November 2011).

Townsend, A. J., Lange, P. J. D., Duffy, C. A. J., Molloy, C. M. M. J. & Norton, D. A. (2008). *New Zealand Threat Classification System Manual.* Wellington: Science & Technical Publishing, Department of Conservation. Available at http://www.doc.govt.nz/upload/documents/science-and-technical/sap244.pdf/ (last accessed 22 March 2011).

UNEP-WCMC (2010). *CITES Trade Database*. Available at http://www.unep-wcmc.org/citestrade/trade.cfm/ (last accessed 15 July 2010).

United Nations Department of Economic and Social Affairs/Population Division (2006). *World population Prospects, the 2004 Revision*, Volume III: Analytical Report. New York, NY: United Nations. Available at http://www.un.org/esa/population/publications/WPP2004/ (last accessed 22 March 2011).

Appendix I. All known international (IUCN Red List) and regional and national classifications of conservation status for syngnathid species (alphabetically by genus and species) and their presence in trade, as noted in surveys and CITES records. The IUCN Red List categories are given (in declining order of threat): CR, Critically Endangered; EN, Endangered; VU, vulnerable; NT, Near Threatened; LC, Least Concern; DD, Data Deficient. But note that although the same classification codes are used here for national assessments of conservation status, in 23% of cases the national criteria differ from those used by the IUCN (www.nationalredlist.org). Thus any particular assessment on the IUCN Red List may or may not denote the same level of threat as a similar assessment under a national or regional scheme (www.redlist.org; www.nationalredlist.org). Purported reasons for population declines and notes explaining status, where necessary, are also offered as footnotes. As only *Hippocampus* spp. are listed by CITES, there are no CITES data on the trade in pipefishes, pipehorses or seadragons (indicated with N/A). This table only includes species for which at least one piece of relevant information was available, so many syngnathid species are not represented here

		Threatened species listing			Trade*			
		Regional			Surveys		CITES	
Scientific name	IUCN status	Location	Status	Listing source and criteria used	Dry	Live	Dry	Live
Bhanotia nuda	LC	—	—	—			N/A	N/A
Bryx veleronis	DD	—	—	—			N/A	N/A
Corythoichthys sp.	—	—	—	—		√	N/A	N/A
Cosmocampus arctus	LC	New Zealand	DD	Hitchmough *et al.* (2005); Townsend *et al.* (2008)			N/A	N/A
Cosmocampus elucens	LC	—	—	—			N/A	N/A
Doryrhamphus dactyliophorus	DD	—	—	—			N/A	N/A
D. janssi	LC	—	—	—			N/A	N/A
Doryrhamphus sp.	—	—	—	—		√	N/A	N/A
Enneacampus ansorgii	LC	—	—	—			N/A	N/A
E. kaupi	LC	—	—	—			N/A	N/A
Entelurus aequoreus	—	Baltic Sea	VU†	HELCOM (2007); IUCN (2010*b*)			N/A	N/A
Halicampus koilomatdon	—	—	—	—	√		N/A	N/A
Heraldia nocturna	—	—	—	—		√	N/A	N/A
Hippichthys cyanospilus	—	—	—	—	√		N/A	N/A
H. heptagonus	LC	Japan‡	EN§	IUCN (2010*b*)			N/A	N/A

Appendix I. Continued

Scientific name	IUCN status	Threatened species listing: Regional: Location	Status	Listing source and criteria used	Trade*: Surveys: Dry	Live	CITES: Dry	Live
H. spicifer	LC	—	—	—			N/A	N/A
Hippocampus abdominalis	DD	New Zealand‡	Not threatened	IUCN (2010*b*)	√	√	√	√
H. algiricus	DD	—	—	—			√	√
H. angustus	DD	—	—	—	√	√		√
H. barbouri	VU: A4cd‖	—	—	—	√	√	√	√
H. bargibanti	DD	—	—	—			√	
H. borboniensis	DD	—	—	—	√			√
H. breviceps	DD	—	—	—		√	√	√
H. camelopardalis	DD	—	—	—	√	√		√
H. capensis	EN: B1 + 2c + 3d¶	—	—	—				
H. comes	VU: A2cd**	—	—	—	√	√	√	√
H. coronatus	DD	—	—	—		√		√
H. denise	DD	—	—	—				√
H. erectus	VU: A4cd‖	Brazil	Overexploited††	List of Threatened Animals of Rio de Janeiro and Sao Paulo States (2008)	√	√	√	√
		Colombia	VU‡‡	Mejia & Acero (2002); IUCN (2010*b*)				

Appendix I. Continued

Scientific name	IUCN status	Threatened species listing			Trade*			
		Regional			Surveys		CITES	
		Location	Status	Listing source and criteria used	Dry	Live	Dry	Live
		Venezuela	NT§	Rodríguez & Rojas-Suárez (2008); IUCN (2010*b*)				
H. fisheri	DD	—	—	—				
H. fuscus	DD	—	—	—	√	√	√	√
H. guttulatus	DD	Croatia	EN§	Jardas *et al*. (2007); IUCN (2010*b*)	√	√	√	√
		North-east Atlantic Ocean	Threatened or declining§§	OSPAR (2008)				
		Slovenia, France, Portugal	Threatened	Project Seahorse (2003)				
H. hendriki	DD	—	—	—				
H. hippocampus	DD	Croatia	DD	Jardas *et al*. (2007); IUCN (2010*b*)	√		√	√
		North-east Atlantic Ocean	Threatened or declining§§	OSPAR (2008)				
		Canary Islands	Species of interest‖‖	BOC (2001)				
H. histrix	DD	Vietnam	VU§	Institute for Science and Technology of Vietnam (2007); IUCN (2010*b*)	√	√	√	√
H. ingens	VU: A4cd‖	Colombia	VU‡‡	Mejia & Acero (2002); IUCN (2010*b*)	√	√	√	√

Appendix I. Continued

		Threatened species listing			Trade*			
		Regional			Surveys		CITES	
Scientific name	IUCN status	Location	Status	Listing source and criteria used	Dry	Live	Dry	Live
H. jayakari	DD	—	—	—				
H. kelloggi	DD	China	—	Project Seahorse (2002)	√		√	√
		Vietnam	VU§	Institute for Science and Technology of Vietnam (2007); IUCN (2010*b*)				
H. kuda	VU: A4cd‖	Vietnam	VU§	Institute for Science and Technology of Vietnam (2007); IUCN (2010*b*)	√	√	√	√
		Singapore	VU§	Davison *et al.* (2008)				
H. lichtensteinii	DD	—	—	—				
H. minotaur	DD	—	—	—				
H. mohnikei (*H. japonicus*)	DD	Vietnam	Rare	Institute for Science and Technology of Vietnam (2007); IUCN (2010*b*)	√	√		?
H. montebelloensis	—	—	—	—				√
H. pontohi	DD	—	—	—				
H. reidi	DD	Colombia	VU¶¶	Mejia & Acero (2002)	√	√	√	√
		Venezuela	DD	Rodríguez & Rojas-Suárez (2008); IUCN (2010*b*)				

Appendix I. Continued

		Threatened species listing			Trade*			
		Regional			Surveys		CITES	
Scientific name	IUCN status	Location	Status	Listing source and criteria used	Dry	Live	Dry	Live
H. satomiae	DD	—	—	—				
H. severnsi	DD	—	—	—				
H. sindonis	DD	—	—	—				
H. spinosissimus	VU: A4cd5	—	—	—	√	√	√	√
H. subelongatus	DD	—	—	—		√		√
H. trimaculatus	VU: A4cd5	Vietnam	VU§	Institute for Science and Technology of Vietnam (2007); IUCN (2010*b*)	√		√	√
H. whitei	DD	—	—	—	√	√		√
H. zebra	DD	—	—	—	√			√
H. zosterae	DD	—	—	—		√	√	√
Idiotropiscis australe	DD	—	—	—			N/A	N/A
Leptonotus elevatus	—	New Zealand‡	Not threatened	IUCN (2010*b*)			N/A	N/A
L. norae	—	New Zealand‡	Not threatened	IUCN (2010*b*)			N/A	N/A
Lissocampus filum	—	New Zealand‡	Not threatened	IUCN (2010*b*)			N/A	N/A
Microphis argulus (coelonotus)	—	Japan‡	CR	IUCN (2010*b*)			N/A	N/A
M. (Doryichthys) boaja	—	—	—	—	√		N/A	N/A
M. caudocarinatus	DD	—	—	—			N/A	N/A
M. cuncalus	LC	—	—	—			N/A	N/A

Appendix I. Continued

Scientific name	IUCN status	Threatened species listing: Regional: Location	Status	Listing source and criteria used	Trade*: Surveys: Dry	Live	CITES: Dry	Live
M. deocata	NT***	—	—	—			N/A	N/A
M. dunckeri	LC	—	—	—			N/A	N/A
M. insularis	VU: B1ab(iii)†††						N/A	N/A
M. jagorii (oostethus)	—	Japan‡	CR	IUCN (2010*b*)			N/A	N/A
M. leiaspis	LC	—	—	—			N/A	N/A
M. retzii (lophocampus)		Japan‡	CR	IUCN (2010*b*)			N/A	N/A
M. spinachioides	DD	—	—	—			N/A	N/A
Nerophis lumbriciformis	—	Baltic Sea	VU†	HELCOM (2007); IUCN (2010*b*)			N/A	N/A
N. maculatus	—	Croatia	DD	Jardas *et al*. (2007); IUCN (2010*b*)			N/A	N/A
N. ophidian	—	Baltic Sea	VU†	HELCOM (2007); IUCN (2010*b*)		√	N/A	N/A
		Croatia	DD	Jardas *et al*. (2007); IUCN (2010*b*)			N/A	N/A
Phycodurus eques	NT‡‡	—	—	—		√	N/A	N/A
Phyllopteryx taeniolatus	NT‡‡	—	—	—		√	N/A	N/A
Siokunichthys nigrolineatus	LC	—	—	—			N/A	N/A

Appendix I. Continued

		Threatened species listing			Trade*			
		Regional			Surveys		CITES	
Scientific name	IUCN status	Location	Status	Listing source and criteria used	Dry	Live	Dry	Live
Solegnathus dunckeri	DD	—	—	—	√		N/A	N/A
S. hardwickii	DD	Vietnam	Threatened§	Institute for Science and Technology of Vietnam (2007); IUCN (2010*b*)	√	√	N/A	N/A
S. lettiensis	DD	—	—	—	√		N/A	N/A
S. robustus	DD	—	—	—	√		N/A	N/A
S. spinosissimus	DD	New Zealand‡	Not threatened	IUCN (2010*b*)	√		N/A	N/A
Stigmatopora argus	—	New Zealand	Sparse	Hitchmough *et al*. (2005); Townsend *et al*. (2008)		√	N/A	N/A
S. macropterygia	—	New Zealand	DD	Hitchmough *et al*. (2005); Townsend *et al*. (2008)			N/A	N/A
S. nigra	—	New Zealand	DD	Hitchmough *et al*. (2005); Townsend *et al*. (2008)			N/A	N/A
Stipecampus cristatus	—	—	—	—		√	N/A	N/A
Syngnathoides biaculeatus	DD	—	—	—	√	√	N/A	N/A
Syngnathus abaster	LC	Croatia	DD	Jardas *et al*. (2007); IUCN (2010*b*)			N/A	N/A

Appendix I. Continued

		Threatened species listing			Trade*			
		Regional			Surveys		CITES	
Scientific name	IUCN status	Location	Status	Listing source and criteria used	Dry	Live	Dry	Live
S. acus	—	Baltic Sea	EN†	HELCOM (2007); IUCN (2010*b*)			N/A	N/A
		Croatia		Jardas *et al*. (2007); IUCN (2010*b*)			N/A	N/A
		Vietnam	VU§	Institute for Science and Technology of Vietnam (2007); IUCN (2010*b*)			N/A	N/A
S. auliscus	LC	—	—	—			N/A	N/A
S. carinatus	DD	—	—	—			N/A	N/A
S. floridae	LC	—	—	—			N/A	N/A
S. macrobrachium	DD	—	—	—			N/A	N/A
S. pelagicus	—	—	—	—		√	N/A	N/A
S. phlegon	—	Croatia	DD	Jardas *et al*. (2007); IUCN (2010*b*)			N/A	N/A
S. rostellatus	—	Baltic Sea	LC	HELCOM (2007); IUCN (2010*b*)			N/A	N/A
S. scovelli	—	—	—	—	√	√	N/A	N/A
S. tenuirostris		Croatia	DD	Jardas *et al*. (2007); IUCN (2010*b*)			N/A	N/A
S. typhle	—	Baltic Sea	VU†	HELCOM (2007); IUCN (2010*b*)		√	N/A	N/A
		Croatia	DD	Jardas *et al*. (2007); IUCN (2010*b*)			N/A	N/A
S. watermeyeri	CR: B1ab(i,ii,iii);C2a(i)§§§	—	—	—			N/A	N/A

Appendix I. Continued

Scientific name	IUCN status	Threatened species listing: Regional: Location	Status	Listing source and criteria used	Trade*: Surveys: Dry	Surveys: Live	CITES: Dry	CITES: Live
Trachyrhamphus serratus	—	Vietnam	Vulnerable§	Institute for Science and Technology of Vietnam (2007); IUCN (2010*b*)	√		N/A	N/A
Vanacampus sp.	—	—	—	—		√	N/A	N/A

*Sourced from second surveys, collated by B. Giles, unpubl. data. CITES data found in UNEP-WCMC (2010); Evanson *et al*. (2011).
†Reason: Habitat loss.
‡Note: This species is listed on http://www.nationalredlist.org/site.aspx/, but is not found on the country's own list of threatened species, according to government or academic publications (November 2010).
§Reason: Unknown.
||Reason: A4cd = Population reduction of ≥30% in the previous or next 10 years resulting from reductions in occupied area, extent, quality of habitat and from actual or potential exploitation, both of which have not ceased or are irreversible.
¶Reason: B1 + 2c + 3d = Extent < 100 km^2, occupied area estimated to be <10 km^2, severely fragmented, continuing decline in occupied area, extent and quality of habitat, and extreme fluctuations in number of mature individuals.
**Reason: A2cd = Population reduction of ≥30% in the previous 10 years resulting from reductions in area, extent, quality of habitat and from actual or potential exploitation, both of which have not ceased or are irreversible.
††Reason: Overexploited (AS in Portugal) = Threatened due to huge biomass reduction; species requires monitoring and management measures.
‡‡Reason: A3d = Population reduction of ≥30% projected in the next 10 years resulting from actual or potential levels of exploitation.
§§Reason: Regional importance (threatened inshore habitat), population declines, sensitivity due to life-history characteristics and threats due to targeted and by-catch fisheries.
||||Reason: This species has a unique ecological role in its ecosystem and there are threats present that may cause population declines.
¶¶Reason: A2ad = Population reduction of ≥30% in the previous 10 years known from direct observation and resulting from actual or potential exploitation, which has not ceased or is irreversible.
***Reason: NT for this example = Known population decline across part of its range (VU by A4) and threats arising from habitat degradation (A4c).
†††Reason: B1ab(iii) = Extent <20 000 km^2, number of known locations <10, and continuing decline in habitat quality.
‡‡‡Reason: NT for this example: Extent is <5000 km^2 (EN by B1) and continuing decline in area and quality of habitat [EN by B1b(iii)].
§§§Reason: B1ab(i,ii,iii);C2a(i) = Extent < 100 km^2, severely fragmented populations, small population size, number of mature individuals in each subpopulation <50 and continuing decline in extent, area occupied, quality of habitat and population size.