Marine extinctions revisited

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Abstract

In recent years, more than 130 extinctions have been estimated to have occurred in the marine realm. Here we review this body of evidence and show that this figure may actually be overestimated by as much as 50%. We argue that previous estimates have not fully taken into account critical uncertainties such as naturally variable geographical distributions, and have misinterpreted documentary evidence. However, current evidence indicates that some sharks, rays and reef-associated species, although not necessarily geographically restricted, are particularly vulnerable to anthropogenic impacts and now occur in very low numbers. Overestimating extinctions is of concern because it could reduce confidence in the credibility of the 'extinct' category in threatened species lists and, ultimately, be used to question the integrity of conservation and management policies. We suggest that when integrating future checklists of marine extinct species, there needs to be a more rigorous use of the terminology of extinction, and participation by specialists in each of the particular taxonomic groups involved.

Keywords biodiversity loss, extirpation, global change, habitat loss, overexploitation

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Received 4 July 2006
Accepted 16 March 2007

Introduction

A sizeable volume of scientific literature has appeared recently on the extinction of marine species (Malakoff 1997; Casey and Myers 1998; Carlton *et al.* 1999; Roberts and Hawkins 1999; Jackson *et al.* 2001). Prior to 1999, there was unequivocal documentary evidence for the global disappearance of 12 marine species: three mammals, five birds and four molluscs (Carlton *et al.* 1999; Roberts and Hawkins 1999), but a few years afterwards, Dulvy *et al.* (2003) presented a broad scale overview which listed 133 cases of global, regional and local extinctions in other marine groups, including the global extinction of the once commercially lucrative white abalone (*Haliotis sorenseni*, Haliotidae). They relate most of these extinctions to direct human impact – predominantly over-harvesting.

Large catalogues of threatened and potentially extinct taxa represent a critical resource for the scientific testing of ideas about extinction processes, but as stressed by MacPhee and Flemming (1999), it is imperative that the databases underpinning such investigations are as reliable as possible. We review the cases reported by Dulvy *et al.* (2003) and other recent literature on marine extinctions (Close 2002; Kranenbarg *et al.* 2002; Donaldson and Dulvy 2004; Dulvy and Polunin 2004; Munday 2004; Ferreira *et al.* 2006), which include mammals, birds, fish, invertebrates and algae, and suggest alternative verdicts regarding the current vulnerability status of many of these species. We conclude by highlighting possible implications of misdiagnosis for the conservation and management of living marine resources.

Mammals

Among the species listed by Dulvy *et al.* (2003), there are seven instances of cetacean extinction, three phocids, two mustelids and two sirenians. It seems to be beyond reasonable doubt that commercial overexploitation and habitat loss caused the demise of Steller's sea cow (*Hydrodamalis gigas*, Dugongidae), the Caribbean monk seal (*Monachus tropicalis*, Phocidae) and the sea mink (*Mustela macrodon*, Mustelidae). The sea otter (*Enhydra lutris*, Mustelidae), a species with an ocean basin-wide distribution is documented as regionally extinct in the North-east Pacific. Nevertheless, other populations of the same species reveal contrasting trends: while the numbers are declining in the Aleutian Islands (Springer *et al.* 2003), there is no overall tendency towards change in the southwest of Prince William Sound (Bodkin *et al.* 2002) and the otter is recovering in other sites of the Sound and along the coast of California [Friends of the Sea Otter (FSO) 2006]. It has also been sighted three times in Baja California, México, during the late 1970s, again in 1994 (Rodriguez-Jaramillo and Gendron 1996) and most recently there have been several sightings in the Central Pacific Coast of Baja California (D. Aurioles-Gamboa, unpublished data); all this more than 70 years after its purported regional disappearance. At a narrower geographical scale, the dugong (*Dugong dugon*, Dugongidae) was reported as locally extinct in China in 2000. However, in that same year, land/boat-based studies found at least five dugongs living in waters off the coast of Hainan Island (within the Gulf of Tonkin) (Marsh *et al.* 2002), and anecdotal reports suggest that the species occurs in greater numbers in adjacent areas. At best, the status of dugongs in Chinese waters is poorly quantified to date and the extent of their current distribution remains uncertain (Marsh *et al.* 2002).

The grey whale (*Eschrichtius robustus*, Eschrichtiidae) has not only been extirpated from the Wadden Sea by overexploitation (as reported by Dulvy *et al.* 2003), but in fact from the entire Atlantic Ocean within the last 300–400 years; clearly a case of regional disappearance. Nonetheless, the species may well be recovering to pre-whaling numbers in the Pacific (Moore *et al.* 2001; Weller *et al.* 2002). The bottlenose dolphin (*Tursiops truncatus*, Delphinidae) has also been regarded as locally extinct since 1981 in waters off the Netherlands and Wadden Sea, due most probably to habitat alteration. However, the species is naturally rare at these latitudes, with the maximum number of dolphins ever sighted standing at around 40 individuals (Verwey and

Wolff 1981). Further, it is feasible that they are simply irregular visitors to these areas or temporarily absent; for instance, through the 1960s the once regularly sighted bottlenose dolphins disappeared from San Diego Bay, possibly due to increased pollution, but as the water quality improved during the 1970s, the dolphins returned (Klinowska 1991).

Birds

Dulvy *et al.* (2003) reported 12 extinctions of marine birds, five of them global. All the reported global extinctions occurred more than 90 years ago; the last sightings of the five species being 1844 (*Pinguinus impennis*, Alcidae), 1850 (*Phalacrocorax perspicillatus*, Phalacrocoracidae), 1875 (*Camptorhynchus labradorius*, Anatidae), 1902 (*Mergus australis*, Anatidae) and 1913 (*Haematopus meadewaldoi*, Haematopodidae). Global extinction reports themselves must be considered with due care, as demonstrated by some instances where species reported as extinct long ago have recently been rediscovered. For instance, the Reunion petrel (*Pterodroma aterrima*, Procellariidae) was assumed extinct since the late 19th century but later reported from two dead specimens in the 1970s and seen again recently, whilst the Cahow petrel (*Pterodroma cahow*, Procellariidae), believed to have vanished in 1621, was found again during 1951 (Fernández y Fernández-Arroyo 2004). Of the seven local extinctions considered by Dulvy *et al.* (2003), six (85%) are limited to the Wadden Sea, a rather short 450 km section of Dutch coast, especially considering the typical flying range of marine birds.

Only one of the species included by Dulvy *et al.* (2003) in the local extinction category of is regarded as vulnerable by the International Union for the Conservation of Nature (IUCN) and this is the Dalmatian pelican (*Pelecanus crispus*, Pelecanidae). This species has not been sighted in the Wadden Sea since AD 200, and its disappearance is attributable to habitat loss and overexploitation. This is confirmed by the skull of this species found in an archeological excavation at Assen-delft, about 65 km south of the Dutch Wadden Sea and dating from the Roman period (0–200 AD) (Wolff 2000a). Although the Dalmatian pelican breeding populations extend to Europe, the Mediterranean and Asia, the North Sea is not indicated as a wintering area. Moreover, this bird is regarded only as an occasional visitor to Germany. Conservation measures have resulted in a population increase in Europe and suggestions have been made to investigate the possibilities of re-introduction (Wolff 2000a). However, rapid population declines are inferred to be continuing in the remainder of its range (BirdLife International 2004).

In contrast, there has actually been a significant increase in the numbers of the lesser black-backed gull (*Larus fuscus*, Laridae) in North America [American Ornithologists' Union (AOU) 1998; Alsop 2001]. The species is also recolonizing the Dutch coast (Wolff 2000a), along with the common eider duck (*Somateria mollissima*, Anatidae), another species reported as locally extinct by Dulvy *et al.* (2003). The four species of gulls and terns (Laridae) mentioned by Dulvy *et al.* (2003) occupy extremely extensive breeding and foraging areas and it is doubtful whether their disappearance from small parts of their normal range should be considered as local extinction.

Fish

Of the three species of marine fish reported as globally extinct, we found evidence to support two of them: the green wrasse (*Anampses viridis*, Labridae) and the New Zealand grayling (*Prototroctes oxyrhynchus*, Retropinnidae; not *Protoctes* as reported by Dulvy *et al.* 2003). Distribution of the green wrasse was restricted to the coast of Mauritius, and it has not been seen since the mid-19th century. Recent surveys suggest that there is little doubt that it is indeed extinct (Letourneur *et al.* 2004), possibly a victim of sedimentation and nutrient pollution (Hawkins *et al.* 2000). The grayling, whose reported distribution is restricted to New Zealand, is an amphidromous fish – an attribute that renders the species more vulnerable to extinction by human impact when compared with a solely marine fish (Del Monte-Luna and Lluch-Belda 2003). It appears that this species was driven extinct as a result of the extensive loss of its critical freshwater breeding sites. The third species, the Galapagos damsel (*Azurina eupalama*, Pomacentridae), apparently endemic to the Galápagos, is presumed to have disappeared completely following the major El Niño-Southern Oscillation (ENSO) event of 1982–1983, but its status is still a matter of debate (Hawkins *et al.* 2000; Victor *et al.* 2001). In addition, there are reservations regarding its restricted distribution around the Galápagos; its affinity for temperate waters leaves open the possibility that it could be found off Ecuador and Perú in areas and at depths that have not been sampled adequately (Victor *et al.* 2001; Robertson and Allen 2002). Only one of these three marine fish species, the New Zealand grayling, is categorized by the IUCN as 'Extinct' [World Conservation Monitoring Centre (WCMC) 1996].

Two species are documented as regionally extinct by Dulvy *et al.* (2003): the smalltooth sawfish (*Pristis pectinata*, Pristidae) from the Western Atlantic and the Chinese bahaba (*Bahaba taipingensis*, Sciaenidae) from South China. The amphidromous sawfish has circumglobal distribution, from North Carolina to Brazil, the Indo-Western Pacific and possibly the Mediterranean Sea and the Eastern Pacific. Recent evidence (Adams *et al.* 2000) certainly supports the view that population numbers of *P. pectinata* are likely to be dangerously below viable levels in most of its former range in US waters, and that conservation measures to assist recovery of this species to 'safe' population levels are urgently needed. However, using the sighting record of *P. pectinata* together with the model of Roberts and Solow (2003) to determine a time interval during which this species may have become extinct, we estimate that the upper probability bound of the period during which *P. pectinata* could become extinct extends to the year 2013 in the US Gulf of Mexico and to the year 2061 from north of Florida to North Carolina. These results highlight the uncertainty involved in any declaration of this species as 'regionally extinct' outside the Florida Keys and the Everglades National Park in US waters.

The Chinese bahaba, described scientifically in the early 1930s from a specimen collected at the markets in China, was reported as commercially extinct in 1997. The information about its near extinction is based on published accounts and interviews with local fishermen (Sadovy and Cheung 2001).

The remaining 59 reports (pertaining to 47 species) refer to local extinctions. Roughly 36% of these species have disappeared from the boundaries of their latitudinal distributions, which could indicate a contraction of their natural geographical range, as has been documented for other fish species (Lluch-Belda *et al.* 1989, 1992). These movements may also be offshore or into pockets of deeper water (Perry *et al.* 2005) as has been observed in some endangered rays (Kulka *et al.* 2002). Regarding such range contractions as 'extinction', even if qualified as 'local', inadequately describes the process underpinning this change and may misrepresent the consequence of these events, given that species distributions are now considered more dynamic than was traditionally the case (Perry *et al.* 2005).

Nine instances of extinction are based on Jukic-Peladic *et al.* (2001), who compared the results of two similar fishing surveys undertaken in the Adriatic Sea and spaced 50 years apart (1948 and 1998). Yet all the species reported as 'extinct' are simply the ones caught during the first survey and not during the second. Not one is a target species for the local fishery, and of the nine species not recorded in 1998, the three most frequently caught ones in 1948 never exceeded 4% occurrence, suggesting that they are naturally rare in the area.

Dulvy *et al.* (2003) declared a dozen shark and ray species to be locally extinct in the Mediterranean Sea. However, a thorough literature survey revealed at least four recent articles in peer-reviewed scientific journals reporting the presence of some of these species in different parts of the Mediterranean Sea, after the supposed last sighting years (Storelli *et al.* 2002; Jardas *et al.* 2004; Ferretti *et al.* 2005; Massutí and Reñones 2005). Another species, the largetooth sawfish (*Pristis perotetti*, Pristidae) – regarded as extinct in the Gulf of California – is actually a synonym of *Pristis pristis* (McEachran and Fechhlem 1998); the information is based on a personal observation by Findley to Musick *et al.* (2000). Yet *P. pristis* and *P. pectinata* have never been formally reported in the Gulf of California (Jordan and Starks 1895; Minckley *et al.* 1986).

Dulvy *et al.* (2003) documented an additional dozen local extinctions of cartilaginous fish in the Gulf of Lions (within the Mediterranean Sea), based upon the work of Aldebert (1997). Whilst this author identified a definite historical decline in the abundance of elasmobranchs and the disappearance of some species since the 1970s (possibly due to an increasing fishing effort), he does not suggest they became extinct, instead cautioning repeatedly that the survey database should only be used to describe possible qualitative changes in groundfish diversity, and that further attention must be paid to the role of environmental factors on these long-term trends. Translating this information directly into instances of extinction is unconvincing.

Invertebrates

Thirty-one extinctions of invertebrate animals were reported by Dulvy *et al.* (2003), eight of which were considered global losses. Two of these species seem to have disappeared for reasons unrelated to human impact (Roberts and Hawkins 1999). The eelgrass limpet (*Lottia alveus*, Lottiidae) a mollusc which formerly occupied seagrass beds along the north-eastern coastline of North America, became extinct in the 1930s when an epidemic disease wiped out its primary habitat. The other species, Boschmai's fire coral (*Millepora boschmai*, Milleporidae), along with several other coral species, suffered a severe reduction in densities on reefs throughout the Eastern Pacific following the very strong El Niño of 1982–1983. This recently described hydrozoan was initially thought to be extinct, but five colonies were later discovered alive (Glynn and Feingold 1992). Its continued survival remains precarious, particularly as the 1997–1998 El Niño was of similar or greater severity than the earlier damaging event (Glynn *et al.* 2001).

Another invertebrate species listed as globally extinct is the rocky shore limpet (*Collisella edmitchelli*, Lottiidae). Data from the USA and Canada regarding its distribution are either known to be incomplete or have not been reviewed. Except for a single live specimen collected from San Pedro, California in 1861 (or 1863), this species is known only from Pleistocene deposits in California (Turgeon *et al.* 1998). The scleractinian coral (*Siderastrea glynni*, Siderastreidae) was recently described by Budd and Guzmán (1994) and assumed to be critically endangered in 1998 (Fenner 2001), but it still persisted in Panamá in 2000 (Maté 2003). Genetic analysis of the species and others of the same genus revealed its likely origin as descendant from a population which arrived from the Caribbean after a breach of the Central American Isthmus (approximately 2 Ma), or perhaps recently via introduction by ship (Forsman *et al.* 2005). The Ivell's sea anemone (*Edwardsia ivelli*, Edwardsiidae), endemic from the Widewater Lagoon in West Sussex, UK, has not been found since 1983. However, this is a small, well-camouflaged anemone of shallow soft mud bottoms; a habitat not well explored for small invertebrates, and thus 'it may well be living, unnoticed, in other localities' (Barnes 1994). Two other globally extinct species, the Periwinkle (*Littoraria flammea*, Littorinidae) from China and the horn snail (*Cerithidea fuscata*, Potamididae) endemic from San Diego, were last seen in 1840 and 1935, respectively, and must be considered strong candidates for actual extinction.

The remaining case of 'global loss' is that of the white abalone, the likely cause of which was over-fishing. However, Rogers-Bennett *et al.* (2004) only consider it to be an endangered species, and it is not included on the IUCN Red List [International Union for the Conservation of Nature (IUCN) 2006]. Whilst white abalone populations have suffered serious stock depletions along the West coast of North America (Hilborn *et al.* 2005), we argue that there is no basis to consider it extinct. Butler *et al.* (2006), using multibeam sonar mapping techniques, estimated that white abalone populations at Tanner Bank, Cortes Bank and San Clemente Island, California, range from several hundreds to thousands of individuals. More estimates are needed in order to determine the status of Southernmost (Mexican) populations.

The long-spined sea urchin (*Diadema antillarum*, Diadematidae) is the only invertebrate reported as regionally extinct for the entire Caribbean by Dulvy *et al.* (2003). During the mid-1980s, this species suffered the most extensive and severe mortality event ever reported for a marine organism (>93% loss of biomass), possibly caused by an unidentified pathogen. At that time, whilst some populations of *Diadema* were heavily diminished, they remained present in some strongholds within the Caribbean (Brandt *et al.* 2005). Recent evidence indicates that dense aggregations of *Diadema* spanning many square kilometres now occur at six locations, scattered along a 4100-km arc of Caribbean, and that this recovery of *Diadema* is occurring at both local and regional scales (Edmunds and Carpenter 2001; Knowlton 2001; Lessios 2005; Carpenter and Edmunds 2006). Certain populations documented as local extinctions may be only temporarily absent from a small part of their wide natural ranges, such as the short-spined sea urchin (*Tripneustes gratilla*, Toxopneustidae) in Bolinao during 1995 (Talaue-McManus 2000). However, at the same time the mariculture of this species was successfully piloted by local communities [GESAMP (Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection) 1996] and there are subsequent reports of significant numbers of natural short-spined sea urchin recruits at various reef areas in Bolinao (Juinio-Meñez 2000).

Other instances of local extinction are difficult to resolve. Earlier, over-fishing of the purple sea urchin (*Paracentrotus lividus*, Echinidae) was presumed to have led to local extinction from Lough Hyne, which was Europe's first marine reserve designated as such in 1981. Censuses of the years 2000 and 2001 revealed no individuals in the Southern Basin (Barnes *et al.* 2002), despite the observed spawning of this species in July 1999 at Lough Hyne (Greenwood *et al.* 2000). Some authors have linked this decline to anomalously cold sea surface temperatures and potentially to the ENSO event and toxic algal blooms (Barnes *et al.* 2001). The California hornsnail (*Cerithidea californica*, Potamididae), a species distributed from Peru to California, is reported as locally extinct from Southern California since 1975, yet in a survey conducted from 1996 to 1998, which covered from Boundary Bay to San Diego Bay, *C. californica* was absent from only 2 of 11 sampling sites from which it was expected: Bodega Harbour and Elkhorn Slough (Byers 1999).

Local extinctions have also been reported for species whose body size does not exceed 8 mm, such as micromolluscs (*Corambe obscura*, Corambidae; *Onoba semicostata* and *Rissoa membranacea*, Rissoidae) and the mud shrimp (*Upogebia bermudensis*, Upogebiidae). Moreover, the observed and potential habitat of these species have not been properly sampled (Bouchet 1997). Likewise, it is anomalous that Dulvy *et al.* (2003) cited invasive or cultured species like the Harris mud crab (*Rhithropanopeus harrisii*, Xanthidae) and the Eastern oyster (*Crassostrea virginica*, Ostreidae) as locally extinct. They also mistakenly considered as an extinction the local disappearance of species with broad geographical ranges, such as the mysid (*Acanthomysis longicornis*, Mysidae) – distributed throughout the Mediterranean and North Sea and the Banded northern lacuna (*Lacuna vincta*, Littorinidae) whose range extends over 30° of latitude in Northern Europe and America.

Algae

We consider 2 of the 12 extinctions of marine algae reported by Dulvy *et al.* (2003) to be valid. Bennett's seaweed (*Vanvoorstia bennettiana*, Delesseriaceae) seems to be the only marine alga registered in any of the World's official threatened species lists (Millar 2003a,b). According to the New South Wales state threatened species criteria, this organism was considered taxonomically valid although its historical distribution was always very restricted (endemic to Sydney Harbour). Once considered relatively common, the alga suffered a catastrophic population decline linked to human impacts and has not been collected for at least 50 years (Millar 2003a,b). That species had a life history strategy which has been linked to high extinction vulnerability, with an isomorphic alternation of generations (no microscopic alternate stage). The other species, the Turkish towel algae (*Gigartina australis*, Gigartinaceae), is nominated as a species 'presumed extinct' and was also found only in Sydney Harbour (Millar 2003a).

All the remaining local extirpations reported for algal species refer to the Wadden Sea, following Wolff (2000b)), with the last sighting years for these species being unknown. Some of these algae are difficult to detect because they have a microscopic phase (*Puncataria latifolia* (Chordariaceae), *Colpomenia peregrine* (Scytosiphonaceae) or they are small filamentous plants (10–15 cm length) such as *Ceramium diaphanum* (Ceramiaceae), *Spermothamnion repens* (Ceramiaceae) and *Sphacelaria cirrosa* (Sphacelariaceae) (Graham and Wilcox 2000).

It should be noticed that all but one, *Antithamnion villosum* (Ceramiaceae), have been reported from many different regions around the world; the species has only been registered in Europe (Guiry *et al.* 2005), but there is no further information to discuss its status. The original sources used to define these extirpations are mainly floristic catalogues which, although providing valuable historical information, may not have involved sufficiently exhaustive surveys to provide a robust basis for inferring local extirpations (see the case of Thibaut *et al.* (2005) in the next section on newly reported extinctions).

Taxonomic reclassifications can also lead to problems in interpreting threatened species lists (Issac *et al.* 2004): *Punctaria hiemalis* (cited as extinct by Dulvy *et al.* 2003), for instance, is a synonym of *P. latifolia* (Punctariaceae). In fact *Colpomenia peregrina* (Scytosiphonaceae) is the only species in the algal listing in Dulvy *et al.* (2003) without a synonym; the rest have been labelled with anywhere from 2 to 37 synonyms, indicating a pressing need for taxonomic revision of this group to better determine the relationships among populations from different regions (Guiry *et al.* 2005).

Marine extinctions reported since 2003

Since Dulvy *et al.* (2003) published their review there has been a number of new reports of marine extinctions which we briefly survey in this section. The rainbow parrotfish (*Scarus guacamaia*, Scaridae) is reported as locally extinct in Brazil by Ferreira *et al.* (2006). In the Caribbean, where it is still present (Mejía and Acero 2002; Kramer 2003; Dorenbosch *et al.* 2006), the species occurs from Bermuda and south of Florida to southern Brazil, with rare reports from Argentina (Mejía and Acero 2002). Verified records of the occurrence of this species in Brazil are restricted to a few museum specimens, a photographed juvenile, and one jaw obtained from spear fishermen. None are more recent than the mid-1980s. Based on museum specimens and anecdotal accounts, this species was once distributed along the entire tropical coast of Brazil. The main threats for the species are over fishing and mangrove destruction, and it is included in the IUCN Red List as 'Vulnerable'.

The work of De Moura and Francini-Filho (2005) contains the only recent field survey data for assessing Abrolhos' reef fishes from north-east Brazil. They show that despite logging more than 90 days (220 h) of surveys over a 5-year period (1997–2001), divided into six sampling trips (Jan. 1997, Jan. 1998, March/Apr. 1999, Nov. 1999, Feb. 2000 and Feb. 2001) they failed to report any occurrence of the rainbow parrotfish. On average, they logged about 1.5 h day^{-1} of underwater assessment during the course of 1 month. Although such a sampling effort is intensive, the rainbow parrotfish is likely to have a low probability of detection (De Moura *et al.* 2001). Non-detection errors for rare species can be substantial (Gu and Swihart 2004).

The bumphead parrotfish (*Bolbometopon muricatum*, Scaridae) is also reported as locally extinct from the Guam and Marshall Islands and from parts of Fiji and East Africa, likely due to overexploitation (Donaldson and Dulvy 2004; citing Bellwood *et al.* 2003 for supporting evidence). However, Bellwood *et al.* (2003) do not mention or use the term 'extinct' anywhere in their paper. Donaldson and Dulvy (2004), however, found that in seven of their eight Indonesian sites, and in all 16 Micronesian and French Polynesian sites (out of a total of 44 sites spread across the Indo-Western Pacific) *B. muricatum* and *Chlorulus* (Scaridae) densities were 'extremely low' (qualitative remark). Given that virtually all bioerosion in those high-production, high-erosion crest habitats is caused by parrotfish, the loss of *B. muricatum* may have profound effects on coral community structure, biogeography and demography.

Using census methods described in Bellwood and Wainwright (2001); Bellwood *et al.* (2003) conducted fish censuses over an important geographical scale in the northern sector of the Great Barrier Reef. The surveys were conducted between November 1998 and February 1999; a short period which may not be sufficient to properly assess extinction. The status of the bumphead parrotfish is further tackled by Dulvy and Polunin (2004). Based on informal questioning of Fijian villagers and fishers, they inferred that the rarity of this species is due to over fishing. They suggest that anecdotal approaches can be used to guide focused research and large scale underwater visual censuses to quantify a more robust abundance index. This suggests that while the local extinction of *B. muricatus* in future cannot be ruled out, more diverse and quantitative data are urgently needed to properly assess its current status.

Munday (2004) reported the disappearance of two clown gobies from Kimbe Bay in Papua, New Guinea (PNG). One is documented as locally extinct *Gobiodon* sp. A (Gobiidae), although it occurs in other parts of the western Pacific; the other *Gobiodon* sp. C (Gobiidae) is considered at risk of global extinction, unless its geographical distribution can be confirmed to be greater than that described for Kimbe Bay. Both gobies have low local abundance and are obligate dwellers of the coral genus *Acropora* (Acroporidae); Gobiodon sp. A, is only found in *A. tenius*, whilst *Gobiodon* sp. C is confined to *A. elseyi* (Munday and Harold 1999). These gobies have suffered serious population declines in Kimbe Bay because of considerable loss of coral habitat on near shore reefs, likely due to coral bleaching and sedimentation.

The local extinction of *Gobiodon* sp. A is based on a positive identification during surveys conducted in 1996 and 1997 and a negative one carried out in 2003. The period between surveys was characterized by an increased frequency of coral bleaching and elevated sedimentation (at the end of 1990s) on coastal reefs in PNG. As these reefs seem to be currently recovering, it may be possible for *Gobiodon* sp. A to recolonize the area in the near future, via larval dispersal from less disturbed areas (Philip Munday, personal communication). The fate of *Gobiodon* sp. C is of greater concern, because it has only been reported from Kimbe Bay, where it inhabits coastal reefs and lagoons. These habitats are the areas of the bay most threatened by the joint effects of coral bleaching and terrestrial disturbances.

Some hold that any candidate for extinction must be supported by a clear, unambiguous taxonomic identity. For instance, according to the new criteria proposed by the Committee on Recently Extinct Organisms (CREO), the rationale for adopting this approach is that when compiling a survey of extinctions, we: (i) need to identify what units to count, and (ii) want the manner in which biologists identify these units to be as uniform as possible. Their rationale is that, because we know different people often adopt different species concepts, we should ask biologists to adopt an approach which comes as close as possible to a uniform way of counting units [Committee on Recently Extinct Organisms (CREO) 2007].

The North Sea houting (*Coregonus oxyrhinchus*, Salmonidae) was regarded as extinct in the Netherlands by 1938 (Kranenbarg *et al.* 2002) and in the Wadden Sea area by the late 1980s. However, two monitoring programmes in estuarine and freshwater systems in the Netherlands show an increasing trend in the numbers of North Sea houting since 1997, reporting up to 120 individuals caught in 2001 (Kranenbarg *et al.* 2002). Additionally, stocking with fingerlings in a 4-year period in the larger rivers of the Wadden Sea, together with anti-fishing protection measures, have rehabilitated the species. It is now common in the area, but still protected (Walday and Kroglund 2003).

In 1998 the barndoor skate (*Dipturus laevis*, Rajidae) was regarded as the first near-extinct, widely distributed fish whose range-wide decline was caused by overexploitation as bycatch (Casey and Myers 1998). This hypothesis was based on analysis of research survey data from the Grand Banks and the Scotian Shelf in Canadian Waters, and George Bank south to the Carolinas in US waters. Its reported geographical range extends from as far north as south-western Grand Banks and Gulf of St Lawrence, south to the waters off north-eastern Florida. However, bycatch records from commercial fisheries shows that the distribution of this species extends much further north than indicated by survey data, indicating their presence along the shelf edge as far north as 62°N (Kulka *et al.* 2002).

Barndoor skates were certainly more common in the 1950s and 1960s compared to later decades. They were only sporadically encountered in the 1970s, and observations were rarer still in 1980s to early 1990s. Since the mid-1990s, however, abundance has increased throughout the central/western Scotian Shelf and Gulf of Maine area. These patterns have been corroborated by seasonal surveys conducted by NMFS in US waters. Recent data suggest that barndoor skate is currently sufficiently numerous to ease concerns about its conservation status (Simon *et al.* 2002), let alone its extinction.

There are other fish species that have not been recently reported in parts of their range, and have suffered important population declines elsewhere, thus deserving special attention and conservation concern. Examples include the Pacific lamprey (*Lampetra tridentata*, Petromyzontiformes) and the common skate (*Dipturus batis*, Rajidae). In the North Pacific, *L. tridentata* occurs in Bering Sea coasts of Asia and Alaska southwards to the Yuhutu River, Hokkaido, northern Japan and Punta Canoas, central Baja California, Mexico (http://www.fishbase.org). The decline of the Pacific lamprey has been documented since the early 1990s (Moyle 1994). Close (2002) reported that counts of Pacific lamprey at Winchester Dam, located in the coastal Umpqua River, USA, decreased exponentially in numbers from a maximum of 46 785 in 1966 to just 34 fish in 2001. Counts at Ice Harbor Dam in the Snake River, a tributary of the Columbia River, USA, similarly decreased from a maximum of 49 454 in 1963 to 203 lamprey in 2001. These declines are closely correlated with human disturbance such as flow regulation, channelization or poor water quality. Given the negative perception that most people have towards lampreys, its ecological and cultural role and current conservation status has not been fully appreciated (Close 2002). However, the population trends of *L. tridentata* in other parts of its distribution is poorly known, and there are documented cases of the re-appearance of non-anadromous lamprey species assumed extinct for more than 40 years (Lorion *et al.* 2000). The Pacific lamprey is not Red Listed.

The common skate occurs in Norway, Iceland and the Faeroes, down to Senegal, including the western Mediterranean and western Baltic. This species may be extirpated from the Irish Sea, mainly by trawling (Brander 1981; Frisk *et al.* 2002; Abdulla 2004; Dulvy and Polunin 2004). In the last few decades its range appears to have retracted to the northern North Sea. It is still caught off Shetland Isles, for instance. The common skate is still locally abundant in the west coast of Scotland, where landings in 2001 were near 184 t whilst in the northern North Sea it was 80 t and around

Rockall almost 10 t. It is also present in the Tyrrhenian Sea, as well as in other parts of the Mediterranean, although in relative low numbers. in France, Spain, the Celtic Sea, Barents Sea, Norwegian Sea, Iceland and East Greenland [Commission of the European Communities 2003; Department of Environment Food and Rural Affairs (DEFRA) 2005; International Council for the Exploration of the Sea (ICES) 2005].

As for other taxonomic groups, we found that Thibaut *et al.* (2005) documented the local extinction of seven algal species of the genus *Cystoseira* and *Sargassum* (Fucales) in the Albères coast, France (±35 km). Overgrazing by sea urchins, out-competition by mussels, habitat destruction, sampling for scientific research and, probably, human trampling and chemical pollution, are to blame for the decline of populations which once thrived in shallow waters. Deep-water species have been affected by an increase in water turbidity and, probably, chemical pollution and direct plant destruction attributed to net fishing. The ecological consequences of these losses are huge, as these natural structural engineers contribute to the functioning of entire communities. When absent, the underwater landscape becomes homogenized.

The work of Thibaut *et al.* (2005) consists in a revision of all available documentary information of *Cystoseira* and *Sargassum* in the region, including scientific field diaries. Additionally, they carried out field work from April to August 2003 in the sites sampled by previous phycologists, and other areas. The authors support their assertions on the basis of comparisons between their survey and historical information.

For instance, *C. crinita* was not found during the 2003 survey and the last record was in 1981; as such, they concluded that the species is extinct in the area. However, the chronology of sightings shows that the largest time lag between two consecutive sightings is 25 years (1937, 1962). If we calculate the upper limit of the extinction date for this species, applying the statistical estimator of Roberts and Solow (2003) on the data prior to 1962, it turns out to be 1961, 1 year before the next sighting record. Moreover, the geographical distribution of the species described in Thibaut *et al.* (2005) is far greater than the ±35 km sampled in their survey (http://www.algaebase.org). This exercise suggests that potential disappearances must be carefully contextualized in time and space before they can be labelled as extinctions. Although increasing habitat loss in this, and other regions of the Mediterranean Sea, is badly affecting importantly macro-algal communities, management measures designed to increase densities and strength of these organisms will probably improve the state of the remaining populations of Fucales (Thibaut *et al.* 2005).

A checklist under scrutiny

In spite of our critiques of many individual cases, the in-depth review of Dulvy *et al.* (2003) is valuable in providing an overarching perspective on marine extinctions. Their painstaking analysis revealed that extinctions (in the broad sense) in the marine realm are considerably more common than previously thought (Carlton *et al.* 1999; Roberts and Hawkins 1999). They list 133 probable extinctions of which 21 are global, 4 regional and 108 local, pertaining to 118 species.

Human fisheries have had particular influence on global marine extinctions; almost 50% (10 of 21 cases) were assumed to be a consequence of overexploitation. These include mammals such as Steller's sea cow, the Caribbean monk seal and the sea mink, as well as large birds taken for food and feathers such as the Great auk (*Pinguinus impennis*, Alcidae), the Labrador duck (*Camptorynchus labradorius*, Anatidae), Pallas's cormorant (*Phalacrocorax perspicillatus*, Phalacrocoracidae) and Auckland Islands merganser (*Mergus australis*, Anatidae). This category included just one fish, the New Zealand grayling, and two invertebrates, the horn snail and the white abalone. With the exception of the white abalone, which we claim is not globally extinct, all these species share certain attributes that make them more sensitive to human and natural influence, such as large body size, restricted distribution ranges and amphidromy (Del Monte-Luna and Lluch-Belda 2003; Brook and Bowman 2005). It is worth noting that more than 70% of these global extinctions are presumed to have occurred prior to the first half of the last century.

Of the remaining 11 cases of global extinction, three were apparently a consequence of natural causes with two of them driven by extreme ENSO events and one by an epidemic disease. The Periwinkle, Bennett's seaweed, the Canary Islands oystercatcher (*Haematopus meadewaldoi*, Haematopodidae) and the Green wrasse had extremely restricted geographical distributions and thus were heavily impacted by localized habitat loss.

The qualification of at least three other invertebrates and one alga as globally extinct is open to question and needs further support. In our view, the grey whale represents the only irrefutable case of regional extirpation caused by overexploitation, while the case of the Chinese bahaba from South China remain inconclusive and the purple sea urchin not only is not regionally extirpated but is actually recovering.

From the 108 local extinctions reported in Dulvy *et al.* (2003), 8% occurred in the Adriatic Sea, 10% in the Gulf of Lions, 13% in Bermuda and 35% in the Wadden Sea, The Netherlands; the remaining 30% were reported from different parts of the Northern Hemisphere. The nine records from the Adriatic Sea were all the result of a trawl survey conducted during spring–summer of 1998, 50 years after a similar one during 1948. With such a low level of sampling intensity, one could hardly regard any of these species as likely to be actually extinct, even locally, considering that the second survey sampled 25 species that were not obtained during the first.

Dulvy *et al.* (2003) indicated 12 instances of local extinction in the Mediterranean Sea's Gulf of Lions, according to surveys performed by commercial and experimental trawling from the late 1950s to 1995 (Aldebert 1997). While the results are incontrovertible in terms of documenting the decline in abundance of several elasmobranch species at the fishing grounds, they are not appropriate for assessing extinction. Indeed, as Aldebert (1997) stated, commercial trawling induces a considerable bias by not sampling all areas, such as rocky grounds and canyons that may serve as natural refuges. Furthermore, given the design of surveys, only those in the last period (1983–1995) are strictly comparable, yet 7 of the 12 reported extinctions occurred before this period. Further, nine elasmobranch species were not captured for a period of 10 years and yet reappeared later in the sampling period. The Kitefin shark (*Dalatias licha*, Dalatiidae), for instance, disappeared twice, accounting for almost 17 years of intermittent absence out of a total of 28 years of sampling.

The 14 reports from Bermuda come from Smith-Vaniz *et al.* (1999) and represent 13 fish and one mud shrimp species. Dulvy *et al.* (2003) do not make it clear why they reported only 13 of the 35 fish species catalogued by Smith-Vaniz *et al.* (1999) as unreported for >50 years. Of these, a single species, the Buffalo trunkfish (*Lactophrys trigonus*, Ostraciidae), is regarded as a resident but rare in Bermuda; whilst the affinity of the other 12 species to this archipelago is questionable. For example, the smalltooth sawfish, commonly associated with coastal environments and surface streams, is unlikely to occur around oceanic islands such as Bermuda. The only likely instance of extirpation ('local extinction') is that of the Arrow stargazer (*Gillellus greyae*, Dactyloscopidae), though even this assertion is considered uncertain (Smith-Vaniz *et al.* 1999). Most of the species deemed locally extinct were affected by habitat degradation, a sizeable toll of which occurred between 1941 and 1944 during the extensive dredging of Castle Harbour for construction of the Kindley Air Force Base. The last collection of the stargazer pre-dates this development, however (Smith-Vaniz *et al.* 1999).

The case of the Wadden Sea is particularly problematical. It is a 450-km long, shallow coastal region with an average width of 10 km and it consists of a highly dynamic ecotone subject to intensive habitat alteration and biological dynamism. Natural variability, ranging from seasonal to multi-decadal (Weijerman *et al.* 2005), induces enormous fluctuations in animal and plant populations and is a characteristic feature of this ecosystem. Thus, the evolution and species composition of any given site at any particular time in the Wadden Sea is naturally highly variable and unpredictable (Reise 1994). It is possible then that many putative extinctions might represent the loss of pseudo-populations – groups of individuals which do not reproduce *in situ*, and usually occupy the edge of the species distributional range (Gaston 2003).

Other mammal species, such as the harbour porpoise (*Phocoena phocoena*, Phocoenidae), are still resident in the Wadden sea, although not in coastal waters. Local numbers of grey seals (*Halichoerus grypus*, Phocidae) and harbour seals (*Phoca vitulina*, Phocidae) are currently growing at a faster rate than can be explained just by an increase in births (Wolff 2000a). Birds regarded as extirpated are either abundant elsewhere or recovering exponentially (eider duck, lesser black-backed gull and the common gull (*Larus canus*, Laridae). Two other species, the white tailed sea eagle (*Haliaeetus albicilla*, Accipitridae) and the osprey (*Pandion haliaetus*, Accipitridae), are increasing in other North Sea countries, and can be expected to recolonize the area soon.

Several fish species listed by Dulvy *et al.* (2003) are anadromous and were either over-fished or diminished by habitat alteration along their migratory routes. One of the fish species, the meagre or Atlantic shadefish (*Argyrosomus regius*, Sciaenidae), was believed last seen in the year 50 AD, and would therefore not be included in most lists of contemporary extinctions, which mark time from 1500 onwards (MacPhee and Flemming 1999). Nonetheless, it persists in the North Sea, albeit in extremely low numbers (Wolff 2000a). Conversely, some invertebrates listed as locally extinct from the Wadden Sea are common worldwide and all are cultured species, including the edible oyster (*Ostrea edulis*, Ostreidae) the waved whelk (*Buccinum undatum*, Buccinidae) and the European lobster (*Homarus gammarus*, Nephropidae). As such, it is difficult to argue that their local disappearance from the Wadden Sea is a biologically significant event for the species as a whole.

In sum, our detailed documentary scrutiny revealed that of the 21 species reported by Dulvy *et al.* (2003) as globally extinct, only 16 can be confirmed as such and all attributed to human impact. Similarly, only one of four regional extinctions and 50 of 108 local extinctions seem valid. Given these revised estimates, the figure of 133 reported in the reference paper may have overestimated the verifiable number by 50% (67 of 133). Thus although Dulvy *et al.* (2003) took care to identify a suite of general caveats associated with attempts to recognize true declines and extinctions from false ones, our analysis highlights the considerable scope for improvement in the reliability of 'extinction lists' based solely on the careful consideration of all known biological and technical circumstances connected with each individual case.

Concluding remarks

Reliable estimations of extinction are clearly very difficult to assess, and there is a pressing need to develop more efficient methods and criteria to do so, especially when applied to the marine realm. We suggest that the scientific community must be much more careful and standardized in its use of terminology; the declaration that a species is 'extinct' should occur only after the available information points to it beyond reasonable doubt. A laissez-faire classification method can lead to higher evils. For instance, we risk the loss of credibility from decision makers who often are not scientists, including members of nongovernmental organizations, park managers, and other personnel from government agencies.

A situation like this is especially delicate in developing countries, because in many instances their endangered species lists are not underpinned by an analyses of the locally relevant data, being instead simply extrapolated from international criteria. In such instances a strong investment of human and material resources might be implemented which ultimately yields little of conservation value to the society.

When declarations of extinctions are premature, or improperly supported by all available scientific evidence, they can potentially affect the public image or perception of conservation science. This is true even if the analyses or predictions themselves are robust, but the timing and/or magnitude are uncertain. Examples of a backlash include retorts against the predicted effects of bleaching and global warming on coral reef health in Australia (Bolt 2006) and the dialogue on the consequences of global warming (Marohasy 2006). Given these difficulties, we suggest that national and international jurisdictions ought to focus attention on species suspected to have declined to less than between 1% and 5% of known historical abundance, as inferred from the best available information.

There are of course cases of *K*-selected species, such as the Pacific grey whale and Northern elephant seal (*Mirounga angustirosris*), which have recovered from population reductions of up to 99% following conservation intervention [Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) 2000; Weber *et al.* 2000; Committee on the Status of Endangered Wildlife in Canada (COSEWIC) 2004]. Similarly, some *r*-selected exploited species, such as the California sardine and anchovy, not only have repeatedly endured heavy declines but have later re-appeared after prolonged (+30 years) absences from large parts of their pristine distributional range, due sometimes to causes unrelated to fishing pressure (Lluch-Belda *et al.* 1992). Such success stories indicate that national and international efforts should also always try carefully to consider the evidence for different causes behind each case of extinction or extirpation such as exploitation, habitat loss or climate variability, in order to establish or appropriately adapt management actions.

Finally, future checklists of marine extinct species must be revised, in depth, by specialists in each of the particular taxonomic groups involved. This practice will bring many benefits, including the use of the most precise, relevant and up-to-date information available, including unpublished data, the elimination of many taxonomic problems, and the promotion of multidisciplinary approaches to species conservation.

Acknowledgements

We thank Jon Elorduy, Francisco Arreguín, Gustavo de La Cruz, José de La Cruz, Rafael Riosmena, Philip Munday, Ernesto Chávez and two anonymous referees for their valuable comments on the manuscript. We also thank financial support of the Instituto Politécnico Nacional through the Estímulo al Desempeño de los Investigadores, Comisión de Operación y Fomento de Actividades Académicas, Secretaría de Investigación y Posgrado (SIP 20070254), the Programa Institucional de Formación de Investigadores and to the Sistema Nacional de Investigadores of the Consejo Nacional de Ciencia y Tecnología.

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