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Marine turtles and IUCN Red Listing: A review of the process, the pitfalls, and novel assessment approaches

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Abstract

Marine turtles have been exploited by humans since pre-history, with particular intensity in the last century, the result of which has been the depletion of most nesting populations in the world. In many cases these declines have been reversed thanks to a variety of effective conservation programs. Several nesting populations maintain positive growth trends, although most are probably depleted relative to historic levels, while others continue in a severely depleted state, with little or no population growth in recent decades. This mosaic of population trajectories along with demographic and life-history traits that buffer against extinction has created unique challenges for marine turtle assessments such as those by the World Conservation Union's (IUCN) Marine Turtle Specialist Group, which conducts global assessments for the IUCN Red List. While the Red Listing approach describes extinction risk, which theoretically can be useful for developing conservation priorities, the descriptors that have been assigned to marine turtles so far (e.g. Vulnerable, Endangered, Critically Endangered) state an unrealistic imminence of extinction, a problem enhanced by the fact that its global resolution fails to reflect the disparate population trends ongoing in different regions worldwide. Coupled with misuse of the Red List by governments and conservation organizations worldwide, these shortcomings have led to increased debate regarding its efficacy for marine turtles. In this paper we describe the Red Listing assessment process, the problems associated with this approach for marine turtles, as well as the overall value of Red List assessments for marine turtle conservation. We suggest that Red list assessments for marine turtles at the global scale do not accurately depict the current status of marine turtles and may have unintended consequences for their conservation. Largely the data do not exist, or are not reliable, making the use of the current criteria intractable. We discuss novel methods for conducting marine turtle assessments, such as using a wider array of the current Red List Criteria, modelling future population dynamics, and developing regional assessments and/or conservation prescriptive assessments.

Keywords: Assessment; IUCN Criteria; Endangered; Modelling; Population trend; Sea turtle

1. Introduction

The conservation of large endangered vertebrates has occupied center stage in the world of nature conservation. Like other charismatic mega-vertebrates such as tigers, whales and pandas, marine turtles are among the most visible icons for endangered species. As with several of these other vertebrates, the notion that they are on the 'brink of extinction' has been widely, although perhaps wrongly, accepted by laypersons, wildlife managers, and scientists alike (Mrosovsky, 2002). This attitude may be due to the focus on a few truly endangered populations by mainstream media that are incorrectly taken to represent the situation for all species and populations, likely fueled by our general tendency to focus on the negative. Even in science, we are guilty of sustaining the misconception, as the terms Threatened or Endangered are commonly found in the opening paragraphs in many peer-reviewed papers on marine turtles when describing the species' status, even though the specific populations under study have often been anything but endangered (Diez and van Dam, 2002; Godley et al., 2003; Seminoff et al., 2003; Dutton et al., 2005; Troëng and Rankin, 2005; Richardson et al., 2006). This, of course, is not to say that

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the vast majority of populations have not been or continue to be severely depleted due to anthropogenic threats, as indeed they are. However, to assume that all populations are doing poorly is incorrect and irrational, as many populations are far from going extinct, not merely holding their own, but actually increasing.

The challenges of biodiversity measurement and assessment of population abundance change are generic and recur in various forms for almost all taxa. In the case of whales, perhaps the greatest of all marine conservation icons, there is increasing recognition of the existence of regionally distinct populations and the need to illuminate these as 'conservation units', which has resulted in concerns about the value of conducting global cetacean assessments (B. Taylor, pers. comm.). Efforts to assess marine fishes have suffered from similar issues, particularly when applied to commercially exploited species (Dulvy et al., 2004). This is also true of large terrestrial flagship vertebrates such as Asian elephants that have been widely described as endangered (e.g. Vidya et al., 2005) despite uncertainty in population assessment approaches and the resulting estimates (Blake and Hedges, 2004). Nevertheless, although there is a lack of consensus regarding the most favorable approach for assessing species, stocks, or populations, there is no doubt that such efforts are necessary for effective wildlife management.

Key to marine turtles and conservation in general is the need to illuminate which populations are doing well and which are truly in danger of disappearing from our planet. This is a critical first step for developing conservation strategies. For marine turtles, management decisions regarding common themes like bycatch reduction and nesting beach protection, as well as more sensitive issues such as sustainable harvest and indigenous use, all require information on the status of marine turtle populations being impacted. Although few would argue this point, consensus regarding the most appropriate status assessment technique has been elusive. With respect to marine turtles, there have been several global assessment initiatives (Table 1), perhaps the most widely recognized is that by the World Conservation Union's (IUCN) Marine Turtle Specialist Group (MTSG) which has assessed the global statuses of marine turtles for the IUCN Red List since the 1960s. To conduct Red List assessments, the MTSG as well as other IUCN Specialist Groups apply a series of rule-based assessment criteria to describe a species' global extinction risk, using terms such as Least Concern, Near Threatened, Vulnerable, Endangered, and Critically Endangered (Appendices 1, 2).

The Red List is intended to be an objective system for classifying all species according to their risk of extinction that can be applied consistently by different people (IUCN, 2001; Mace et al., 2006). In theory, the system should facilitate comparisons across widely different taxa, which could be useful for developing conservation priorities (Mace et al., 2006; Marsh et al., 2007). However, it is paradoxical considering that endemic species with restricted ranges are likely to have profoundly different extinction probabilities than a globally distributed species such as a marine turtle (Heppell et al., 2000; Mazaris et al., 2005). This disparity has resulted in considerable skepticism regarding the current statuses of marine turtles on the Red List, a sentiment fueled by perceived inadequacies of the current assessment criteria (Webb and Carillo, 2000; Mrosovsky, 2003; Seminoff, 2004b, Broderick et al., 2006). Yet whether or not these concerns are warranted, an unfortunate consequence of this debate has been that it detracts attention from the reality that most marine turtle populations are conservation dependent. These complications are further enhanced by that fact that, although the Red List is intended to be a scientific assessment of extinction risk, it is often used for setting policy and instigating wildlife conservation advocacy, purposes that it was not designed to serve (Possingham et al., 2002).

In this paper we examine the process of Red Listing, its products and consequences in detail, using the case of marine turtles. We also explore the IUCN Red List categories and criteria and elaborate on possible future methods to conduct assessments, including a more regional approach and incorporating threats and conservation recommendations into assessments. The discussions herein add to a dialogue that will hopefully lead to assessments that more accurately depict the status of marine turtles and other widely distributed or long-lived taxa, and are more broadly accepted and applicable to local and regional conservation efforts.

Table 1
Summary of global assessment initiatives for the seven species of marine turtles

Sponsoring organization	Species	Assessment criteria	Abundance trends	Extinction risk	Threat summary	Conservation recommendations	Policy relevance	Citation
MTSG	Dc, Cc, Nd, Lk	IUCN 1996	●	●	●		IUCN Red List	See Table 2
MTSG	Cm, Lo, Ei	IUCN 2001	●	●	●		IUCN Red List	See Table 2
USFWS	Dc, Cc, Lo, Lk, Ei, Cm	USFWS, unpubl.	●	●	●	●	Endangered Species Act	NMFS & USFWS, 1998
CITES	All species	Qualitative			●	●	International trade restrictions	
FAO	Dc, Cc, Lo, Nd, Cm	Qualitative	●		●	●	Fisheries management	FAO, 2004
SWoT	Dc, Cc	Quantitative	●		●		None	Mast et al., 2006b, 2007
IAC	All species	Qualitative	●	●	●	●	Regional conservation	Pritchard, 2002; Chacon, 2002

Organization names include: MTSG, IUCN Marine Turtle Specialist Group; USFWS, United States Fish & Wildlife Service; CITES, Convention on International Trade in Endangered Species of Wild Flora and Fauna; FAO, Food and Agriculture Organization of the United Nations; SWoT, State of the World's Turtles; IAC, Inter-American Convention for the Protection and Conservation of Sea Turtles. Species codes include: Dc, leatherback; Cc, loggerhead; Lo, olive ridley; Lk, Kemp's ridley, Ei, hawksbill; Nd, flatback; Cm, green turtle.

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2. Red Listing in practice

2.1. The 2001 Red List Criteria

Since the initial inclusion of hawksbills and leatherbacks on the IUCN Red List over four decades ago (IUCN, 1963), a variety of criteria for assessing a species' global status have been implemented. Originally these assessments were a largely qualitative process, in which the statuses of each species were determined based on consensus opinion among taxon-specific experts around the world. Today, Red Listing is more quantitative, following criteria that are generalized to facilitate consistent application and comparisons among taxa. Established in 2001, the current Red List Criteria (hereafter referred to as the '2001 Criteria'; Appendix 2) detail a variety of approaches for classifying a species' extinction risk based on changes in global population abundance, distribution, degree of fragmentation and quantitative extinction risk modelling approaches. The 2001 Criteria call for these attributes to be assessed over a 10-year or 3-generation time frame, the latter of which is used for marine turtles and other long-lived animals. Recall that for marine turtles, a single generation (defined as the mean age to maturity + 1/2 the reproductive longevity; Pianka, 1974) may be 40 years or more, which would result in total duration of at least a century and perhaps as many as 150 years during which population changes are to be determined (Chaloupka and Musick, 1997; Seminoff, 2004a).

The IUCN criterion most commonly applied to marine turtles is Criterion A, which prescribes ways to characterize extinction risk based on thresholds in population abundance change over a 3-generation timeframe (Appendix 2). The temporal interval for this assessment can encompass past, present, and projected future trends, although to date, the assessments undertaken with the 2001 Criteria have focused on a timeline starting 3 generations in the past and continuing through to the present (Criterion A-2; Seminoff, 2004a; Abreu-Grobois and Plotkin, 2007; Mortimer and Donnelly, in review)(Table 2).

Additional methods have also been outlined in the 2001 Criteria (Appendix 2), although they have not yet been applied to marine turtles. These include strategies such as making status designations based on changes in the geographic range of a population (Criterion B), absolute number of mature individuals in the population (Criteria C and D), and quantitative analysis to determine the probability of extinction in the wild (Criterion E). Although these additional criteria show a degree of flexibility within the current Red List framework, most are considered inappropriate for marine turtles or widely distributed marine species in general, as they may not adequately reflect population decline or overall depletion. For example, Criterion B, which determines a species' status based on changes in its geographic range, cannot adequately capture declines in a sea turtles because even after a severe reduction in population size, the migrations of a few individuals would largely maintain the original range for a population. Criteria C and D, which base status listings on the reduction of population sizes for mature individuals below specific population size thresholds, would also not reflect the conservation dependence of marine turtles due to the fact that even the most depleted of species (e.g. Kemp's ridley, *Lepidochelys kempii*) have adult populations that are well above the size threshold necessary for being listed in a threat category (i.e., Vulnerable, Endangered, Critically Endangered; Appendix 2). Perhaps the only criterion other than Criterion A that is applicable to marine turtles is Criterion E, which allows for inference and projection based on extrapolation of current or potential threats into the future (including their rate of change), or of factors related to population abundance or distribution (Appendix 2).

2.2. Measuring population size

Because marine turtles spend the vast majority of their lives in the marine environment, monitoring and assessment efforts have targeted marine turtles during their terrestrial life-history phase, while females come ashore to nest. Annual reproductive effort has been determined by monitoring projects during which

Table 2
Summary of current status of marine turtles on the IUCN Red List

Species	Red List status	Year	Assessor
Leatherback (*Dermochelys coriacea*)	Critically Endangered	2000	L. Sarti Martínez (MTSG)
Hawksbill (*Eretmochelys imbricata*)	Critically Endangered [a, b]	1996	Red List S & PS
Kemp's ridley (*Lepidochelys kempii*)	Critically Endangered	1996	MTSG
Olive ridley (*Lepidochelys olivacea*)	Vulnerable [c]	2007	A. Abreu-Grobois and P. Plotkin (MTSG)
Loggerhead (*Caretta caretta*)	Endangered	1996	MTSG
Green turtle (*Chelonia mydas*)	Endangered	2004	J. Seminoff (MTSG)
Flatback (*Natator depressus*)	Data deficient [d, e]	1996	Red List S & PS

Five assessments were conducted by the Marine Turtle Specialist Group (MTSG) or biologists therein, and two assessments were conducted by the Red List Standards and Petitions Subcommittee (S&PS).

[a] This revised assessment is a ruling made by the Red List S & PS in response to a petition that challenged the Critically Endangered status (for further details see the IUCN SSC web site). The Critically Endangered status was subsequently justified in a global review by Meylan and Donnelly (1999).

[b] The most recent global assessment of the hawksbill turtle, which recommends a listing of Critically Endangered, is currently in review with the MTSG (Mortimer and Donnelly, in review).

[c] The current assessment was accepted in September 2007. Although this document was in preparation for several years, its submission to IUCN came after an official appeal to IUCN for MTSG to develop a new assessment based on 2001 Red List Criteria.

[d] This revised assessment is a ruling made by the Red List S&PS in response to a petition that challenged the Vulnerable status (for further details see the IUCN SSC web site).

[e] There is currently a draft assessment of the flatback turtle in review with the MTSG (Whiting, in review).

females, nests, or eggs are quantified on fixed length of beach over the course of a nesting season (Schroeder and Murphy, 1999). Although a tremendous advantage of this approach is its logistic feasibility — even projects with few financial resources can muster the human-power to monitor a beach — it is by no means an error-free approach, as there are limits to what nesting beach counts actually measure.

When determining annual reproductive effort, there is a major caveat that must be recognized relating to the proportion of the adult females that nest in any given year. On numerous occasions it has been shown that the proportion of a population's adult female cohort nesting each year oscillates over decadal or longer time frames (Limpus and Nicholls, 1988; Miller, 1997; Hays, 2000; Broderick et al., 2001). These oscillations may be affected by environmental processes such as the El Niño Southern Oscillation (Limpus and Nicholls, 1988; Saba et al., 2007), making it even more difficult to infer population trends and human impact. Unless this inter-annual variability is accounted for, and unless assessments are based on long-term time series data, the trends determined from nesting beach reconnaissance programs may not accurately reflect changes in *total* population abundance (Hays, 2000; Broderick et al., 2001; Mrosovsky, 2003).

An additional problem is that evaluations based exclusively on nesting activity fail to consider the adult males or juvenile cohorts within a population, the latter of which can be problematic for determining population trends. The large age to maturity for most hard-shelled marine turtles means that populations will have numerous immature cohorts that are not counted by nesting beach monitoring efforts (Crouse et al., 1987; Chaloupka and Musick, 1997; Heppell, 1998). Even populations with vastly depleted immature cohorts and only a handful of reproductive adults could trickle along for decades due to the continual maturation of a few adults each year. This is also facilitated by the mating system of most if not all marine turtle species, in which males are capable of inseminating numerous females (Hoekert et al., 2002; Lee and Hays, 2004; Zbinden et al., 2007), and may thus be able to sustain their reproductive role even at severely reduced levels. In this light, the pitfalls of basing population trends on the count of nesting females become apparent: namely that such assessments provide information on only a small segment of the population, of which only a portion (and a highly variable portion at that) of individuals are available for counting each year. With such an approach, populations that are seemingly healthy as determined via nesting beach counts may potentially be eroding from the bottom up due to immature stock's inability to replace older adults as they die. Appropriately referred to as a 'population time-bomb' (Mortimer, 1991; Fig. 1), this potential should be of great concern for populations that are subjected to excessive egg harvest or overexploitation of juveniles in marine habitats.

By the same token, such dependency on counts of adults may also lead to overestimation of extinction likelihoods. For example, a population undergoing recent, but catastrophic exploitation at the nesting beach may appear nearly or completely extirpated when in fact there are numerous juveniles 'in the pipeline' that eventually reach maturity, leading to a restoration of nesting activity (Fig. 2). Although the vastness of threats to marine turtles suggests this 'restoration through maturation' recovery scenario may be less frequent than the 'population time-bomb', it is worth noting that this is the very demographic structure necessary for a nesting population's rebound assuming there is negligible crossover among nesting sites. Considering the ongoing increases in adult nesting activity by several isolated populations that were once at extremely low nesting abundance (e.g., green turtles in Hawaii, Balazs and Chaloupka 2004; leatherbacks in St. Croix, Dutton et al., 2005; Kemp's ridleys, Heppell et al., 2005) it is apparent that counting only adults can mislead by giving an overly negative view of a population's status just as it can give an overly positive view. Further, recovery from low population size has often been considered to be inhibited by the Allee effect, in which populations at low density can suffer from a lower recruitment or a higher mortality, leading to a further population decrease (Courchamp et al., 1999). However, as suggested by Hays (2004) and supported by the aforementioned population recoveries, Allee effects do not seem to affect the recovery of sea turtles.

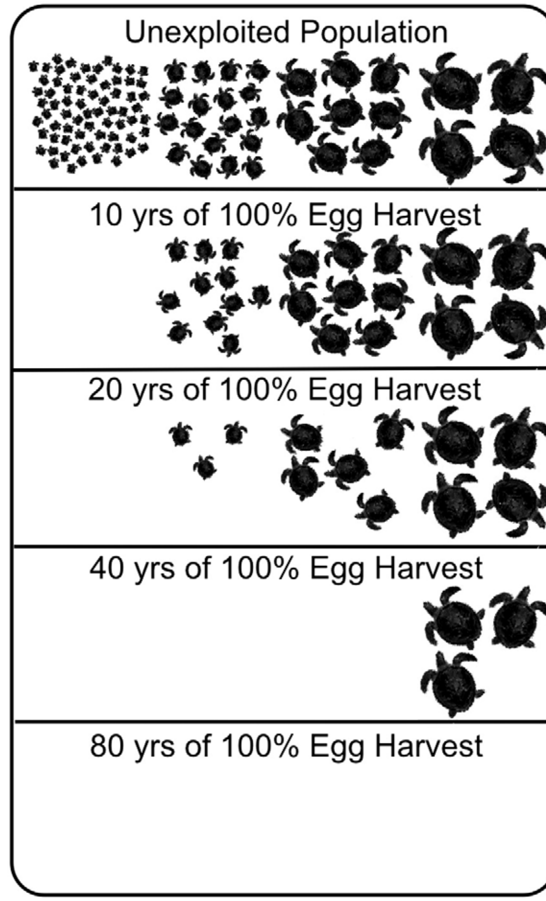

Fig. 1. Population time-bomb: schematic showing the demise of a marine turtle population due to egg harvest over multiple decades. The lack of hatchling production is manifested as a progressive reduction in the number of turtles in each size cohort. Because many marine turtles take 30 or more years to reach maturity, at least 30 years will pass before there is a decrease in the number of nesting females. Modified from Mortimer (1991).

2.3. Implications of shifting baselines

Although extrapolations as per the 2001 Criteria have provided some understanding about marine turtle population sizes

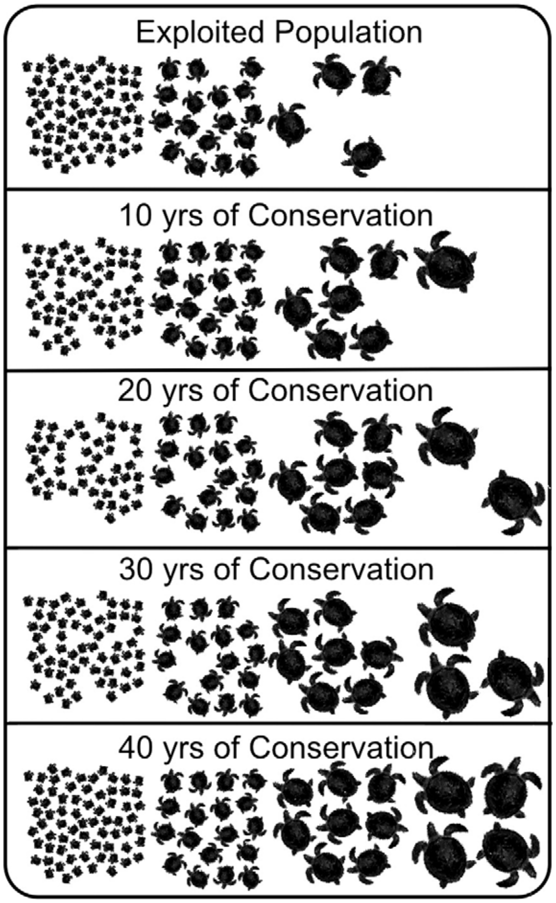

Fig. 2. Restoration through maturation: populations that have undergone catastrophic exploitation at nesting beaches can recover as long as there are healthy juvenile cohorts in the marine environment. The disappearance of nesting females is eventually reversed as juveniles in the population mature. Although the initial depletion of nesting activity creates a negative feedback on hatchling production, this is eventually overcome as the annual nesting activity progressively increases.

in the past, assessments of how today's populations compare to those from pre-exploitation years may be erroneous. In many cases, the greatest rates of marine turtle population declines have taken place prior to the earliest period for which population abundance data are available (McClenachan et al., 2006). This a prime example of the 'shifting baseline syndrome' (Pauly, 1995) and has resulted in populations being characterized as *stable or increasing* even if they are depleted relative to historic levels. For example, the Chichi-jima green turtle nesting population in the Ogasawara Islands of southern Japan has been increasing since the early 1980s (Chaloupka et al., 2007), although harvest data suggest the current population size is substantially lower than that at the beginning of the 20th century (Fig. 3; Horikoshi et al., 1994). Contributing to the initial declines were substantial turtle harvests from at least the 1880s until 1945, which marked the start of occupation by US forces after World War II (Horikoshi et al., 1994). Beginning in the early 1980s, an increase in the nesting population at the Chichi-jima rookery was observed, likely a result of decreased harvest during US occupation (1945–1972) and the resultant increase in egg production. Similarly, at the Seychelles Islands in the western Indian Ocean, annual green turtle reproduction reached its all time low of ~1700 nesting females in the late 1968s, but had increased to nearly 5000 females in the 1990s (J. Mortimer, pers. comm., 2002), indicating a >2 fold increase in annual reproduction over 2 decades. However, this population remains well below that from historic times, as estimates from the early 1900s suggest upwards of 10,000 females nested each year (Hornell, 1927). Perhaps recovery to pre-Columbus levels in unrealistic; however, it is important that the correct baseline is chosen for whatever the classification scheme or recovery goal may be.

The 'increasing but depleted' population abundance dynamic underscores the importance of determining the temporal baseline required for addressing the goals of the assessment being undertaken. If, for example, we are defining ecological roles, or looking at the entire history of the population, then the depletion relative to pre-Columbus levels, regardless of any perceived recent increasing trend would come into focus. On the other hand, if we are characterizing extinction risk, then a population on the rise, no matter how depleted relative to prior levels, should be taken as a good sign, as extinction *per se* would be less likely. However, in the current IUCN system,

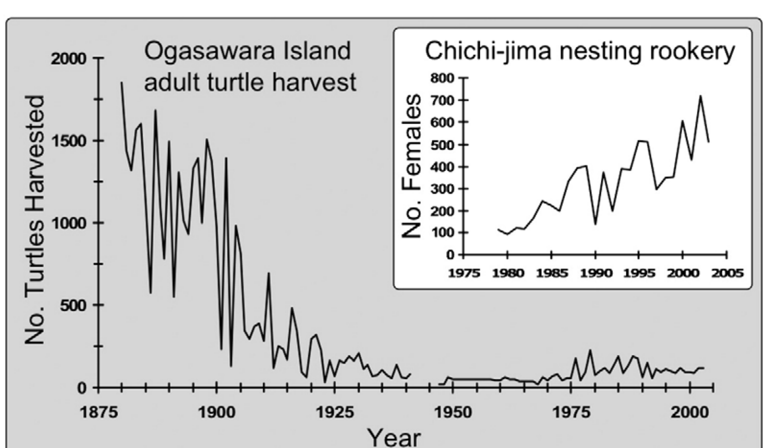

Fig. 3. Recent increase for a historically depleted population of green turtles (*Chelonia mydas*) from Ogasawara Islands, Japan. Larger data summarized in Chaloupka et al. (2007).

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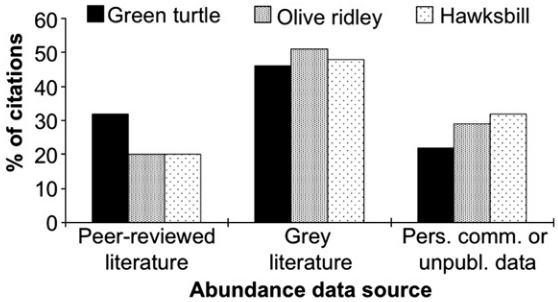

Fig. 4. Summary of data sources for population abundance estimates in three marine turtle Red List assessments. Included here are the MTSG assessments of green turtles (Seminoff, 2004a), olive ridley turtles (Abreu-Grobois and Plotkin, 2007), and hawksbill turtles (Mortimer and Donnelly, in review).

such a positive trend may be overshadowed by the population's depletion relative to abundance 3-generations ago, an unfortunate consequence being that it may be listed in an IUCN threat category even if in a recovery mode.

2.4. Data availability

A major hurdle for Red Listing efforts is the lack of reliable studies that have been published. To date, assessments have had far too much emphasis on grey literature and personal communications (Fig. 4). Because reliable data create the foundation of a good assessment (Holmes, 2001), we consider this deficiency to be a serious problem that should be more openly discussed. While there are numerous grey literature sources that summarize trade and fisheries statistics, which in turn can be used to estimate relative population abundance (Parsons, 1962; Groombridge and Luxmoore, 1989), there is still a great need for published, peer-reviewed long-term datasets. Among the three assessments that have been conducted using the 2001 Criteria (Seminoff 2004a, Abreu-Grobois and Plotkin, 2007; Mortimer and Donnelly, in review), a mean of only 24% of citations was from published literature (Fig. 4). Considering that the green turtle (*Chelonia mydas*), olive ridley (*Lepidochelys olivacea*), and draft hawksbill (*Eretmochelys imbricata*) assessments focused on 32, 31, and 25 index sites, respectively, this indicates that only 0.42 to 1.08 publications were available per index site, which is dangerously low since determinations of long-term population trends usually require at least two (a 'past' and a 'present') datasets on which to base assessments (IUCN, 2001).

Adding to this publication shortfall is the paucity of historic data. With species such as green turtles and hawksbills, both of which have been economically important to humans for centuries, these historic data are more likely to be available, although commonly in unpublished form. For the remaining species long-term data are virtually non-existent. To demonstrate this within the context of marine turtle Red List assessments, we tallied the number of pre-1960 and pre-1970 publications included in the literature cited

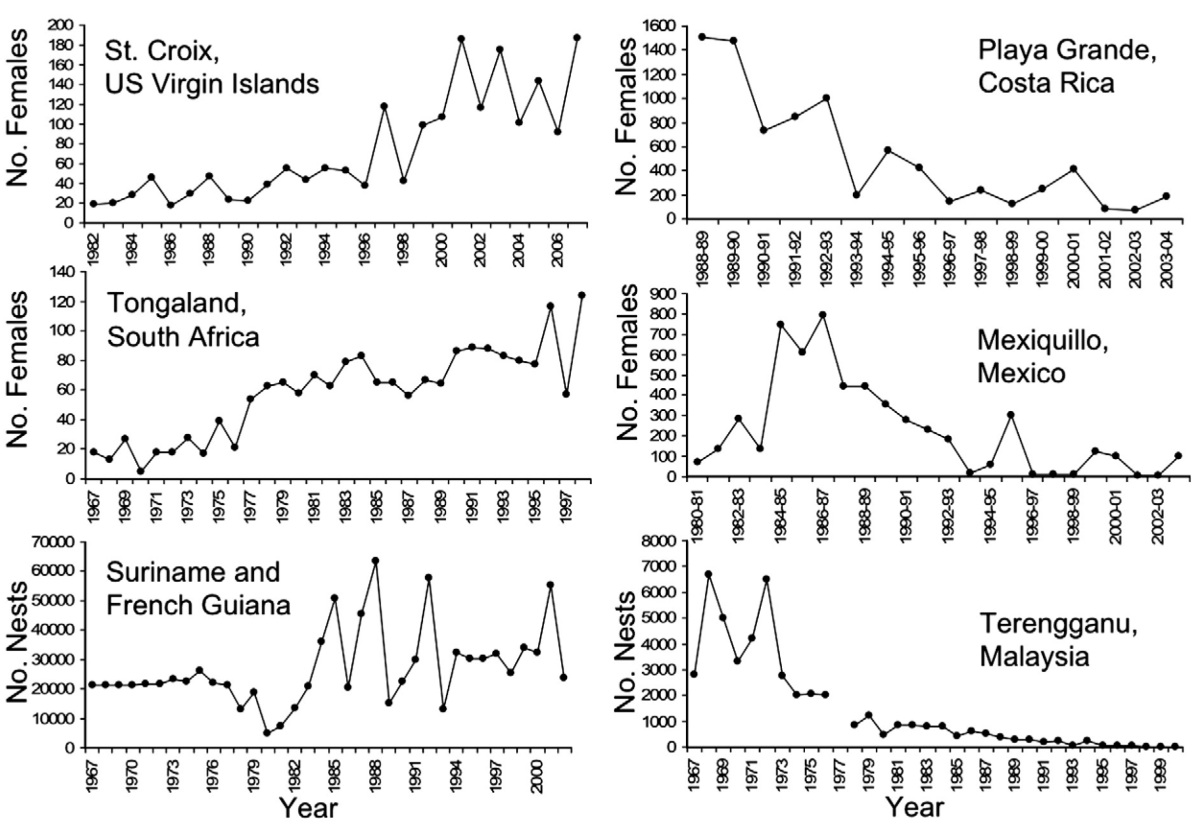

Fig. 5. Positive and negative annual nesting population growth trends for the leatherback turtle (*Dermochelys coriacea*). Data summarized for St. Croix, USVI (Dutton et al., 2005), Tongland, South Africa (Hughes, 1996), Suriname and French Guiana (Girondot et al., 2007), Playa Grande, Costa Rica (Santidrián Tomillo et al., 2007), Mexiquillo, Mexico (Sarti Martínez et al., 2007), and Terengganu, Malaysia (Chan and Liew 1996; Chan 2006).

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sections of the three assessments completed to date using the 2001 Criteria (Table 2). We recognize that the number of publications from any particular time period is only a loose proxy for data availability and that much of the historic data may be summarized in more recent writings, but considering the exhaustive literature searches that were undertaken for each of these assessments, we find the paucity of older publications to be reflective of the lack of published historic data. In both time cut-offs, green turtles had substantially more historic publications (15 pre-1970 citations, 5 pre-1960 citations; Seminoff, 2004b) than did hawksbills (6 pre-1970 citations, 4 pre-1960 citations; Mortimer and Donnelly, in review), or olive ridleys (4 pre-1970 citations, 0 pre-1960 citations; Abreu-Grobois and Plotkin, 2007). Of all pre-1970 citations among the three assessments, only 56% (14/25) were peer-reviewed sources.

Even when historic accounts are available, the quantitative data therein must be viewed with great caution. For example, in a review of population trends for olive ridley turtles in Orissa, Shanker et al. (2004) examined over 20 published sources of population data, and found little concordance between published values, no account of methods, no estimates of variance, systematic errors, and in summary, little evidence that the data were reliable. The fact that these same data had previously been used by other authors to draw conclusions about this population underscores the troubling reality that historic accounts are often interpreted as 'truth' (Mrosovsky, 2002).

3. The products of Red Listing

3.1. How accurate are global listings for marine turtles?

Making an assessment based on the global mean fails to reflect the local or regional differences that may be ongoing within a widely distributed species. In the case of marine turtles, all six species that have been listed in a threat category (i.e., Vulnerable, Endangered or Critically Endangered; Table 2) have one or more nesting populations that are increasing. For example, leatherback turtles (*Dermochelys coriacea*) have been classified as 'Critically Endangered' on the basis of an overall decrease in global nesting trends, particularly in the Pacific (Chan and Liew, 1996; Spotila et al., 2000; Sarti Martínez et al., 2007). However, such a listing fails to reflect the fact that there are stable and increasing nesting populations, some of substantial size, in the Atlantic Ocean (Chacon-Chaveri and Eckert, 2007; Dutton et al., 2005; Thome et al., 2007) and Indian Ocean (Hughes, 1996)(Fig. 5). Likewise, hawksbills are substantially depleted due to centuries of harvest for the tortoiseshell trade (Parsons, 1972; Groombridge and Luxmoore 1989, Meylan and Donnelly, 1999; Fleming, 2001) but they too have populations that are increasing (Richardson et al., 2006, Beggs et al., 2007; Marcovaldi et al., 2007; Mortimer and Donnelly, in review). The same can be said of olive ridley turtles (Shanker et al., 2004; Abreu-Grobois and Plotkin, 2007), green turtles (Seminoff, 2004a; Broderick et al., 2006; Chaloupka et al., 2007) and loggerheads (*Caretta caretta*; Chaloupka et al., 2007, Marcovaldi and Chaloupka 2007), all of which are currently listed as Vulnerable or Endangered.

Based on these examples, it appears that a Red List designation does not reflect a *species*' true risk of extinction. And in fact, the definitions associated with these status listings seem far from accurate: whereas the IUCN defines a Critically Endangered species as one that is "facing an extremely high risk of extinction in the wild", an Endangered species is defined as one that is "facing a very high risk of extinction in the wild" (IUCN, 2001; Appendix 1). We agree that all marine turtle nesting populations are susceptible to declines if additional anthropogenic threats develop in the future, and catastrophic events such as global change and sea level rise are of particular concern (Fish et al., 2005; Baker et al., 2006). But we see no plausible scenario by which anthropogenic impacts, either direct or indirect, could wipe out an entire species within the foreseeable future. Recall that in the case of a globally distributed species, the risk of extinction must be defined as the risk of every last population. Based on these considerations, we argue that no marine turtle *species* is currently endangered with imminent extinction, although we do agree that there are many *populations* that may disappear unless conservation measure are promptly developed and enforced.

3.2. Species databases as a valuable byproduct

Whether or not one agrees with the outcome of Red Listing for marine turtles, the assessment *process* itself provides exceptional value. It serves as a mandate to amass abundance data from a wide variety of sources. It forces a closer examination of data from grey literature, and stimulates researchers to carry out their work more rigorously and publish in peer-reviewed journals. It forces the compilation of a global database that can be accessed by people around the world that may otherwise not have the information resources to gather such data. It highlights gaps in our understanding of nesting abundance and threats around the world. Recognizing where additional data are needed will help us to develop more appropriate research and conservation strategies as well as develop research funding priorities.

3.3. The Red List's influence on marine conservation

The Red List is intended to be "policy relevant, not policy prescriptive" (D. Brackett, pers. comm.), which theoretically should eliminate political and social considerations in the construction of the criteria used for its development (Possingham et al., 2002). That is, the Red List should not prescribe the specific conservation measures necessary to repair damaged populations, but rather highlight where conservation is most needed (Lamoreux et al., 2003; Mace et al., 2006). Indeed, assessing a population's status is quite separate from developing a conservation plan. However, there is a foggy line between what is 'relevant' and what is 'prescriptive' and we suggest that the IUCN Red List is highly influential, perhaps even prescriptive of national conservation programs and conservation legislation regardless of its original intent. In fact, the Red List, like other lists of Threatened Species, is used for a variety of purposes that it was not designed to serve such as for reserve design, to set priorities for resource allocation and to constrain development (Possingham et al., 2002). Supporting this misapplication in the context of marine turtle conservation, Abreu-Grobois and Plotkin (2007) state that “Most of the conservation actions on behalf of the olive ridley at national and international levels have been based on the species’ listing under the Endangered category in the IUCN Red List”, and that “On the basis of the species’ classification in the IUCN Red List or in national endangered species lists, local legislatures of range states confer protection to the olive ridley”.

Because of the variety of users (composed of the public, conservation organizations, and government) the interpretation of the Red List is variable, and most are used for multiple purposes, regardless of the IUCN’s original intent (Possingham et al., 2002). If conservation action were based on independent knowledge based decisions, then the misapplication of the Red List and the lack of spatial scale in a species’ status listing would not be so problematic. However, among the biggest shortcomings described in decision-making theory is that the decision makers tend not to examine all available data and commonly base actions on preconceptions and personal biases (Rittel and Webber, 1973; Hamazaki and Tanno, 2001; Campbell, 2002). For example, although the recent global assessments (Seminoff, 2004a; Abreu-Grobois and Plotkin, 2007; Mortimer and Donnelly, in review) contain substantial information on biology, population abundance, threats which illustrates differences among populations, the reality is that the end product — a one or two word status listing — is what is viewed by the overwhelming majority of practitioners, most of whom never read the actual assessment. We believe this is a major shortcoming of the Red List global approach: the amassing of data from around the world that has taken countless hours to access and organize, and the resultant knowledge of differences among populations is overshadowed by the single status listing. The Red List is intended to provide a method for illuminating what species have greater extinction risk, which has clear relevance to conservation priority setting. Many countries refer to the IUCN global lists for constructing national Red Lists and protected species lists (Lamoreux et al., 2003; M. Marcovaldi, pers. comm.), while others simply use the Red List Criteria for developing these national lists (Amorocho, 2003; J. Alava, pers. comm.; Table 1). Miller et al. (2007) found that out of 180 countries examined, 77% had developed national lists of threatened species, and out of these 78% used a version of the IUCN criteria. Although application of the 2001 Criteria at national or local levels alleviates many of the inaccuracies associated with larger spatial scales; we believe that adopting the statuses verbatim from the Global Red List for national Red Listing can lead to incorrect and misleading results. More troubling is the potential for inaccurate status designations to result in misallocation of resources to populations or species that are not those with the greatest need of such attention.

Conservation groups and governments are likely to interpret species’ listings in higher threat categories as a need for greater conservation (Possingham et al., 2002). Let us first take a species that is listed at the highest threat categories of Endangered and Critically Endangered. There are populations among all marine turtle species that are classified as such that are stable and increasing, yet they may receive conservation attention simply because of their Red List status. While we certainly do not suggest that a species need be on the verge of collapse to warrant conservation attention, we do suggest that such misnomers create unique challenges for conservation priority setting and budgeting exercises. In the reciprocal case, let us consider a situation where the global mean results in a Near Threatened or even Vulnerable, despite the fact that one or more populations may be declining or truly threatened with local or regional extirpation. Should a Near Threatened or Vulnerable listing mean that the focal population does not require conservation action or management, or is of lower priority to recover? Certainly not, and it is conceivable that such disparities between Red Listings and reality may some day lead to localized extinctions. For example, whereas olive ridleys in the Atlantic constitute a small, isolated population susceptible to extirpation, the species is listed at a lower Red List category (Vulnerable, IUCN, 2007) than are hawksbills and leatherbacks, both of which are more widely distributed and under comparatively lesser threat of extirpation in the Atlantic (both listed as Critically Endangered, IUCN, 2007).

4. The MTSG Mandate: perspectives from within the membership

Although the MTSG does a variety of activities (Mast et al., 2006a), the fact that it was originally created by the IUCN (Davis, 2005) suggests that its primary mandate is to conduct global Red List assessments. And here lies the conundrum: should MTSG continue with its primary mandate, even though it has been increasingly apparent that global Red List assessments do not adequately reflect the differences in population statuses worldwide and thus are not directly useful for marine turtle conservation efforts? Or should this organization develop new approaches to marine turtle status assessments, irrespective of its acceptance by the IUCN?

To examine these issues, a questionnaire was circulated to the MTSG membership, asking opinions on a variety of questions relating to Red Listing, regional assessments, and the perceived best approaches for assessing marine turtles. There were 50 respondents representing 23 countries, and their responses were highly informative of the general sentiments among the MTSG membership (Fig. 6). Overall, 98% of the respondents identified themselves as being either somewhat or very familiar with the Red Listing process, and of these 62% believed that the MTSG invested the appropriate amount of time in conducting Red List assessments. Interestingly, the vast majority of respondents also felt it would be appropriate for these assessments to move beyond focusing only on extinction risk, and also include a thorough threat assessment along with conservation recommendations. Respondents were divided in their beliefs of whether the MTSG should focus on regional assessment efforts, and if so, just how to go about conducting them (Fig. 6). The majority agreed that the MTSG should only undertake regional assessments if they are included in the Red List, although only 10% believe that the MTSG should follow the 2001 Criteria for regional assessments. Although this apparent incongruity may result from respondents’ lack of understanding of the likelihood that non-use of the 2001

Criteria would likely scuttle any attempts to have sub-global listings on the Red List, it nevertheless underscores the need for MTSG and IUCN to discuss alternative approaches to the 2001 Criteria for assessing marine turtles.

Clearly, not all MTSG members are fluent in the IUCN Red List process, and the results of this survey should therefore be viewed cautiously. However, the MTSG is an IUCN-mandated organization and it is incumbent upon both the leadership and the membership of the MTSG to work together to ensure that each and every member is versed in the Red List Criteria so that any future deliberations on how to proceed with Red Listing and species status assessments in general can benefit from the input of a wider cross-section of MTSG members.

5. Improving marine turtle status assessments

The spatial and temporal problems inherent in the 2001 IUCN criteria are taxing on the biologists involved and distracting to the larger issue of conservation dependency for most marine turtle populations. Perhaps more than ever there is a need to address the apparent deficiencies in the current criteria and to more explicitly inform the List's constituents of its caveats and appropriate applications. In considering ways to make the process better, there are a variety of modifications that may prove beneficial. These include efforts to: *i*) make greater use of modelling approaches applicable to Criteria A and E, *ii*) adjust criteria to include other status metrics in addition to abundance and extinction risk, *iii*) more explicitly state recovery goals and establish mechanisms to monitor population trends relative to these goals, *iv*) conduct regional assessments; and *v*) make assessments more policy prescriptive by including a separate and explicitly headed section that describes threats and solutions. We acknowledge that greater use of Criteria A and E would only be necessary if the MTSG continues with its role of conducting global assessments within the IUCN framework; however, the remaining alternatives would ideally lead to more accurate assessments that are of greater use at local and regional scales. In addition to these approaches, a list of priority activities that we believe is necessary for improving assessments is provided in Table 3.

5.1. Broader use of 2001 IUCN Criteria

It is evident that for long-lived and widely distributed taxa such as marine turtles, a new assessment strategy is necessary. However, before looking outside of the current Red List system, it may be worth pursuing alternative approaches within the 2001

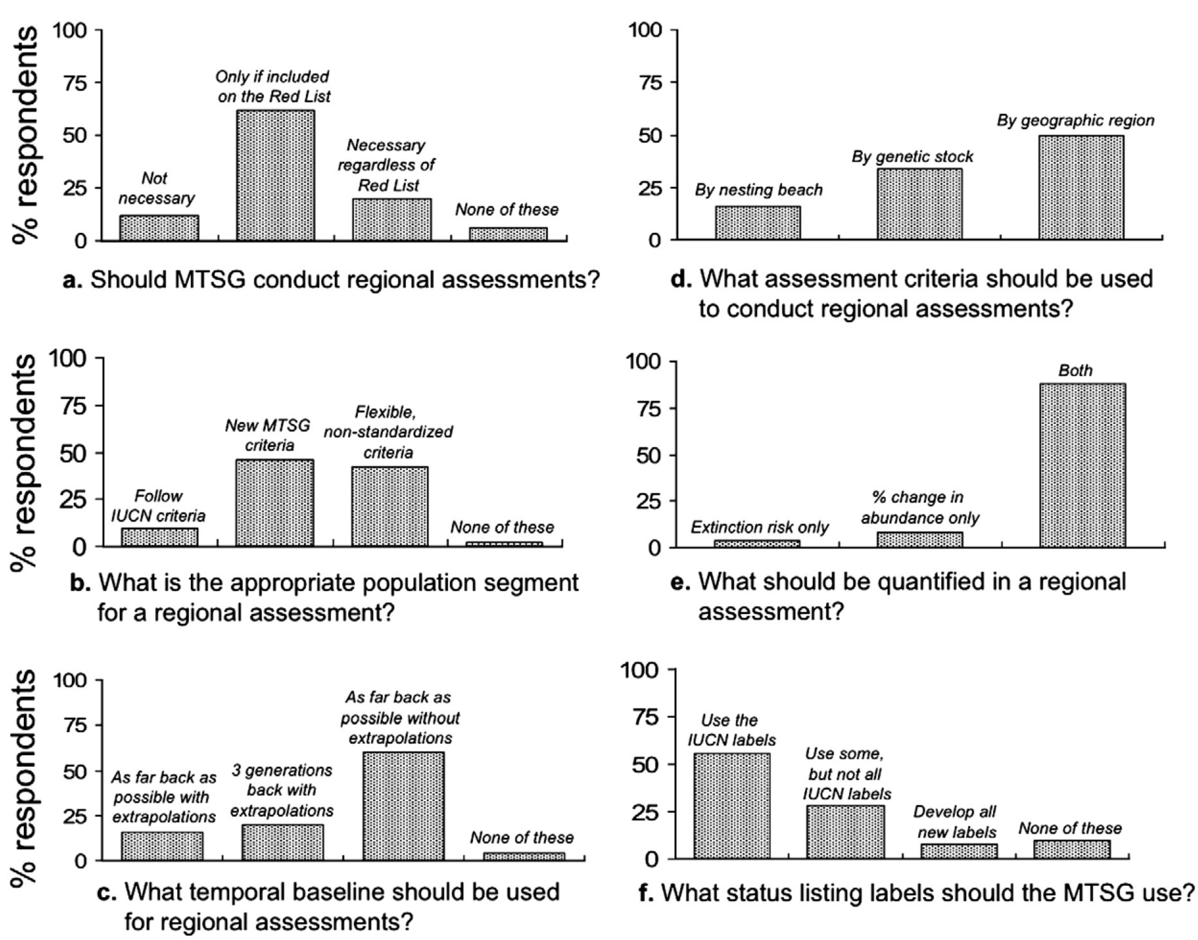

Fig. 6. Summary of responses to a Red List Questionnaire circulated to the MTSG membership. There were 50 respondents (18% of total membership) representing 23 countries.

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Table 3
Research and assessment priorities for improving marine turtle assessments

Research/assessment priorities
Assessment strategy development
• Establish a demography/modelling working group to evaluate the value of different population parameters for use as indices of population trends (e.g., adult size distributions, nesting frequency, rate of abundance change, in-water catch per unit effort, in-water size class distributions, fisheries landings, etc.).
• Establish genetics working group to determine the analytical approaches that are necessary for elucidating distinct population segments, with emphasis on distinguishing the roles of female and male-mediated gene flow.
Data collection
• Promote the standardization of data collection protocols through professional workshops and educational outreach.
• Conduct long-term monitoring of nesting and in-water populations, with emphasis on collecting the appropriate data necessary for population trend analyses.
• Conduct basic research on the biology of marine turtles at nesting beaches, foraging areas, and migratory corridors with emphasis on generating long-term time series data sets.
Data analysis
• Use appropriate genetic analyses to define boundaries between population units.
• Develop models for projecting future population status and extinction probability.
Data dissemination
• Disseminate data in peer-reviewed wide-access publications

These are organized into four groups based on the stage in an assessment process in which they are relevant. These priorities are not intended to be an exhaustive list, and instead represent the most urgent needs.

Criteria that can lead to more acceptable assessments. For example, instead of using a baseline that is 3-generations in the past, which results in data availability issues, it may be appropriate to use a more recent baseline and conduct population abundance projections into the future (Criterion A4) or conduct explicit modelling of future extinction risk of a population (Criterion E) (Appendix 2). There have been several efforts to model population dynamics of marine turtles, including deterministic models (Crouse et al., 1987; Heppell et al., 2005), stochastic simulation models (Chaloupka, 2002, 2004), Bayesian state-space models (Chaloupka and Balazs, 2007), and individual-based models (Mazaris et al., 2005). Such approaches may prove useful within a Red List framework, particularly those which examine extinction risk (Chaloupka, 2002, 2004; Mazaris et al., 2005). Further, although not yet applied widely to marine turtles, the MTSG should explore modelling efforts such as Population Viability Analysis (PVA; Akçakaya and Sjögren-Gulve, 2000; Morris et al., 2002). There are variable types of PVA that have been used to effectively project extinction risk in several non-marine turtle species based on demographic and threat input data (Beissinger and McCullough, 2002). Although these methods have drawbacks, particularly when demographic data are lacking (Taylor, 1995; Coulson et al., 2001), they make classification decisions less arbitrary and more grounded in scientific information by allowing explicit estimation of the likelihood that a population will persist for a particular time period (Wilcove et al., 1993; Akçakaya and Sjögren-Gulve, 2000; Gerber et al., 2007).

5.2. Modify existing criteria

Based on the results of the MTSG Red Listing questionnaire, it is apparent that the membership sees a need to either develop entirely new rule-based criteria, or maintain flexible, non-standardized criteria for conducting marine turtle assessments (Fig. 6). Although the remit of this paper does not include prescribing the criteria that should be followed — determining the appropriate criteria will require careful consideration among members of the MTSG — there are a number of components that have emerged from previous discussions that appear necessary to address, including the needs to: *i*) target non-reproductive cohorts in assessments, *ii*) include demographic trends other than abundance into assessments, *iii*) incorporate genetic information from populations under assessment, and *iv*) establish reference markers so that recovery can be identified. In the process it would be useful to examine how other groups have dealt with assessment approaches outside of the IUCN Red List framework (Ocean Wildlife Campaign, 1997; Gerber and Demaster, 1999; Musick, 1999; Sheldon et al., 2001; Andelman et al., 2004), perhaps adopting elements of these alternative approaches for future marine turtle assessments.

Monitoring in-water stocks is of the utmost importance for crosschecking trends observed on nesting beaches (Bjorndal et al., 2005a). For example, Chaloupka and Limpus (2001) reported an increase in annual nesting abundance of ~3% per year from 1974–1998 at Heron Island in the southern Great Barrier Reef (sGBR) and corroborated this increase with data showing an increase of 8% per year from 1985–1992 for the sGBR green turtle population. While such examples are few, and in-water work logistically challenging, we nonetheless must promote the research of non-nesting cohorts so as to reduce uncertainty in our assessments. These efforts may be a combination of extensive and intensive aerial and in-water surveys at selected monitoring sites that represent the range of habitats, cohorts, and turtle densities (Bjorndal et al., 2005a). It is also of great importance that such efforts are undertaken long enough so as to produce time series data that are comparable to information from nesting beaches.

Demographic trends other than absolute abundance can also be very informative for population assessments. Aspects that may prove useful for such exercises include examinations of rates of change in population size (S. Heppell, pers. comm.), stage-based survival rates (Crouse et al., 1987; Seminoff et al., 2003; Campbell and Lagueux, 2005; Chaloupka and Limpus, 2005; Troëng and Chaloupka, 2007), stage-based recruitment rates (Chaloupka and Musick, 1997; Chaloupka and Limpus, 2001), nesting remigration intervals (Troëng and Chaloupka, 2007), and changes in mean body size of nesters and annual proportion of neophyte nesters (Hatase et al., 2002; Limpus et al., 2003). The increased focus on immature cohorts and the use of metrics other than annual nesting abundance will ideally be effective for providing early warning signs that help conservation efforts avoid the aforementioned ‘population time-bomb’ dilemma (Fig. 1).

Maintaining genetic diversity is very important and even small populations can have large value for preserving the resilience of a species (Bowen and Karl, 1997). It is therefore important that assessments integrate stock-specific genetic information into a weighting system for determining the relative importance of each population to overall species genetic diversity. Criteria aimed at determining effective population size and conserving genetic diversity have been developed for fish (Reiman and Allendorf, 2001) and could also be adopted for marine turtles. A likely outcome of such efforts will be the clear importance of the need to carefully manage smaller rookeries as well as the primary rookeries within each region so as to conserve genetic diversity (Bjorndal et al., 2005b).

Considering that marine turtle assessments and their status recommendations commonly dictate conservation efforts, it is important that a system be developed by which we can measure the success of these efforts. With respect to population size, we suggest that explicit population size goals be developed, and that efforts are enhanced to compare how ongoing recovery, or decline, compares to these goals. Establishing these benchmarks may also be useful for determining when and if a change in listing status is warranted. Knowledge on former abundance can be gained using both reconstructions of past population size (Broderick et al., 2006; Bjorndal and Jackson, 2003; Chaloupka and Balazs, 2007) as well as genetic approaches such as determining the ratio of current population size to effective population size (Rivalan et al., 2006), or the development of coalescent models for mitochondrial DNA sequence variation for estimating genetic diversity and historic population sizes (Roman and Palumbi, 2003; Baker and Clapham, 2004). Although the latter has been a fairly controversial approach, there are lessons to be learned from this effort that can be applied to marine turtles.

5.3. Regional assessments

Calls for a regional approach to marine turtle status assessments have been ongoing for several years (e.g. Mrosovsky 2003, Seminoff 2004b). For a taxon such as marine turtles, that experience varied anthropogenic pressures in different parts of the world, this is requisite for effective management. By identifying nesting populations that are declining as well as highlighting those that are doing relatively well, finer-scale assessments will be more useful for conservationists and resource managers on-the-ground.

A regional approach would clearly benefit marine turtle status assessments and conservation efforts, but there are a number of important realities to keep in mind. First, regional assessments will demand additional time from volunteer assessors and may result in unwanted delays in the production of these assessments. Second, regional assessments will likely result in the down-listing or delisting of some marine turtle populations. The Hawaiian green turtle stock, which has recovered since reaching a population low point in the early 1970s, provides a perfect example (Balazs and Chaloupka, 2004; Chaloupka and Balazs, 2007). If and when an assessment for this population is undertaken it is likely, and warranted, that the status will be changed from its current Endangered listing. While downlisting may seem counterintuitive to the precautionary nature of marine turtle conservation, it is an important step forward if status listings are to be useful in the development of conservation priorities. When scientifically justified, downlisting and/or delisting also underscore the rooting of assessments on science rather than political or fund-raising agendas. Third, changing conservation status listings could result in a rearrangement of our conservation priorities. By revealing those populations that are doing poorly, regional assessments may shift emphasis from those that are doing relatively well, even if they too are depleted. Fourth, regional assessments and the resultant status changes may have profound impacts on issues of international trade of marine turtle products. Although trade itself may not be bad (the IUCN has a mandate to explore Sustainable Use), unregulated take and trade may lead to further declines, and this is a genuine concern. While in theory such trade would be heavily monitored and managed via quotas within participating countries, the issues surrounding potential trade possibilities must be carefully examined by all stakeholders. However, this is not the remit of Red Listing, and instead needs to be addressed by instruments that control international trade such as the Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES), rather than a biological assessment of population change.

Although the IUCN has prescribed ways to apply the Red List Criteria for regional assessments (Gärdenfors et al., 2001, IUCN, 2003), which have been used with a variety of taxa including marine turtles (Amorocho, 2003; Eaton et al., 2005; De Iongh and Bal, 2007), regional assessments per se are not eligible for inclusion on the IUCN Red List. Instead, the only sub-global listings that are acceptable for inclusion on the Red List are ‘subpopulation’ assessments. Subpopulations are defined by the IUCN as “geographically or otherwise distinct groups in a global population between which there is little demographic or genetic exchange (typically one successful migrant individual or gamete per year or less)” (IUCN, 2001). Indeed, the Cetacean Specialist Group is light years ahead of the MTSG in this regard, as they have used genetic data to delineate stock boundaries for a variety of species and conducted subpopulation assessments on several of these (e.g., grey whale, *Eschrichtius robustus*, harbor porpoise, *Phocoena phocoena*)(IUCN, 2007). For marine turtles; however, the stock structure patterns are less resolved and genetic isolation may be unlikely for the vast majority of populations because the global criss-crossing undertaken by individuals and male-mediated gene flow via copulation in mixed-stock foraging areas typically maintain high gene flow among populations (Bowen and Karl, 1997).

Despite the emphasis on genetic thresholds, there is no consensus on the relative importance of nuclear DNA or mitochondrial DNA research in Red List categorization of marine turtles (Naro-Maciel and Formia, 2006; Mrosovsky, 2006). Addressing this issue is of paramount importance for determining whether the MTSG should pursue IUCN subpopulation listings or maintain focus on regional listings outside the Red List framework. We suggest that exclusion from the Red List should not preclude the MTSG's efforts to undertake regional assessments, but it would be worth exploring ways to integrate a genetic approach in such assessments that would result in the full global complement of regional assessments of a species being packaged together by the MTSG and accepted as the Red List global assessment for that species. This would be mutually beneficial for both the MTSG and IUCN as it would allow the development of meaningful regional assessments while at the same time meeting the need to produce a global assessment.

5.4. Conservation assessments

While still operating on a regional scale, we suggest there is a great need for an assessment system that evaluates the success and failure of conservation and policy strategies, and advocates appropriate changes if necessary. While many species may be in need of conservation, not all species in a system may need to be assessed and habitat indices and other indicators may be used to deduce trends for large numbers of taxa. There have been a variety of indicators of biodiversity, including some that are direct measures of biodiversity, such as habitat indices, population indices and trophic level indices (Brooks and Kennedy, 2004; Mace, 2005). Here we present a new framework for conservation assessments for species that may require direct conservation or management intervention. This approach can be applied to a wide range of taxa across different habitats, wherever the species can be individually monitored. It consists of an assessment of risk of current threats which have a negative impact on the population (low, medium or high), and the consequent recommended conservation action (Fig. 7). This is not unlike the use of population reference points in fisheries management (Kell et al., 2007), although we recognize that this approach has had varying levels of success in fisheries (e.g., Piet and Rice, 2004; Rice and Legace, 2007). When the species is first assessed, it may be assigned on the basis of existing knowledge to one of three categories (i.e. low, medium or high risk from current threats; Fig. 7). Medium and high risk species should, after the first and subsequent assessments, be monitored on a regular basis. Monitoring should examine explicitly two parameters — the population or index thereof and the efficacy of conservation efforts. Thus, when the species or population is reassessed after 1 or 5 years, there may be a change in status, and closely linked to this, a change in conservation action. If, for example, the status of a species does not change despite strong conservation efforts, then a change in strategy should be recommended.

Following the earlier discussion of criteria (see Section 5.2), we will not enter into a debate of which quantitative criteria should be used to track population trends. The important point is to monitor both the population and the impact of the management action. Admittedly, regardless of the criteria used, the precautionary principle will be invoked to place species in medium and high risk categories. However, taking a quasi-Bayesian approach, we suggest that the status will be self-correcting, and the original status designation will not matter greatly since a population will be regularly monitored and future status assignments will depend on contemporary data. Going further, tools such as Bayesian Belief Networks can be used to derive conservation decisions from empirical data (Chaloupka, 2007). Bayesian networks are graphical models that represent variables and their probabilistic interdependencies. Their utility in this context is the ability to incorporate uncertainty, which derives from a number of sources, including the nature of the system, the accuracy of the data, and the relationship between variables. Such uncertainty is characteristic of natural systems, and especially so in the data used for conservation assessments.

It would also be useful to describe, as a consequence of these local assessments, the status, threats and conservation actions at different scales, namely nesting beaches, genetic stocks, ocean basins, and species. There are both different threats as well as different conservation actions at different scales, and such a matrix can provide guidance to conservation practitioners who function at these different scales (Harcourt and Parks, 2003). A caveat is that there will be contention over what constitutes a region or population for these assessments (see Section 5.5 below). Increasingly sophisticated molecular genetic tools and comprehensive datasets will no doubt assist in addressing this difficult issue, with the caveats mentioned earlier. There are several reasons that assessing conservation action is at least as critical as assessing population trend. The first obvious one is that the trend itself provides no guidance to action. A second important and often ignored point is that many conservation actions come with significant social and economic costs (Dowie, 2005; Brockington and Igoe, 2006). These costs can have negative impacts for conservation itself in the long term. Therefore, what are broadly considered to be ‘good for conservation’ (e.g., pristine habitats, exclusion of people and activity) may have negative social and political consequences that would impact long-term success. Therefore, while it may be important to institute such actions for population recovery in the short term, the long-term success of recovery efforts may benefit from modifications in such conservation practices to lessen their negative social impact.

5.5. Species concepts and the implications of lumping and splitting

Notable amongst the challenges of biodiversity assessment is the issue of the definition of species. Though ‘species’ form the currency of biology and conservation, there is little agreement on what constitutes a species between people who work on different taxa, and sometimes even on the same taxa, leading to problems in biodiversity studies, assessment of trends and consequently, conservation (Isaac et al., 2004, Agapow et al., 2004; Mace 2004). For example, when a species is split into two, not only does the diversity of the taxon increase, but the range and abundance of each declines, thus making it more likely to be classified in one of the IUCN Red List ‘Threatened’ categories. Sea turtles have not been without debate on this particular issue, with considerable controversy surrounding the species status of the east Pacific green turtle or black turtle (Karl and Bowen, 1999; Pritchard 1999). In fact, it was argued that conferring species status on this population would help in its conservation. In this context as well, one can question the ethics of mixing science and advocacy (Shrader-Frechette and McCoy, 1999).

The same problem arises when dividing sea turtles into regional or local populations for assessment. While this makes the application of IUCN criteria more practical, the basis for

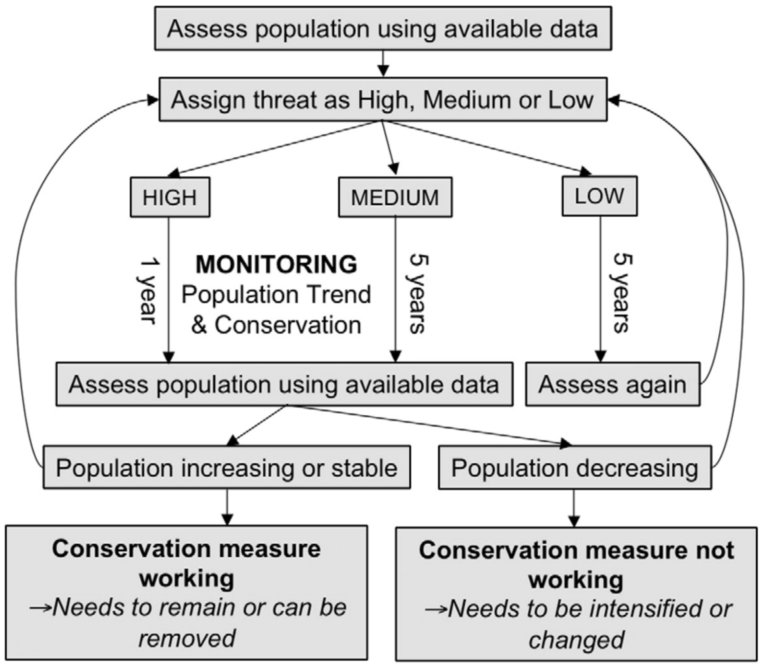

Fig. 7. A proposed conservation assessment framework for focal or flagship species such as marine turtles.

separation remains as intractable as ever. Genetic data, while providing some resolution, offer no more insights into where the line should be drawn (i.e., what degree of difference?) for populations than they do for species. One suspects that these decisions will be mired in similar arguments over 'splitting' and 'lumping', the former being driven by the fact that splitting results in higher threat categorization for the population concerned, and therefore greater conservation action. This leads back to the issue that the science of assessment is being influenced by the advocacy of conservation.

6. Conclusion

Marine turtle species are not single entities unto themselves, but instead comprised of a mosaic of individuals and populations, all of which may live in slightly different habitats, experience varied anthropogenic threats, conservation efforts, and abundance trajectories. While there may be no perfect assessment system for capturing these differences, the current Red Listing criteria have resulted in biological assessments of marine turtles that are inaccurate and misleading for conservation priority setting. The lack of spatial resolution has resulted in major discrepancies between Red List status and true extinction risk, the result of which has been skepticism on the part of scientists and conservation practitioners, which may undermine the intended utility of the Red List. A new approach to assessing marine turtles is clearly needed and there are several areas of research that will facilitate improvements over existing assessment methods (Table 3). Application of novel criteria, and development of regional assessments and/or conservation assessments are the most plausible alternatives, and would serve assessments well by establishing more precise metrics for measuring abundance change while at the same time resulting in assessments that are more useful for local and regional conservation (Fig. 7).

Constructing the best approaches will require thoughtful deliberations by a wide cross-section of biologists, telemetry specialists, geneticists, and demographic modellers. In the meantime, marine turtle biologists and experts must weigh the value of proceeding with global assessments that employ Criterion A of the 2001 IUCN criteria. While continuing these efforts would garner the aforementioned benefits of fulfilling the IUCN mandate, highlighting knowledge gaps, and amassing data in one source, we argue that continuing along this path is unwarranted for a group such as the MTSG, that should have scientific rigor among its core values. A more prudent approach would be to suspend species assessments until a new approach is developed, and ideally blessed by the IUCN.

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Appendix 1. IUCN Red List assessment categories (IUCN, 2001)

Red List category	Definition
Extinct (EX)	A taxon is Extinct when there is no reasonable doubt that the last individual has died.
Extinct in the wild (EW)	A taxon is Extinct in the Wild when it is known only to survive in cultivation, in captivity or as a naturalized population (or populations) well outside the past range
Critically Endangered (CR)	A taxon is Critically Endangered when the best available evidence indicates that it meets any of the criteria A to E for Critically Endangered (see Section 5), and it is therefore considered to be facing an extremely high risk of extinction in the wild.
Endangered (EN)	A taxon is Endangered when the best available evidence indicates that it meets any of the criteria A to E for Endangered (see Section 5), and it is therefore considered to be facing a very high risk of extinction in the wild.
Vulnerable (VU)	A taxon is Vulnerable when the best available evidence indicates that it meets any of the criteria A to E for Vulnerable (see Section 5), and it is therefore considered to be facing a high risk of extinction in the wild.
Near Threatened (NT)	A taxon is Near Threatened when it has been evaluated against the criteria but does not qualify for Critically Endangered, Endangered or Vulnerable now, but is close to qualifying for or is likely to qualify for a threatened category in the near future.
Least Concern (LC)	A taxon is Least Concern when it has been evaluated against the criteria and does not qualify for Critically Endangered, Endangered, Vulnerable or Near Threatened.
Data Deficient (DD)	A taxon is Data Deficient when there is inadequate information to make a direct, or indirect, assessment of its risk of extinction based on its distribution and/or population status.
Not Evaluated (NE)	A taxon is Not Evaluated when it is has not yet been evaluated against the criteria

Appendix 2. Summary of the IUCN 2001 Criteria (IUCN, 2001)

A. Reduction in population size based on any of the following:

1. An observed, estimated, inferred or suspected population size reduction of ≥90% (CR) or ≥70% (EN) or ≥50% (VU) over the last 10 years or three generations, whichever is the longer, where the causes of the reduction are clearly reversible AND understood AND ceased, based on (and specifying) any of the following:
   - (a) direct observation
   - (b) an index of abundance
   - (c) a decline in area of occupancy, extent of occurrence and/or quality of habitat
   - (d) actual or potential levels of exploitation
   - (e) the effects of introduced taxa, hybridization, pathogens, pollutants, competitors or parasites.
2. An observed, estimated, inferred or suspected population size reduction of ≥80% (CR) or ≥50% (EN) or ≥30% (VU) over the last 10 years or three generations, whichever is the longer, where the reduction or its causes may not have ceased OR may not be understood OR may not be reversible, based on (and specifying) any of (a) to (e) under Appendix 1.
3. A population size reduction of ≥80% (CR) or ≥50% (EN) or ≥30% (VU) projected or suspected to be met within the next 10 years or three generations, whichever is the longer (up to a maximum of 100 years), based on (and specifying) any of (b) to (e) under Appendix 1.
4. An observed, estimated, inferred, projected or suspected population size reduction of ≥80% (CR) or ≥50% (EN) or ≥30% (VU) over any 10 year or three generation period, whichever is longer (up to a maximum of 100 years in the future), where the time period must include both the past and the future, and where the reduction or its causes may not have ceased OR may not be understood OR may not be reversible, based on (and specifying) any of (a) to (e) under Appendix 1.

B. Geographic range in the form of either B1 (extent of occurrence) OR B2 (area of occupancy) OR both:

1. Extent of occurrence estimated to be less than 100 km^2 (CR) or 1000 km^2 (EN) or 10,000 km^2 (VU), and estimates indicating at least two of a–c:
   - a. Severely fragmented or known to exist at only a single location.
   - b. Continuing decline, observed, inferred or projected, in any of the following: (i) extent of occurrence (ii) area of occupancy (iii) area, extent and/or quality of habitat (iv) number of locations or subpopulations (v) number of mature individuals.
   - c. Extreme fluctuations in any of the following: (i) extent of occurrence (ii) area of occupancy (iii) number of locations or subpopulations (iv) number of mature individuals.
2. Area of occupancy estimated to be less than 100 km^2 (CR) or 1000 km^2 (EN) or 10,000 km^2 (VU), and estimates indicating at least two of a–c:
   - a. Severely fragmented or known to exist at only a single location.
   - b. Continuing decline, observed, inferred or projected, in any of the following: (i) extent of occurrence (ii) area of occupancy (iii) area, extent and/or quality of habitat (iv) number of locations or subpopulations (v) number of mature individuals.
   - c. Extreme fluctuations in any of the following: (i) extent of occurrence (ii) area of occupancy (iii) number of locations or subpopulations (iv) number of mature individuals.

C. Population size estimated to number fewer than 250 mature individuals and either:

1. An estimated continuing decline of at least 25% within three years or one generation, whichever is longer, (up to a maximum of 100 years in the future) OR
2. A continuing decline, observed, projected, or inferred, in numbers of mature individuals AND at least one of the following (a–c):
   - (a) no subpopulation estimated to contain more than 50 mature individuals, OR
   - (b) at least 90% of mature individuals in one subpopulation.
   - (c) Extreme fluctuations in number of mature individuals.

D. Population size estimated to number fewer than 50 mature individuals

E. Quantitative analysis showing the probability of extinction in the wild is at least 50% within 10 years or three generations, whichever is the longer (up to a maximum of 100 years)

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References

Abreu-Grobois, A., Plotkin, P., 2007. MSTG global assessment of olive ridley turtles for the IUCN Red List. Submitted to IUCN Species Survival Commission, April 2007. http://www.iucn-mtsg.org/red_list/lo/RLA_Lepidochelys_olivacea_revised.pdf.

Akçakaya, H.R., Sjögren-Gulve, P., 2000. Population viability analysis in conservation planning: an overview. Ecol. Bull. 48, 9–21.

Amorocho, D., 2003. Monitoring nesting loggerhead turtles (*Caretta caretta*) in the central Caribbean coast of Colombia. Mar. Turt. Newsl. 101, 8–13.

Andelman, S.J., Groves, C., Regan, H.M., 2004. A review of protocols for selecting species at risk in the context of US Forest Service viability assessments. Acta Oecol. 26, 75–83.

Agapow, P.-M., Bininda-Edmonds, O.R.P., Crandall, K.A., Gittleman, J.L., Mace, G.M., Marshall, J.C., Purvis, A., 2004. The impact of species concept on biodiversity studies. Q. Rev. Biol. 79, 161–179.

Baker, C.S., Clapham, P.J., 2004. Modelling the past and future of whales and whaling. Trends Ecol. Evol. 19, 365–371.

Baker, J.D., Littnan, C.L., Johnston, D.W., 2006. Potential effects of sea level rise on the terrestrial habitats of endangered and endemic megafauna in the Northwestern Hawaiian Islands. Endanger. Species Res. 2, 21–30.

Balazs, G.H., Chaloupka, M.Y., 2004. Thirty-year recovery trend in the once depleted Hawaiian green sea turtle stock. Biol. Conserv. 117, 491–498.

Beggs, J.A., Horrocks, J.A., Krueger, B.H., 2007. Increase in hawksbill sea turtle *Eretmochelys imbricata* nesting in Barbados, West Indies Endang. Spec. Res. 3, 159–168.

Beissinger, S.R., McCullough, D.R. (Eds.), 2002. Population Viability Analysis. University of Chicago Press, Chicago. 593 pp.

Bjorndal, K.A., Jackson, J.B.C., 2003. Roles of sea turtles in marine ecosystems: reconstructing the past. In: Lutz, P.L., Musick, J.A., Wyneken, J. (Eds.), The Biology of Sea Turtles, vol. 2. CRC Press, Boca Raton, Florida, pp. 259–274.

Bjorndal, K.A., Bolten, A.B., Chaloupka, M.Y., 2005a. Evaluating trends in abundance of immature green turtles, *Chelonia mydas*, in the greater Caribbean. Ecol. Appl. 15, 304–314.

Bjorndal, K.A., Bolten, A.B., Troëng, S., 2005b. Population structure and genetic diversity in green turtles nesting at Tortuguero, Costa Rica, based on mitochondrial DNA control region sequences. Mar. Biol. 147, 1449–1457.

Blake, S., Hedges, S., 2004. Sinking the flag-ship: the case of forest elephants in Asia and Africa. Conserv. Biol. 18, 1180–1190.

Bowen, B.W., Karl, S.A., 1997. Population genetics, phylogeny, and molecular evolution. In: Lutz, P.L., Musick, J.A. (Eds.), The Biology of Sea Turtles. CRC Press, Boca Raton, pp. 29–50.

Brockington, D., Igoe, J., 2006. Eviction for conservation: a global overview. Conserv. Soc. 4, 424–470.

Broderick, A.C., Godley, B.J., Hays, G.C., 2001. Trophic status drives interannual variability in nesting numbers of marine turtles. Proc. R. Soc. 268, 1481–1487.

Broderick, A.C., Frauenstein, R., Glen, F., Hays, G.C., Jackson, A.L., Pelembe, T., Ruxton, G.D., Godley, B.J., 2006. Are green turtles globally endangered? Glob. Ecol. Biogeogr. 15, 21–26.

Brooks, T., Kennedy, E., 2004. Biodiversity barometers. Nature 431, 1046–1047.

Campbell, L.M., 2002. Science and sustainable use: views of marine turtle conservation experts. Ecol. Appl. 12, 1229–1246.

Campbell, C.L., Lagueux, C.J., 2005. Survival probability estimates for large juvenile and adult green turtles (*Chelonia mydas*) exposed to an artisanal marine turtle fishery in the western Caribbean. Herpetologica 61, 91–103.

Chacon-Chaveri, D., 2002. Synopsis of the leatherback turtle (*Dermochelys coriacea*). Inter-american convention for the protection and conservation of sea turtles. Document: INF-16-04. http://www.iacseaturtle.org/iacseaturtle/English/download/INF-16-04%20eng.

Chacon-Chaveri, D., Eckert, K.L., 2007. Leatherback sea turtle nesting at Gandoca Beach in Caribbean Costa Rica: management recommendations from fifteen years of conservation. Chelonian Conserv. Biol. 6, 101–110.

Chaloupka, M., 2002. Stochastic simulation modelling of southern Great Barrier Reef green turtle population dynamics. Ecol. Model. 148, 79–109.

Chaloupka, M., 2004. Exploring the metapopulation dynamics of the southern Great Barrier Reef green turtle stock and possible consequences of sex-biased local harvesting. In: Akçakaya, H., Burgman, M., Kindvall, O., Wood, C., Sjogren-Gulve, P., Hattfield, J., McCarthy, M. (Eds.), Species Conservation and Management: Case Studies. Oxford University Press, New York, pp. 340–354.

Chaloupka, M., 2007. Using Bayesian belief networks to evaluate the relative risk of fishing gear types on sea turtle stocks. SEAFDEC-FAO Workshop on assessing the relative importance of sea turtle mortality due to fisheries in Southeast Asia 19–23 March 2007, Bangkok, Thailand.

Chaloupka, M.Y., Musick, J.A., 1997. Age, growth, and population dynamics. In: Lutz, P.L., Musick, J.A. (Eds.), The Biology of Sea Turtles. CRC Press, Boca Raton, Florida, pp. 233–273.

Chaloupka, M.Y., Limpus, C.J., 2001. Trends in the abundance of sea turtles resident in southern Great Barrier Reef waters. Biol. Conserv. 102, 235–249.

Chaloupka, M., Limpus, C., 2005. Estimates of sex- and age-class-specific survival probabilities for a southern Great Barrier Reef green sea turtle population. Mar. Biol. 46, 1251–1261.

Chaloupka, M., Balazs, G., 2007. Using Bayesian state-space modelling to assess the recovery and harvest potential of the Hawaiian green sea turtle stock. Ecol. Model. 205, 93–109.

Chaloupka, M., Bjorndal, K.A., Balazs, G., Bolten, A.B., Ehrhart, L.M., Limpus, C., Suganuma, H., Troëng, S., Yamaguchi, M., 2007. Encouraging outlook for recovery of a once-severely-exploited marine megaherbivore and restoration of its ecological function. Global Ecol. Biogeogr. doi:10.1111/j.1466-8238.2007.00367 (published online).

Chan, E.-H., 2006. Marine turtles in Malaysia: on the verge of extinction? Aquat. Ecosys. Health Manag. 9, 175–184.

Chan, E.-H., Liew, H.-C., 1996. Decline of the leatherback population in Terengganu, Malaysia, 1956–1995. Chelonian Conserv. Biol. 2, 196–203.

Coulson, T., Mace, G.M., Hudson, E., Possingham, H., 2001. The use and abuse of population viability analysis. Trends Ecol. Evol. 16, 219–221.

Courchamp, F., Clutton-Brock, T., Grenfell, B., 1999. Inverse density dependence and the Allee effect. Trends Ecol. Evol. 14, 405–410.

Crouse, D.T., Crowder, L.B., Caswell, H., 1987. A stage-based population model for loggerhead turtles and implications for their conservation. Ecology 68, 1412–1423.

Davis, F.R., 2005. Saving sea turtles; the evolution of the IUCN Marine Turtle Group. Endeavor 29, 114–119.

De Iongh, H.H., Bal, D., 2007. Harmonization of Red Lists in Europe: some lessons learned the Netherlands when applying the new IUCN Red List Cateories and Criteria version 3.1. Endanger. Species Res. 3, 53–60.

Diez, C.E., van Dam, R.P., 2002. Habitat effect on hawksbill turtle growth rates on feeding grounds at Mona and Monito Islands, Puerto Rico. Mar. Ecol. Prog. Ser. 234, 301–309.

Dowie, M., 2005. Conservation refugees. When protecting nature means kicking people out. Orion Nov/Dec.

Dulvy, N.K., Ellis, J.R., Goodwin, N.B., Grant, A., Reynolds, J.D., Jennings, S., 2004. Methods of assessing extinction risk in marine fish. Fish Fish. 5, 255–276.

Dutton, D.L., Dutton, P.H., Chaloupka, M., Boulon, R.H., 2005. Increase of a Caribbean leatherback turtle *Dermochelys coriacea* nesting population linked to long-term nest protection. Biol. Conserv. 126, 186–194.

Eaton, M.A., Gregory, R.D., Noble, D.G., Robinson, J.Z., Hughes, J., Procter, D., Brown, A.F., Gibbons, D.W., 2005. Regional IUCN Red Listing: the process as applied to birds in the United Kingdom. Conserv. Biol. 19, 1557–1570.

FAO, 2004. Papers presented at the expert consultation on interactions between sea turtles and fisheries within an ecosystem context. Rome, 9–12 March 2004. FAO Fisheries Report. No. 738, Suppl. Rome. 238 pp.

Fish, M.R., Cote, I.M., Gill, J.A., Jones, A.P., Renshoff, S., Watkinson, A.R., 2005. Predicting the impact of sea-level rise on Caribbean sea turtle nesting habitat. Conserv. Biol. 19, 482–491.

Fleming, E.H., 2001. Swimming Against the Tide: Recent Surveys of Exploitation, Trade, and Management of Marine Turtles in the Northern Caribbean. Traffic North America, Washington D. C. 161 pp.

Gärdenfors, U., Hilton-Taylor, C., Mace, G.M., Rodríguez, J.P., 2001. The application of IUCN Red List Criteria at regional levels. Conserv. Biol. 15, 1206–1213.

Gerber, L.R., Demaster, D.P., 1999. A quantitative approach to Endangered Species Act classification of long-lived vertebrates: application to the North Pacific humpback whale. Conserv. Biol. 13, 1203–1214.

Gerber, L.R., Keller, A.C., DeMaster, D.P., 2007. Ten thousand and increasing: is the western Arctic population of bowhead whale endangered? Biol. Conserv. 137, 577–583.

Girondot, M., Godfrey, M.H., Ponge, L., Rivalan, P., 2007. Modeling approaches to quantify leatherback nesting trends in French Guiana and Suriname. Chelonian Conserv. Biol. 6, 37–46.

Godley, B.J., Lima, E.H.S.M., Akesson, S., Broderick, A.C., Glen, F., Godfrey, M.H., Luschi, P., Hays, G.C., 2003. Movement patterns of green turtles in Brazilian coastal waters described by satellite tracking and flipper tagging. Mar. Ecol. Prog. Ser. 253, 279–288.

Groombridge, B., Luxmoore, R., 1989. The green turtle and hawksbill (Reptilia: Cheloniidae): world status, exploitation and trade. Secretariat of the Convention on International Trade in Endangered Species of Wild Fauna and Flora, Lausanne, Switzerland. 601 pages.

Hamazaki, T., Tanno, D., 2001. Approval of whaling and whaling-related beliefs: public opinion in whaling and non-whaling countries. Hum. Dimens. Wildl. 6, 131–144.

Harcourt, A.H., Parks, S.A., 2003. Threatened primates experience high human densities: adding an index of threat to the IUCN Red List Criteria. Biol. Conserv. 109, 137–149.

Hatase, H., Goto, K., Sato, K., Bando, T., Matsuzawa, Y., Sakamoto, W., 2002. Using annual body size fluctuations to explore potential causes for the decline in a nesting population of the loggerhead turtle *Caretta caretta* at Senri Beach, Japan. Mar. Ecol. Prog. Ser. 245, 299–304.

Hays, G.C., 2000. The implications of variable remigration intervals for the assessment of population size in marine turtles. J. Theor. Biol. 206, 221–227.

Hays, G.C., 2004. Good news for sea turtles. Trends Ecol. Evol. 19, 349–351.

Heppell, S.S., 1998. Application of life-history theory and population model analysis to turtle conservation. Copeia 1998, 367–375.

Heppell, S.S., Caswell, H., Crowder, L.B., 2000. Life histories and elasticity patterns: perturbation analysis for species with minimal demographic data. Ecology 81, 654–665.

Heppell, S.S., Crouse, D.T., Crowder, L.B., Epperly, S.P., Gabriel, W., Henwood, T., Marquez, R., Thompson, N.B., 2005. A population model to estimate recovery time, population size, and management impacts on Kemp's ridley sea turtles. Chelonian Conserv. Biol. 4, 767–773.

Hoekert, W.E.J., Neuféglise, H., Schouten, A.D., Menken, S.B.J., 2002. Multiple paternity and female-biased mutation at a microsatellite locus in the olive ridley sea turtle (*Lepidochelys olivacea*). Heredity 89, 107–113.

Holmes, E.E., 2001. Estimating risks in declining populations with poor data. Proc. Nat. Acad. Sci. 98, 5072–5077.

Horikoshi, K., Suganuma, H., Tachikawa, H., Sato, F., Yamaguchi, M., 1994. Decline of Ogasawara green turtle population in Japan. In: Bjorndal, K.A., Bolten, A.B., Johnson, D.A., Eliazar, P.J. (Eds.), Proceedings of the Fourteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-351, pp. 235–237.

Hornell, J., 1927. The Turtle Fisheries of the Seychelles Islands. H.M. Stationery Office, London. 55 pp.

Hughes, G.R., 1996. Nesting of the leatherback turtle (*Dermochelys coriacea*) in Tongaland, KwaZulu-Natal, South Africa, 1963–1995. Chelonian Conserv. Biol. 2, 153–158.

Isaac, N.J.B., Mallet, J., Mace, G.M., 2004. Taxonomic inflation: its influence on macroecology and conservation. Trends Ecol. Evol. 19, 464–469.

IUCN, 1963. Animals and Plants Threatened with Extinction I. IUCN Survival Service Commission, Switzerland.

IUCN, 2001. Guidelines for Assessing Taxa with Widely Distributed or Multiple Populations against Criterion A. Developed by the Standards and Petitions Subcommittee. June 2001.

IUCN, 2003. Guidelines for Application of IUCN Red List Criteria at Regional Levels: Version 3.0. IUCN Species Survival Commission. IUCN Gland Switzerland and Cambridge, UK. 26 pp.

IUCN, 2007. Red list of threatened species. <http://www.iucnredlist.org>.

Karl, S.A., Bowen, B.W., 1999. Evolutionary significant units versus geopolitical taxonomy: molecular systematics of an endangered sea turtle (genus *Chelonia*). Conserv. Biol. 13, 990–999.

Kell, L.T., Mosqueira, I., Grosjean, P., Fromentin, J.-M., Garcia, D., Hillary, R., Jardim, E., Mardle, S., Pastoors, M.A., Poos, J.J., Scott, F., Scott, R.D., 2007. FLR: an open-source framework for the evaluation and development of management strategies. ICES J. Mar. Sci. 64, 640–646.

Lamoreux, J., Akcakaya, H.R., Bennun, L., Collar, N.J., Boitani, L., Brackett, D., Brautigam, A., Brooks, T.M., de Fonseca, G.A.B., Mittermeier, R.A., Rylands, A.B., Gärdenfors, U., Hilton-Taylor, C., Mace, G., Stein, B.A., Stuart, S., 2003. Value of the IUCN Red List. Trends Ecol. Evol. 18, 214–215.

Lee, P.L.M., Hays, G.C., 2004. Polyandry in a marine turtle: females make the best of a bad job. Proc. Natl. Acad Sci. U.S.A. 101, 6530–6535.

Limpus, C.J., Nicholls, N., 1988. The Southern Oscillation regulates the annual numbers of green turtles (*Chelonia mydas*) breeding around northern Australia. Aust. J. Wildl. Res. 15, 157–161.

Limpus, C.J., Miller, J.D., Parmenter, C.J., Limpus, D.J., 2003. The green turtle, *Chelonia mydas*, population of Raine Island and the northern Great Barrier Reef, 1843–2001. Mem. Queensl. Mus. 49, 349–440.

Mace, G.M., 2004. The role of taxonomy in species conservation. Philos. Trans. R. Soc. Lond., B Biol. Sci. 359, 711–719.

Mace, G.M., 2005. Biodiversity — an index of intactness. Nature (News and Views) 434, 32–33.

Mace, G.M., Possingham, H.P., Leader-Williams, N., 2006. Prioritizing choices in conservation.. In: Macdonald, D., Service, K. (Eds.), Key Topics in Conservation Biology. Blackwell Publishing, Oxford, UK, pp. 17–34.

Marcovaldi, M.A., Chaloupka, M., 2007. Conservation status of the loggerhead sea turtle in Brazil: an encouraging outlook. Endanger. Species Res. 3, 133–143.

Marcovaldi, M.A., Lopez, G.G., Soares, L.S., Santos, A.J.B., Bellini, C., Barata, P.C.R., 2007. Fifteen years of hawksbill (*Eretmochelys imbricata*) sea turtle nesting in northern Brazil. Chelonian Conserv. Biol. 6 (2), 223–228.

Marsh, H., Dennis, A., Hines, H., Kutt, A., McDonald, K., Weber, E., Williams, S., Winter, J., 2007. Optimizing allocation of management resources for wildlife. Conserv. Biol. 21, 387–399.

Mast, R.B., Seminoff, J.A., Hutchinson, B.J., Pilcher, N.J., 2006a. The role of IUCN Marine Turtle Specialist Group in setting priorities for sea turtle conservation. Mar. Turt. Newsl. 113, 16–18.

Mast, R.B., Bailey, L.M., Hutchinson, B.J., Rowe, M.S., 2006b. The State of the World's Sea Turtles Report, vol. 1. Conservational International, Washington, D.C. 40 pp.

Mast, R.B., Bailey, L.M., Hutchinson, B.J., Hutchinson, A., Koenig, K., Rowe, M.S., 2007. The State of the World's Sea Turtles Report, vol. 2. Conservational International, Washington, D.C. 48 pp.

Mazaris, A.D., Fiksen, O., Matsinos, Y.G., 2005. Using an individual-based model for assessment of sea turtle population viability. Pop. Ecol. 47, 179–191.

McClenachan, L., Jackson, J.B.C., Newman, M.J.H., 2006. Conservation implications of historic nesting beach loss. Front. Ecol. Environ. 4, 290–296.

Meylan, A.B., Donnelly, M., 1999. Status justification for the listing of the hawksbill turtle (*Eretmochelys imbricata*) as critically endangered on the 1996 IUCN Red List of Threatened Animals. Chelonian Conserv. Biol. 3, 201–224.

Miller, J.D., 1997. Reproduction in sea turtles. In: Musick, J.A., Lutz, P.L. (Eds.), The Biology of Sea Turtles. CRC Press, Boca Raton, Florida, pp. 51–82.

Miller, R.M., Rodríguez, J.P., Aniskowicz-Fowler, T., Bambaradeniya, C., Boles, R., Eaton, M.A., Gärdenfors, U., Keller, V., Molur, S., Walker, S., Pollock, C., 2007. National Threatened species listing based on IUCN Criteria and regional guidelines: current status and future perspectives. Conserv. Biol. 21, 684–696.

Morris, W.F., Bloch, P.L., Hudgens, B.R., Moyle, L.C., Stinchcombe, J.R., 2002. Population viability analysis in endangered species recovery plans: past use and future improvements. Ecol. Appl. 12, 708–712.

Mortimer, J.A., 1991. Recommendations for the management of marine turtle populations of Pulau Sipadan, Sabah. Final Report to WWF Malaysia, Project No. 3868. 36 pp.

Mortimer, J.A., Donnelly, M., in review. MTSG Red List assessment of the global status of hawksbill turtles (*Eretmochely imbricata*).

Musick, J.A., 1999. Criteria to define extinction risk in marine fishes. Fisheries 24, 6–14.

Mrosovsky, N., 2002. Hype. Mar. Turt. Newsl. 96, 1–4.

Mrosovsky, N., 2003. Predicting extinction: fundamental flaws in the IUCN's Red List system, exemplified by the case of sea turtles. University of

Toronto Press, Toronto, Ontario, Canada. http://members.seaturtle.org/mrosovsky.

Mrosovsky, N., 2006. Does the Mediterranean green turtle exist? Mar. Turt. Newsl. 111, 1–2.

Naro-Maciel, E., Formia, A., 2006. Sea turtle subpopulations and the IUCN Red List: a complementary role for conservation genetics. Mar. Turt. Newsl. 114, 6–8.

National Marine Fisheries Service and U.S. Fish and Wildlife Service, 1998. Recovery Plans for U.S. Pacific Populations of Marine Turtles. National Marine Fisheries Service, Silver Spring, MD.

Ocean Wildlife Campaign, 1997. The Forgotten Giants: giant ocean fishes of the Atlantic and Pacific. Ocean Wildlife Campaign. Washington, D.C.

Parsons, J.J., 1962. The Green Turtle and Man. University of Florida Press, Gainesville. 126 pp.

Parsons, J.J., 1972. The hawksbill turtle and the tortoise shell trade. Études de géographie tropicale offertes a Pierre Gourou. Mouton, Paris, pp. 45–60.

Pauly, D., 1995. Anecdotes and the shifting baseline syndrome of fisheries. Trends Ecol. Evol. 10, 430.

Pianka, E.R., 1974. Evolutionary Ecology. Harper and Row, New York. 356 pp.

Piet, G.J., Rice, J.C., 2004. Performance of precautionary reference points in providing management advice on North Sea fish stocks. ICES J. Mar. Sci. 61, 1305–1312.

Possingham, H.P., Andelman, S.J., Burgman, M.A., Medellín, R.A., Master, L.L., Keith, D.A., 2002. Limits to the use of threatened species lists. Trends Ecol. Evol. 17, 503–508.

Pritchard, P.C.H., 1999. Status of the black turtle. Conserv. Biol. 13, 1000–1003.

Pritchard, P.C.H., 2002. Global status of sea turtles: an overview. Document INF-001 prepared for Inter-American Convention for the Protection and Conservation of SeaTurtles, First Conference of the Parties (COP1IAC), First Part 6–8 August, 2002.

Reiman, B.E., Allendorf, F.W., 2001. Effective population size and genetic conservation criteria for Bull Trout. North Am. J. Fish. Manage. 21, 756–764.

Rice, J.C., Legace, I., 2007. When control rules collide: a comparison of fisheries management reference points and IUCN criteria for assessing risk of extinction. ICES J. Mar. Sci. 64, 718–722.

Richardson, J.I., Hall, D.B., Mason, P.A., Andrews, K.M., Bjorkland, R., Cai, Y., Bell, R., 2006. Eighteen years of saturation tagging data reveal a significant increase in nesting hawksbill sea turtles (*Eretmochelys imbricata*) on Long Island, Antigua. Anim. Conserv. 9, 302–307.

Rittel, H.W.J., Webber, M.M., 1973. Dilemnas in a general theory of planning. Policy Sci. 4, 155–169.

Rivalan, P., Dutton, P.H., Baudry, E., Roden, S.E., Girondot, M., 2006. Demographic scenario inferred from genetic data in leatherback turtles nesting in French Guiana and Suriname. Biol. Conserv. 130, 1–9.

Roman, J., Palumbi, S.R., 2003. Whales before whaling in the North Atlantic. Science 301, 508–510.

Saba, V.S., Santidrian-Tomillo, P., Reina, R.D., Spotila, J.R., Musick, J.A., Evans, D.A., Paladino, F.V., 2007. The effect of the El Niño Southern Oscillation on the reproductive frequency of eastern Pacific leatherback. J. Appl. Ecol. 44, 395–404.

Santidrián Tomillo, P., Vélez, E., Reina, R.D., Piedra, R., Paladino, F.V., Spotila, J.R., 2007. Reassessment of the leatherback turtle (*Dermochelys coriacea*) nesting population at Parque Nacional Marino Las Baulas, Costa Rica: effects of conservation efforts. Chelonian Conserv. Biol. 6, 54–61.

Sarti Martínez, L., Barragán, A.R., García Muñoz, D., García, N., Huerta, P., Vargas, F., 2007. Conservation and biology of the leatherback turtle in the Mexican Pacific. Chelonian Conserv. Biol. 6, 70–78.

Schroeder, B., Murphy, S., 1999. Population surveys (ground and aerial) on nesting beaches. In: Eckert, K.L., Bjorndal, K.A., Abreu-Grobois, F.A., Donnelly, M. (Eds.), Research and Management Techniques for the Conservation of Sea Turtles. IUCN/SSC Marine Turtle Specialist Group Publication, vol. 4, pp. 45–55.

Seminoff, J.A., 2004a. MSTG global assessment of green turtles (*Chelonia mydas*) for the IUCN Red List. IUCN Species Survival Commission, April 2004. http://www.iucn-mtsg.org/red_list/cm/MTSG_Chelonia_mydas_Assessment_April-2004.pdf.

Seminoff, J.A., 2004b. Guest Editorial: sea turtles, Red Listing, and the need for regional assessments. Mar. Turt. Newsl. 106, 4–6.

Seminoff, J.A., Jones, T.T., Resendiz, A., Nichols, W.J., Chaloupka, M.Y., 2003. Monitoring green turtles (*Chelonia mydas*) at a coastal foraging area in Baja California, Mexico: multiple indices to describe population status. J. Mar. Biol. Assoc. UK 83, 1355–1362.

Shanker, K., Pandav, B., Choudhury, B.C., 2004. An assessment of olive ridley (*Lepidochelys olivacea*) nesting population in Orissa, India. Biol. Conserv. 115, 149–160.

Sheldon, K.E., DeMaster, D.P., Rugh, D.J., Olson, A.M., 2001. Developing a classification criterion under the U.S. Endangered Species Act: Bowhead whales as a case study. Conserv. Biol. 15, 1300–1307.

Shrader-Frechette, K., McCoy, E.D., 1999. Molecular systematics, ethics, and biological decision making under uncertainty. Conserv. Biol. 13, 1008–1012.

Spotila, J.R., Reina, R.D., Steyermark, A.C., Plotkin, P.T., Paladino, F.V., 2000. Pacific leatherback turtles face extinction. Nature 405, 529–530.

Taylor, B.L., 1995. The reliability of using population viability analysis for risk classification of species. Conserv. Biol. 9, 551–558.

Thome, J.C.A., Baptistotte, C., Moreira, L.M.P., Scalfoni, J.T., Almeida, A.P., Rieth, D.B., Barata, P.C.R., 2007. Nesting biology and conservation of the leatherback sea turtle (*Dermochelys coriacea*) in the State of Espírito Santo, Brazil, 1988–1989 to 2003–2004. Chelonian Conserv. Biol. 6, 15–27.

Troëng, S., Rankin, E., 2005. Long-term conservation efforts contribute to positive green turtle *Chelonia mydas* nesting trend at Tortuguero, Costa Rica. Biol. Conserv. 121, 111–116.

Troëng, S., Chaloupka, M., 2007. Variation in adult annual survival probability and remigration intervals of sea turtles. Mar. Biol. 151, 1721–1730.

Vidya, T.N.C., Fernando, P., Melnick, D.J., Sukumar, R., 2005. Population genetic structure and conservation of Asian elephant (*Elephas maximus*) across India. Anim. Conserv. 8, 377–388.

Webb, G.J.W., Carillo, E., 2000. Risk of extinction and categories of endangerment: perspectives from long-lived reptiles. Pop. Ecol. 42, 11–17.

Whiting, A., in review. MSTG global assessment of the flatback turtle (*Natator depressus*) for the IUCN Red List. Draft review submitted to MTSG Assessment Steering Committee.

Wilcove, D.S., McMillan, M., Winston, K.C., 1993. What exactly is an endangered species? An analysis of the Endangered Species List, 1985–1992. Conserv. Biol. 7, 87–93.

Zbinden, J.A., Largiader, C.R., Leippert, F., Margaritoulis, D., Arlettaz, R., 2007. High frequency of multiple paternity in the largest rookery of Mediterranean loggerhead sea turtles. Mol. Ecol. 16, 3703–3711.