Measuring progress toward global marine conservation targets

Colette CC Wabnitz[1*], Serge Andréfouët[2], and Frank E Muller-Karger[3]

Marine species and their habitats are facing widespread overexploitation and degradation, respectively. In response to urgent calls for their protection, the international community agreed to establish representative networks of marine protected areas by 2012 that would conserve and protect 10–30% of specific habitats. To achieve these goals will require reliable estimates of the total area occupied by each habitat. We evaluated this assumption for coral reefs by generating estimates of coral reef area from high-spatial-resolution, remotely sensed imagery (30-m resolution Landsat data), and comparing these with existing published data (usually >1-km resolution). Discrepancies between previous estimates and our values ranged from +1316% to –64%. This uncertainty is incompatible with realistic achievement of the 10–30% conservation targets. We conclude that currently available estimates of the global extent of most coastal marine habitats are based on data that are too poorly resolved to be useful in evaluating progress toward the 2012 targets. Most countries will therefore be unable to demonstrate that they have fulfilled their commitments to marine biodiversity conservation. We urge that accurate inventories be conducted, in a cost-effective fashion, through analyses of available high-spatial-resolution satellite imagery.

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In an effort to halt and reverse the worldwide destruction and overexploitation of marine resources, members of the international community (ie governments, non-governmental organizations, international organizations, and research institutions) have agreed to ambitious biodiversity conservation goals. For marine systems, three global protection initiatives have been outlined over the past decade. The 2002 World Summit on Sustainable Development (WSSD), in its Plan of Implementation, pledged to establish a network of marine protected areas (MPAs), representative of a variety of habitats around the world, by 2012. The following year, members of the 2003 Fifth World Parks Congress (WPC) adopted a recommendation to "[g]reatly increase the marine and coastal area managed in marine protected areas by 2012", further specifying that "these networks should include strictly protected areas that amount to at least 20–30% of each habitat". In 2006, the Eighth Ordinary Conference of the Parties to the Convention on Biological Diversity (CBD) set a target involving effective conservation of 10% of each of the world's ecological regions, specifically including coastal and marine realms, by 2010. For marine systems, emphasis has been placed on vulnerable tropical marine and coastal habitats such as coral reefs (Figure 1), mangroves, and seagrass beds.

Two assumptions are implicit in these habitat targets. First, the protection of representative tropical habitats will conserve biodiversity at species and population levels (Ferrier *et al.* 2002). Although there are few explicit tests of this assumption, protecting representative habitats does seem to be a feasible compromise between "perfect" biodiversity information and the current capacity for data collection (Pressey 2004). Second, for the establishment of MPA networks, it is assumed that quantitative and standardized estimates of the surface area and level of representation (commonly referred to as "representativity" or "representativeness") of individual ecosystems at country and regional scales exist. Such standardized metrics are currently lacking (Green *et al.* 2005), and existing datasets tend to be confounded by errors, either when a user excludes an area from the category to which it belongs (error of omission) or when the user includes an area in an incorrect category (error of commission).

[1]*Sea Around Us Project, Fisheries Centre, Aquatic Ecosystems Research Laboratory, University of British Columbia, Vancouver, BC, Canada* [*]*(colette.wabnitz@gmail.com);* [2]*Institut de Recherche pour le Développement, Anse Vata, Nouméa Cedex, New Caledonia;* [3]*School for Marine Science and Technology, University of Massachusetts Dartmouth, New Bedford, MA*

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Need for accurate baselines on tropical marine habitat areas

All three initiatives listed above concede that there are large gaps in our knowledge of the area of selected tropical ecosystems. However, the quality of data currently used to define the area of habitats for inclusion in MPA networks is often poorer than acknowledged in the literature. Representative results from a global comparison between reef sections mapped by remote sensing under the Millennium Coral Reef Mapping Project (MCRMP; Andréfouët *et al.* 2006) and published estimates of reef area that are currently in use show substantial discrepancies.

Methods

Based on the complete array of reef structures that can be identified from a global set of Landsat satellite images (http://seawifs.gsfc.nasa.gov/cgi/landsat.pl), the MCRMP developed a reef typology applicable at a global scale. This led to the definition of 800 individual reef classes that can be mapped accurately and consistently worldwide (for more detailed information on the development of the classification scheme and the principles followed, see Andréfouët *et al.* 2006). Evidently, no single reef is mapped using all 800 classes; it is the global diversity of detectable reef units with satellite imagery that results in this number. The number of classes for any given reef varies between 1 (eg a fringing slope around a new volcanic island) and several tens (eg < 30 for a complex Maldivian atoll [Figure 2] or > 40 for a large barrier reef system in New Caledonia). The actual mapping process relied on image segmentation and photo-interpretation techniques to delineate individual image segments into meaningful, homogeneous sections, and to label them appropriately. This procedure allows a customized, hierarchical description of any coral reef, anywhere in the world, for any chosen application. In other words, initiatives with a fisheries' focus may wish to include large back-reef sedimentary zones as part of their coral reef definition, whereas applications looking at carbonate production, for instance, may wish to only include zones with a high probability of high hard coral cover and coralline cover (eg fore-reefs, reef crests, and reef flats). Given the methodology developed, MCRMP geographic information system (GIS) products allow for such customized estimates to be calculated at regional scales, according to a consistent "reef labeling" system.

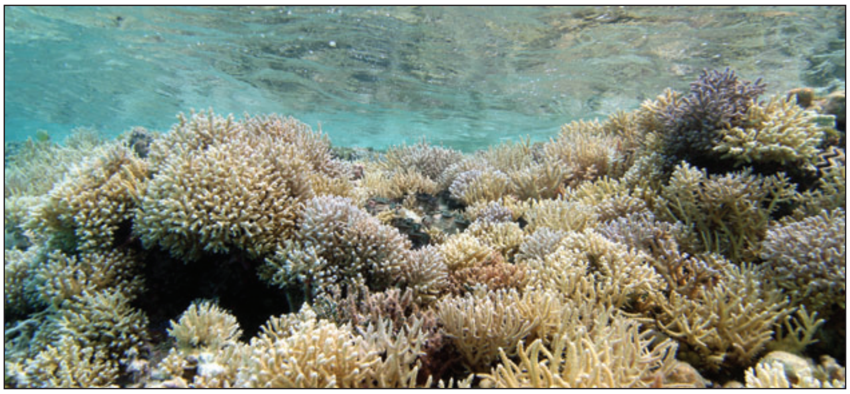

Figure 1. *Shallow, dense area of branching corals in Napuka Atoll, French Polynesia.*

Results and discussion

MCRMP estimates of total reef areas for countries or regions presented here (Table 1) were obtained by merging only the highly productive sections, such as reef flats, fore-reefs, and lagoons, with dense construction (Klumpp and McKinnon 1989). Previously published reef-area data (used here in Table 1 for comparative purposes) were estimated by authors from various sources, sampled at different spatial resolutions. Underestimations reached 1316% (for Palau) and overestimations were on average 50% (Table 1). For the Maldives (Figure 2), a comparison of MCRMP values with previously published reef-area estimates (Naseer and Hatcher 2004), also derived from Landsat imagery, seems to indicate an overestimate of 37%. However, once MCRMP classes were merged in accordance with Naseer and Hatcher's (2004) reef classification scheme (ie including sandy back-reef flat sections), total reef area was found to be comparable: 4092 km^2 and 4285 km^2, respectively. This highlights the importance of using a consistent "reef" definition, even when using similar data sources.

Reef-area estimates for Myanmar (Table 1) are presented to draw attention to challenges associated with habitat mapping in turbid waters. Myanmar's coastal waters are characterized by heavy suspended particulate loads, mainly as a direct result of the large sedimentary discharge from the Ayeyarwady (Irrawaddy) River. Sediment-laden waters reflect more sunlight, rendering the coastal zone "opaque" to optical satellite sensors. As a consequence, small and shallow reefs that may exist even in turbid waters may not be visible on Landsat imagery. Although the MCRMP value suggests that reef area has previously been overestimated, this figure should be considered as a conservative estimate (Table 1). However, in the case of Myanmar, we do believe that earlier values were overestimated, given the spatial distribution of reefs evident from previously available maps (Spalding *et al.* 2001).

Thus, overall, discrepancies highlighted in Table 1 are primarily due to:

(1) *Differences in spatial resolution, environmental quality of the data, and mapping methodology*. MCRMP's dataset was developed from satellite imagery with a 30-m resolution. Most previously published reef estimates listed in Table 1 were derived from navigational charts and topographic map series, re-sampled at a 1-km scale (Spalding *et al.* 2001). In most instances, this coarse spatial resolution will lead to an overestimation of reef area, as most reefs (especially fringing reefs) are unlikely to attain such width. Improving the spatial resolution of data used (ie from navigational charts to Landsat imagery) will tend to reduce the calculated reef area. For example, in the case of the Maldives, image analysis by the MCRMP and Naseer and Hatcher (2004) led to reef estimates of approximately 4000 km^2, whereas calculations based on 1-km re-sampled data provided a value of 8920 km^2 (Spalding *et al.* 2001). However, depending on the spatial configuration of reefs, increasing the spatial resolution of the source data may in some instances lead to new estimates actually being *greater*

than the previous figures. In the case of Palau, for example, reef area mapped by means of Landsat imagery was 1316% greater than previously estimated. Using images with finer spatial resolution allowed for small reef structures, invisible at coarse resolution, to be detected. Trends may also in part be influenced by image quality, so that a poor high-resolution image may ultimately be less useful than mapping based on an excellent quality lower-resolution image.

(2) *The lack of a consistent and systematic definition of "coral reef" driving the inclusion (or exclusion) of information held in currently used databases.* As pointed out above, in the case of the Maldives, differences in the definition of what constitutes a "reef" led to discrepancies in estimates of reef area. This in itself does not constitute a problem if the data used for mapping purposes can consistently deal with changes in reef definitions, ie systematically include or exclude given sections, depending on the chosen definition.

The wide-scale application of remote sensing would also considerably improve mapped distributions of seagrass and mangrove habitats, for which existing and currently used inventories suffer from similar problems of inaccuracy (Spalding *et al.* 2003), as well as saltmarshes, for which reliable inventories do not exist, other than a few studies at local and regional scales (eg Isacch *et al.* 2006). The International Society for Mangrove Ecosystems (ISME) and its partners, including the Food and Agriculture Organisation of the United Nations (FAO) and the United Nations Environment Programme-World Conservation Monitoring Centre (UNEP-WCMC), are presently working on updating the digital database of mangrove forests around the world (M Spalding pers

Table 1. Estimates of coral reef habitat area

Countries/territories	*MCRMP (km^2)*	*Previous figure (km^2)*	*% difference*	*Site specifics and conservation planning activities*
Bahamas and Turks and Caicos	6213	3880 [1]	(+)60	Ongoing MPA network design and implementation; reef connectivity study.
Belize	893	1330 [1]	(–)33	Includes fringing, patch, and barrier reefs and atolls. Numerous conservation activities, currently probably the most studied Caribbean site.
French Polynesia	2140	6000 [1]	(–)64	Here, four archipelagos (Tuamotu, Austral, Society [including Tahiti], and Gambier); atolls with Biosphere Reserve status.
Maldives	2697	4285 [2]	(–)37	Largest reef and lagoon system in the central Indian Ocean.
Micronesia	3172	5440 [1]	(–)42	Focus of the Micronesia Challenge, which targets protection of 30% of coastal waters.
Myanmar	577	1870 [1]	(–)69	Extensive island archipelagos and fringing reef systems; turbid waters limit remote-sensing performances (see main text for details).
New Caledonia	4537	5980 [1]	(–)24	Includes the Chesterfield-Bellona reef system. New Caledonia reefs designated UNESCO World Heritage Site in July 2008.
Guam and Commonwealth of the Northern Mariana Islands	284	263 [3]	(+)8	Pacific Ocean reefs recently mapped by NOAA with 4-m-resolution IKONOS imagery. Areas comparable to MCRMP, based on 30-m Landsat data.
Palau	708	<50 [1]	(+)1316	One of longest barrier reefs in the Pacific Ocean, and one of the highest diversity of reef geomorphological units for a single oceanic island.
Papua New Guinea (Milne Bay)	3009	8110 [1]	(–)63	Focus of numerous conservation planning activities.

Notes: Representative comparison of coral reef areas as estimated by the Millennium Coral Reef Mapping Project (MCRMP), based on remote sensing (Landsat Enhanced Thematic Mapper Plus [ETM+]), with previously published estimates from Spalding *et al.* (2001); Naseer and Hatcher (2004); NCCOS (2005). Spalding *et al.* (2001) based their estimates on Nautical Chart Digitization, while Naseer and Hatcher (2004) and NCCOS (2005) also used remote sensing, but followed a different reef classification scheme from the MCRMP (see main text). "% difference" values were calculated as follows: ([MCRMP – previous figure]/previous figure)*100; (+) in the column indicates countries for which MCRMP-derived reef area is greater than previous estimates, ie, latter were underestimates; (–) indicates countries for which previous figures represent overestimates of reef area. NOAA = National Oceanic and Atmospheric Administration. [1]Spalding *et al.* (2001); [2]Naseer and Hatcher (2004); [3]NCCOS (2005).

comm). In contrast, to our knowledge, there are no current efforts to systematically map seagrass and saltmarsh systems globally, based on consistent remote-sensing methods and habitat typology (but see Wabnitz *et al.* [2008], in which the authors present results from a mapping exercise conducted for seagrasses at the scale of the Wider Caribbean Region).

Given the magnitude of estimated discrepancies between datasets in the case of coral reefs and the lack of a baseline for most other marine habitats, it is clearly challenging to realistically evaluate actions seeking to conserve at least 10% of tropical coastal habitats (eg Wells *et al.* 2007). Current estimates of loss rate average a minimum of 1–2% yr^{-1} for saltmarsh (Lotze *et al.* 2006), 2–5% yr^{-1} for seagrass (Orth *et al.* 2006), and 2–4% yr^{-1} for mangrove (Valiela *et al.* 2001) ecosystems. Although declines in the distribution and area of coastal habitats have been identified as an important indicator of environmental change, the utility of such trend metrics may be futile if we do not have accurate baselines against which to assess the magnitude of losses. Large-scale mapping based on remote sensing and consistent habitat classifications would allow us to revise, and to estimate more accurately, regional/global loss rates of these unique, highly productive, and valuable ecosystems.

Need for functional information derived from habitat inventories

Effective MPAs will require the protection of not only taxonomic biodiversity, but also the functional processes of ecosystems. For example, connectivity between reefs and other marine habitats is an important feature listed under the CBD framework, and needs to be incorporated into MPA network design. As such, the mapped resolution of habitats, as well as their shape and spacing, has important implications for managing connectivity between reefs. Data generated by the MCRMP show that many small coral reef areas are entirely missing from current regional resource maps (Figure 3), with considerable implications for the efficient design of "connected" MPA networks.

The way forward: the use of remote sensing to establish accurate baselines

Although the benefits of remote sensing for biodiversity conservation purposes have been recognized for well over a decade (eg Roughgarden *et al.* 1991) and repeatedly emphasized since then, wide-scale application to the coastal realm remains limited. Remote sensing offers the potential to inform conservation planning meaningfully, for example by: (1) collecting data at scales that cannot be realized through traditional methods; and (2) allowing map classification schemes to be developed in a manner that is consistent, systematic, repeatable, and spatially exhaustive. Consistency and comparability of habitat datasets, in terms of information quality and quantity, are essential for any future assessment of large-scale conservation priorities for biodiversity protection and reserve efficacy. The applicability of remote sensing to the development of reliable, accurate, and relatively detailed, large-scale coastal habitat maps makes it an invaluable tool for effectively realizing desired habitat protection targets (eg 10–30%).

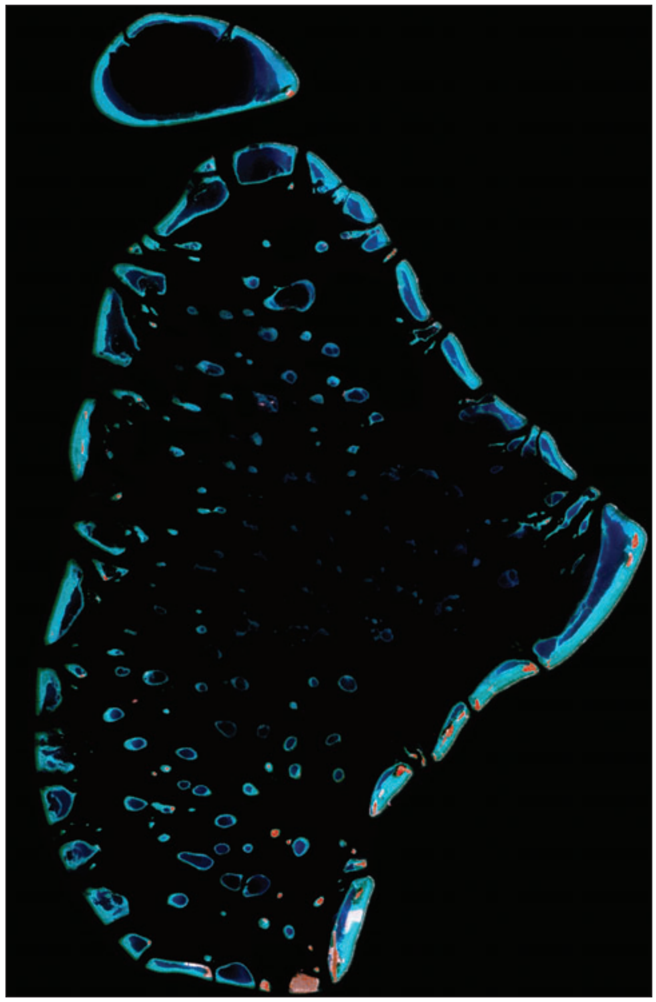

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Figure 2. *Satellite view of North Male Atoll, Maldives. An atoll such as this one is typically characterized by < 30 classes (see Methods section for details on how reef sections were classified).*

There are currently several obstacles to the international community's ability and willingness to map biodiversity via remote sensing. First, donor agencies most often invest in novel approaches to biodiversity conservation and/or visible campaign actions, rather than in the generation of large-scale baseline habitat databases. Second, there is a common misconception that reliable coastal ecosystem inventories are readily available, which has greatly impeded the funding of large-scale, high-resolution mapping programs (but see MCRMP initiated by the US National Aeronautics and Space Administration). Third, although many small-scale habitat mapping programs are being conducted around the world to inform the designation of conservation areas at a national

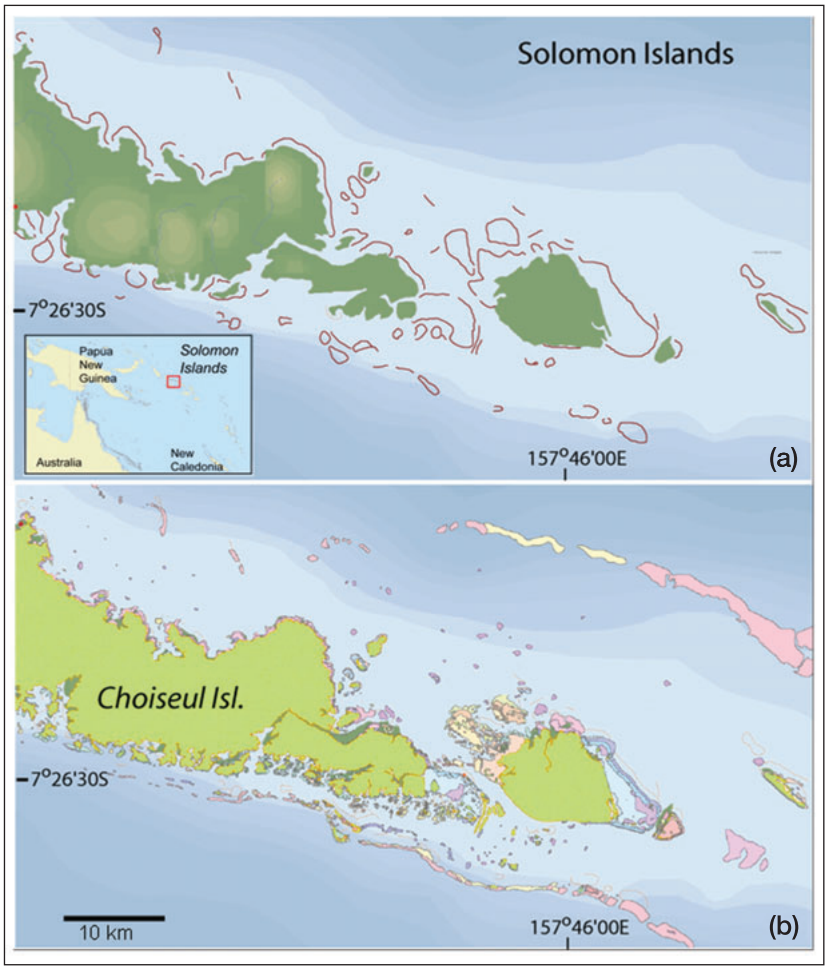

Figure 3. *Reef habitat connectivity. (a) UNEP-WCMC reef layers displayed with the online Reefbase GIS system (www.reefbase.org) for part of the Solomon Islands (see inset map for location – also from Reefbase). Coral reefs (light orange lines) were digitized from nautical charts, and reef areas were estimated from 1-km buffers around these lines (Spalding* et al. *2001). (b) MCRMP coral reef polygons. Different colors depict different reefal and non-reefal categories. Small reefs are adequately resolved, providing a markedly different perception of reef density and potential habitat connectivity from that given by the top panel.*

level, most of these lack systematic labeling, protocols, and standards that would enable their integration into consistent regional databases (Mumby and Harborne 1999). Yet this is fundamental to achieving successful conservation planning at these larger scales. Fourth, comprehensive, detailed mapping efforts conducted at the scale of < 5000 km^2 (eg NCCOS 2005) may have given the impression that such programs, if conducted at the global scale, would be prohibitively expensive. Effective biodiversity conservation need not require such costly programs. For example, the 2005 rezoning of the Great Barrier Reef Marine Park was largely based on simple, but spatially accurate, maps derived from satellite imagery acquired in the 1980s (Jupp *et al.* 1985). The current state of technology and know-how should allow a low-cost strategy to achieve globally what the Australian government created for the Great Barrier Reef.

Conclusions

Globally consistent measures of habitat areas are essential to meaningful assessments of how we are faring with respect to international conservation targets for 2010–2012, and for the large-scale application of predictions and recommendations currently being generated from innovative biodiversity and conservation research. The concept is simple, and the current lack of momentum toward this task is both surprising and unfortunate (but see call for action in the Group on Earth Observations Biodiversity Observation Network initiative; Scholes *et al.* 2008). Standardized global habitat mapping through remote sensing is a cost-effective and high-resolution solution that should be the conservation community's top priority, if we are serious about commitments expressed at the 2002 WSSD. Although we strongly agree with Roberts *et al.* (2003) that "it is a poor strategy to postpone the creation of reserves on the grounds that we are still ignorant of scientific subtleties", the problems identified and detailed here are not subtleties. They represent the basic knowledge required for the effective implementation of MPAs and our ability to ascertain global progress toward international conservation targets.

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