Ecoregional scale seagrass mapping: A tool to support resilient MPA network design in the Coral Triangle

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Ecoregional scale seagrass mapping: A tool to support resilient MPA network design in the Coral Triangle

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ABSTRACT

Seagrass beds are of exceptional economic, ecological and social value in the Coral Triangle. The large number of people who live close to the coast and rely directly on marine resources for food and income paradoxically increases the value of, but also the threats to, these ecosystems. A key strategy of the Coral Triangle Initiative is to protect shallow coastal ecosystems through the design and implementation of resilient networks of marine protected areas (MPAs). This strategy requires accurate spatial data on the distribution and extent of coastal habitats (coral reef, seagrass and mangrove) at scales which match conservation planning decisions. In the Coral Triangle, seagrass distribution maps are not readily available at ecoregional scales. The Lesser Sunda ecoregion, extending from Bali, Indonesia to Timor-Leste, is one of 11 ecoregions of the Coral Triangle and a high priority for conservation and sustainable management of marine resources. To support the design of a resilient MPA network for the Lesser Sunda ecoregion, a seagrass distribution map was generated based on Landsat imagery, literature review and groundtruth data. Seagrass beds were estimated to cover an area of 273 km^2 at an overall accuracy of 78%. Use of the seagrass distribution map in the MPA design improved the habitat representation and connectivity – key criteria for resilient MPA design. The final MPA design included 80 km^2 of seagrass beds, with more than half the beds adjacent to coral reefs and mangroves. This study demonstrates the effective use of Landsat imagery and remote sensing techniques to derive ecoregional scale seagrass maps supporting MPA network design.

1. Introduction

The world's most species rich coastal ecosystems occur within the Coral Triangle (Short et al., 2007; Veron et al., 2009; Spalding et al., 2010), a region which includes all or part of six Indo-Pacific countries; Indonesia, the Philippines, Malaysia (Sabah), Timor-Leste, Papua New Guinea and the Solomon Islands (Fig. 1). These tropical coastal ecosystems are comprised of three main habitats – coral reefs, seagrass beds and mangrove forests. In 2009, governments of the Coral Triangle countries committed to improved management of their coastal ecosystems through the 'Coral Triangle Initiative on Coral Reefs, Fisheries and Food Security' (CTI-CFF). This initiative recognizes the critical importance of coastal ecosystems as sources of food and income to sustain over 100 million people living within the Coral Triangle (Hoegh-Guldberg et al., 2009).

While the focus of CTI-CFF is on coral reefs, Unsworth and Cullen (2010) specifically emphasize the importance of seagrass beds as a conservation priority to sustain coral reef ecosystem functions and fisheries production. Seagrass beds play fundamental ecological roles by providing food, nursery and shelter to fish and invertebrates, many of which are targeted by commercial and subsistence fisheries (Kiswara, 1994; Hemminga and Duarte, 2000; Unsworth et al., 2008; Campbell et al., 2011). Moreover, seagrass beds serve as critical foraging habitats for endangered marine green turtles and dugongs (Lanyon et al., 1989) and play a major role in reducing nutrient and sediment loads to adjacent, naturally

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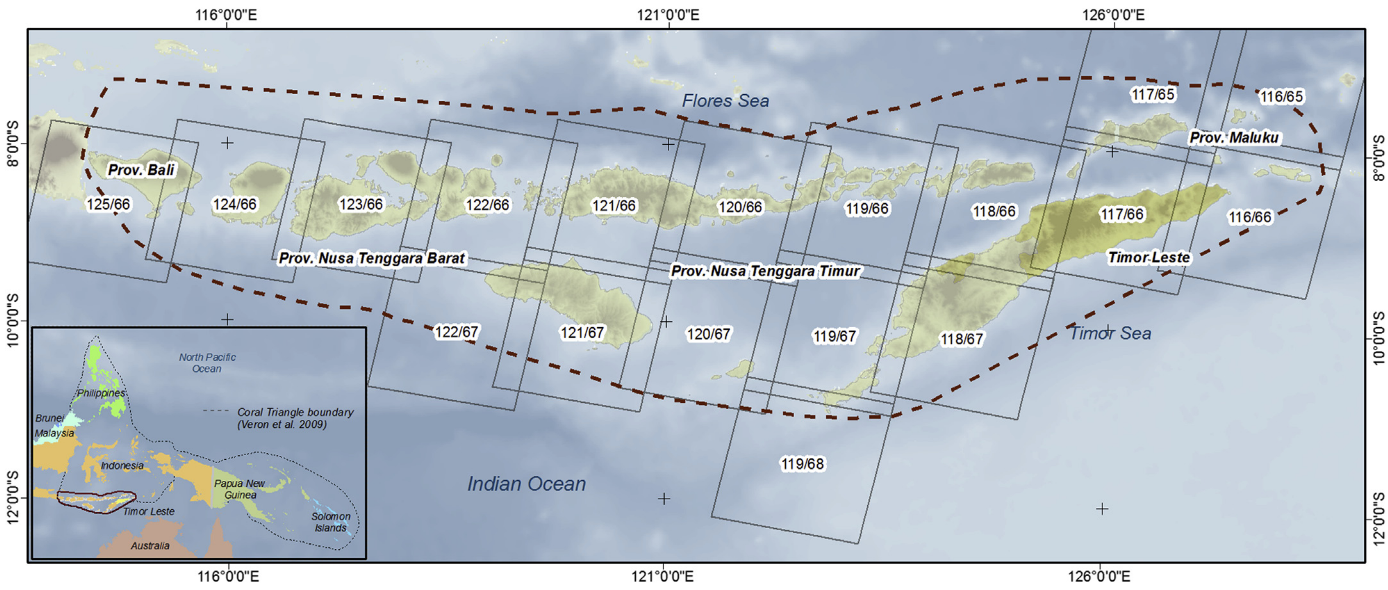

Fig. 1. The Lesser Sunda Ecoregion boundary (dashed line) as part of the Coral Triangle (inset), illustrating the coverage of Landsat scenes (squares) processed for this study.

oligotrophic, coral reefs (Hemminga and Duarte, 2000). Furthermore, seagrasses provide a wealth of economic and ecological goods and services to humans, while at the same time they are intrinsically associated with many cultural and traditional ways of life including food, recreation, and spiritual fulfillment (de la Torre-Castro and Ronnback, 2004).

Despite their importance, extensive areas of seagrass have already been lost in the Coral Triangle (Orth et al., 2006; Waycott et al., 2009) and much of the remaining areas are degraded and at risk from coastal development, reclamation, sedimentation, pollution, seaweed farming, gleaning and overfishing (summarised in Unsworth and Cullen, 2010). On a larger scale, elevated seawater temperatures and extreme weather associated with climate change have been identified as critical threats that will lead to further degradation and loss of existing seagrass beds (Short and Neckles, 1999; Campbell et al., 2006).

The Lesser Sunda Ecoregion (LSE) is the southern-most of 11 marine ecoregions of the Coral Triangle (Green and Mous, 2008) containing 35,802,039 ha of ocean and 10,886 km of coastline (Green and Mous, 2008). This ecoregion encompasses the Indonesian provinces of Bali, West Nusa Tenggara, East Nusa Tenggara, part of southeast Maluku, and the country of Timor-Leste (Fig. 1). LSE is characterized by exceptionally strong and complex currents generated by the passage of the Indonesian Throughflow through the narrow passages between the islands of this archipelago (Gordon and Fine, 1996). Shallow coastal habitats are more extensive on the northern coasts as these are more protected from the high wave energy typical on southern coastlines.

Reef geomorphology plays a significant role in the distribution, abundance and diversity of seagrasses (Carruthers et al., 2002). In LSE, historical knowledge and visual inspection of satellite images suggest that most seagrass habitats occur in close association with fringing reefs, which are the most common reef type within the ecoregion. Fringing reefs dissipate the energy of incoming waves and create a protective environment in the backreef zones and coastal lagoons where seagrasses reach their highest abundances (Tomascik et al., 1997).

One of the main recommended strategies to protect reefs and associated coastal ecosystems in the Coral Triangle from local anthropogenic threats and climate change impacts is to increase their resilience through the design and implementation of networks of marine protected areas (MPAs) (CTI, 2009). The fundamental principles for incorporating resilience into the design of networks of MPAs (McLeod et al., 2009) include; 1) representation and replication of conservation targets (habitats or species) in MPAs, 2) inclusion of critical habitats such as spawning grounds or reef communities that can resist or recover from coral bleaching 3) meeting ecological connectivity standards within and among MPAs (i.e. including multiple habitats and ensuring MPAs are spaced not more than 100-200 km apart) and 4) incorporating factors which would support effective management. Through the application of these principles, multiple areas of reef and associated coastal habitats which have the best chance of resisting or recovering from climate change impacts would be protected from local anthropogenic threats and able to replenish surrounding areas. Identifying areas for management that contain multiple habitats such as coral reef, seagrass and mangrove is the key to increasing resilience through the maintenance of ecological processes that support biodiversity and sustainable fisheries. Based on these principles and extensive stakeholder consultation, an MPA network was designed for the LSE (Wilson et al., 2011) as a key strategy for national and local governments of Indonesia and Timor-Leste. This approach has also been applied successfully in Palau (Hinchley et al., 2007) and Papua New Guinea (Green et al., 2009).

Application of these principles requires information on the distribution and extent of major habitats and species. However, like most tropical Indo-Pacific regions where seagrass is abundant, LSE proved to be data deficient, particularly for seagrass (Waycott et al., 2009). In most cases, available data were too scarce or inadequate to satisfactorily support the MPA network design.

Remote sensing technologies have repeatedly proved to be an effective, if not the only, means to map entire ecosystems and provide practical georeferenced digital documents, at both local (Armstrong, 1993; Andréfouët et al., 2003; Schweizer et al., 2005; Pu et al. 2012) and regional scales (Dahdouh-Guebas et al., 1999; Andréfouët and Guzmán, 2005; Wabnitz et al., 2008; Andréfouët, 2011). At local scales, the potential of very high resolution (1–5 m) remote sensing imagery to map and study tropical seagrass beds is well understood (Dekker et al., 2006). Pilot sites have provided a wealth of quantitative data to demonstrate the potential application to the range of variation in taxonomic composition and density of seagrass beds (Phinn et al., 2008). What remains challenging and poorly understood is the potential for large scale regional seagrass mapping, where *in-situ* data is limited or non-existent, using lower resolution images to map a large variety of seagrass beds across a range of coastal configurations (Wabnitz et al., 2008). Yet, large scale coverage is needed for regional conservation planning (Wilson et al., 2011). As such, this study reports on the production of an ecoregional scale baseline map of shallow-water seagrass habitats derived from Landsat imagery, how the new seagrass product contributed to the design of a resilient MPA network for LSE (Wilson et al., 2011), and the potential to replicate similar methods in other regions under different environmental contexts and data availability.

2. Material and methods

2.1. Landsat, ancillary image data and image pre-processing

The Millennium Coral Reef Mapping Project (Andréfouët et al., 2006) made publicly available archived Landsat imagery acquired between 1999 and 2003 for all coral reef areas in the world (http://oceancolor.gsfc.nasa.gov/cgi/landsat.pl). This archive was the primary source of Landsat imagery used for the mapping, which was performed in 2008 before the free release of the complete Landsat data archive (Wulder et al., 2012).

The LSE is contained within 18 Landsat scenes, or predefined track grids that each represent a ground area of 170 km by 185 km (Fig. 1). For every scene, there was at least one Landsat image available in the Millennium archive. Image data quality was assessed for cloud cover, atmospheric and sea state conditions, tide level and water turbidity to select the optimal set of images for seagrass mapping. In cases where multiple images were available for a single location, images were inspected visually to identify any relevant temporal changes that could assist the interpretation process. Landsat images selected for analysis were collected within a year from each other and mainly under dry season conditions.

In a few cases, where none of the available Landsat images was suitable for a given area, it was necessary to employ ancillary imagery composed of higher spatial resolution (15 m) ASTER images and online visualization tools such as Google Earth®. Despite a higher spatial resolution, the lack of blue band in ASTER imagery is a serious impediment to coastal mapping (Capolsini et al., 2003; Torres-Pulliza, 2004), and Google Earth® images were useful but could not be considered as a reliable source for many areas due to frequent poor image quality.

Landsat images from the Millennium archive are radiometrically and geometrically corrected. Radiance values were converted to at-sensor-reflectance using the Landsat reflectance conversion equations (http://landsathandbook.gsfc.nasa.gov). A standard dark pixel subtraction method was performed to reduce atmospheric effect on image data values. Land, clouds and breaking waves were masked. When required, image mosaics were created to provide continuous coverage of marine features (Fig. 1). Fig. 2 summarizes the steps in image processing applied to this study. This flow chart obeys a fairly standard protocol, intermediate between the producer and user flow charts presented in Andréfouët (2008).

2.2. Groundtruthing – in-situ reference data

The original seagrass groundtruthing protocol for this study explored the potential of obtaining probabilistic ground validation data representative of the entire LSE (Fig. 1) following Congalton (1991) and Stehman (2001). However, this could not be done due to the immense logistic difficulties and high costs of access in such a

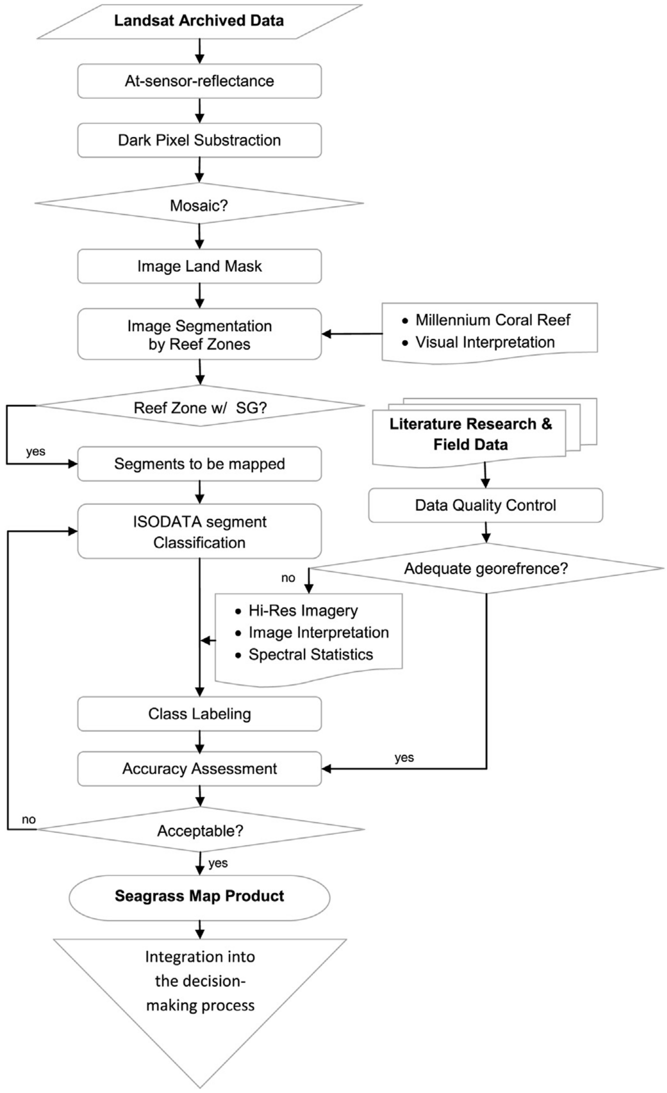

Fig. 2. Flow chart for large-scale seagrass cover mapping in the Lesser Sunda Ecoregion.

remote region. Consequently, a reduced and more realistic number of accessible and representative localities were chosen for field surveys.

We conducted a detailed *in-situ* survey of sites which were both difficult and easy to interpret representing different types of seagrass beds (in composition and density). In June 2008, five field sites were surveyed, four in Bali (Gilimanuk Bay, Terima Bay, Serangan and Sanur) and one along the northeast coast of Komodo Island (Fig. 3). These sites could all be accessed easily and data collected by wading or on snorkel. They included a good representation of all seagrass community types found in Indonesia from sheltered lagoons to more exposed reef flats and encompass muddy and sandy substrates (Ooi et al., 2011). In addition, both challenging heterogenous and easier homogenous areas from the shore to the reef crest were surveyed to give the widest possible range of conditions for the accuracy assessment (Andréfouët, 2008). Seagrass beds were semi-quantitatively characterized along visual transects perpendicular to the coastline, from shore to the seaward limit of the seagrass bed. Transect length and distance between transects varied according with bed size. A total of 356 stations were visually characterized by surveying an area of 5 × 5 m per station, accounting for dominant sediment type, depth and percentage cover

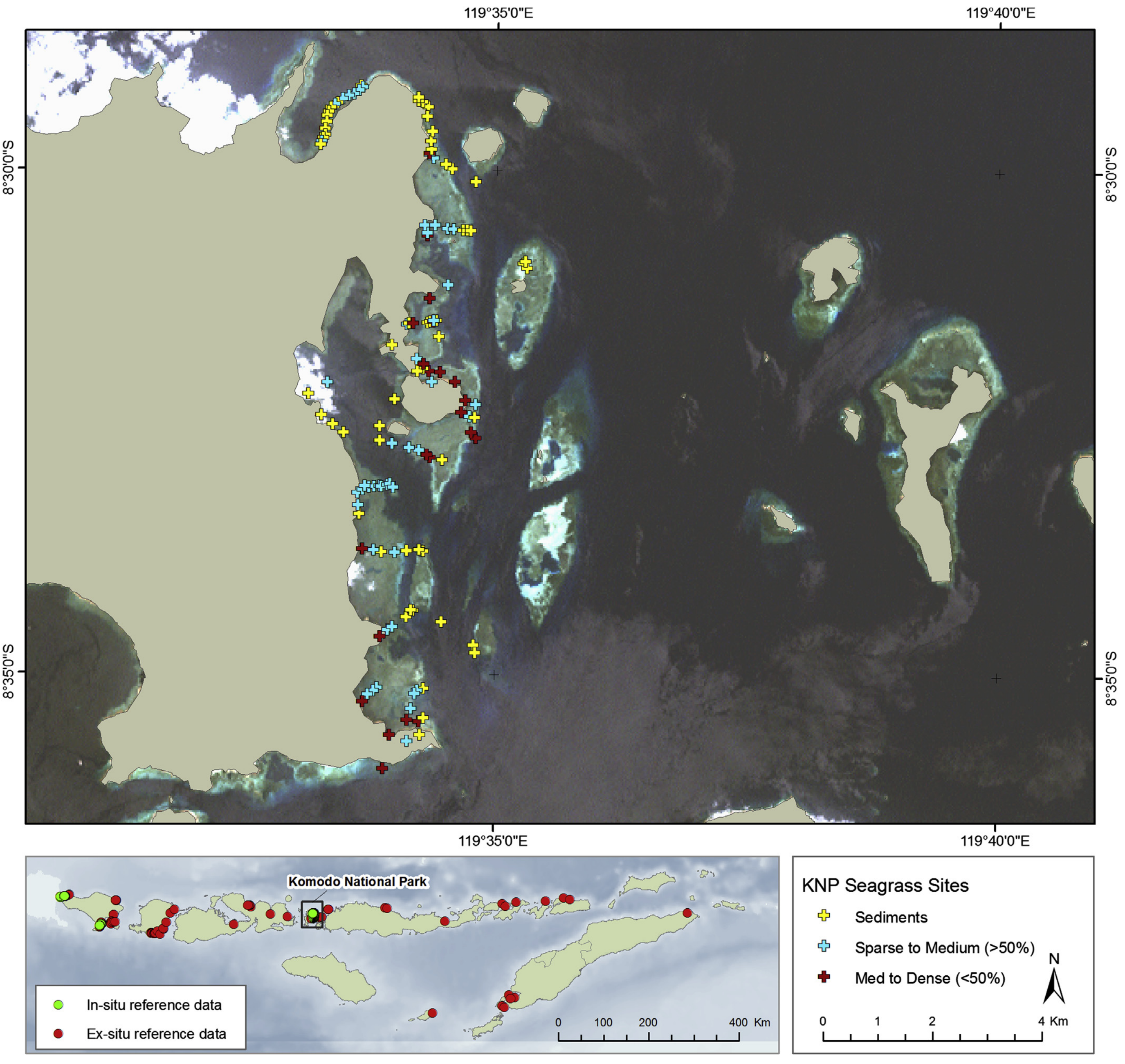

Fig. 3. The upper panel displays the *in-situ* sites and general seagrass density measures collected in part of the Komodo National Park with a Landsat image as background. The bottom panel shows the *in-situ* (green) and *ex-situ* (red) data localities. It also illustrates the enriched spatial data distribution attained with the addition of *ex-situ* reference data points. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

of seagrass, macroalgae, hard corals and soft corals (S Campbell, 2008; unpublished data). Of these, 320 were suitable for use in the accuracy assessment, the remaining 36 were used for class labeling (Section 2.4).

2.3. Literature review – ex-situ reference data

A practical solution to complement the limited number of *in-situ* measurements collected to guide and test an ecoregional-scale seagrass map was to gather an ancillary dataset of reference data *ex-situ*. That is, we relied on existing peer reviewed and grey literature documents to gain insights on the general distribution of seagrasses in LSE. A similar strategy was used by Wabnitz et al. (2008) to enhance seagrass mapping in the Caribbean. Published and unpublished articles, books, technical reports and maps (Kvalvagnaes and Halim, 1979; Polunin et al., 1980; Trono, 1980; Pagcatipunan et al., 1981; Salm, 1984; Nienhuis et al., 1989; Coppejans and Prud'homme van Reine, 1992; Kiswara and Winardi, 1994; Kiswara, 1994; Kiswara et al., 1994; Kiswara, 1996; Tomascik et al., 1997; van Woesik, 1997; Baktosurtanal, 1998; Azkab et al., 1999; Susetiono, 1999; Kiswara and Winardi, 1999; Matsuura et al., 2000; Lourie, 2001; Rudiyanto, 2001, 2002; Pet-Soede, 2002; Kuriandewa et al., 2003a, 2003b; Renema, 2003; Soemodihardjo et al., 2003; Pedju, 2004; Coles and McKenzie, 2005; Short et al., 2007), as well as georeferenced pictures, personal communications and reliable Internet sources were compiled to support the seagrass habitat mapping. The World Atlas of Seagrasses (Green and Short, 2003) served as useful visual reference for general seagrass distribution, but only few of the plotted data points and polygons were geographically precise or documented enough to be helpful. Similar observations were made for Caribbean seagrass mapping (Wabnitz et al., 2008). Overall, the set of compiled documents provided reference data for 126 additional locations. Nonetheless, the additional data confirmed the presence or absence of remote seagrass beds as inferred from Landsat imagery. To avoid bias, a third-party remote sensing analyst, not involved in the image classification and accuracy assessment process, identified and digitized over the imagery the compiled *ex-situ* reference data and made them available for inclusion as additional reference data points.

2.4. Image segmentation and classification based on reef geomorphology

Five reef geomorphologic "zones" (herein called reef zones) were visually identified and digitized over the imagery (Table 1) in agreement with the Millennium Coral Reef Mapping terminology (Andréfouët et al., 2006). Biogeographically homogenous reef-zone segments were analyzed independently. The goal of this segmentation was to *a-priori* minimize spectral confusion during automated image classification. This process of reef classification stratified by geomorphological type has been used in the Caribbean (Wabnitz et al., 2008).

A multi-step image classification approach allowed determination of seagrass occurrence in selected reef zone segments (Fig. 4). First, we ran several iterations of ISODATA unsupervised classification to the segments in search of groups of pixels that shared similar spectral characteristics (Selim and Ismail, 1986). Up to 15 spectral classes for each segment were discriminated, which was considered a reasonable number to separate spectrally all the major benthic habitats present. Second, the classes were assigned to either "seagrass" or "other" classes. Class assignment was based on image interpretation guided by the *ex-situ* dataset, spectral signature similarity with unambiguous local seagrass beds and 36 *in-situ* reference points not included for accuracy assessment. The "other" class consists of habitat classes with likely none or very sparse (<10% coverage) seagrass beds. Third, an error matrix was developed to test seagrass classification accuracy on the basis of presence or absence using 446 *in-situ* and *ex-situ* "seagrass" or "other" groundtruth reference points. In order to reduce any potential errors due to the changes in seagrass density over time, seagrass map accuracy was based on presence and absence and not categorical

Table 1
Reef geomorphologic zones description and area size for LSE as depicted from Landsat imagery (in agreement with the Millennium Coral Reef Project), used as proxies for seagrass presence-absence classification.

Reef-zone	Description	Area (km^2)
Reef flat	An intertidal or subtidal well-developed and relatively horizontal hard substrate, located between the reef crest and shore, and sometimes a not clearly bounded inner slope of a depression (moat or lagoon).	670
Back-reef	A shallow intertidal to subtidal (0.5–15 m) nearly horizontal or gently sloping sedimentary area between the coastline and the reef flat inner edge.	700
Enclosed basin	A sedimentary subtidal depression on reef flats forming pools, deeper than their surroundings.	40
Forereef slope	A gentle or steep coral slope extending seaward from the reef crest. When not obvious, its outer edge is defined by the depth detection limit of corresponding Landsat image.	780
Deep terrace	A subtidal reef feature or platform whose cover cannot be categorized precisely in much detail because of depth.	90
Land	All above-water terrain, including mangrove forests.	390

percent cover values. Map accuracy was estimated by means of user, producer and overall accuracy along with Cohen's kappa, an alternative measure of accuracy between resulting map and one generated by chance (Cohen, 1960; Congalton, 1991).

Isolated pixels were filtered from the resulting raster map and pixel clusters were clumped to reduce noise and produce a more spatially coherent product. The classification map was converted to a vector file suited for conservation planning modeling tools such as Marxan (Ball and Possingham, 2000).

2.5. Use of the seagrass map for LSE MPA network design

Biological and socioeconomic design principles for the LSE MPA network (Wilson et al., 2011) were based on: 1) criteria for identifying MPAs in Indonesia (Wiryawan et al., 2006) and 2) the resilience principles of MPA network design developed by McLeod et al. (2009). Further, the LSE MPA network strategy followed a combination of scientific analysis and expert and stakeholder consultation. Seagrass was considered a key thematic habitat layer along with coral reef, mangrove and estuary layers (Wilson et al., 2011).

Habitat thematic layers served as data variables in the Marxan decision support software (Ball and Possingham, 2000), to identify areas that could be included in a network of MPAs to meet resilient design criteria. Areas that met goals for representation of conservation targets but identified as high 'cost' (i.e. areas used for port development, fishing ground, etc.) were avoided as existing or planned uses would be incompatible with the objectives of an MPA. Inputs from experts and stakeholders were obtained at several stages throughout the project and the MPA network design was modified accordingly (see Wilson et al., 2011). However, after each modification, Marxan was used to check that criteria for representation and connectivity were still being met. Within the MPA network design, areas identified as potential future sites for MPA development were called Areas of Interest (AoI) to prevent confusion with existing or planned MPAs.

3. Results

3.1. Inventory and distribution of seagrass beds

Seagrass beds were estimated to cover 273 km^2 of the 2500 km^2 of shallow-water shelf areas analyzed for the LSE. Seagrass is distributed fairly continuously along the sinuous northern coastlines of the west-east elongated islands throughout the archipelago (Fig. 5). The most extensive seagrass cover (70 km^2) occurs in Flores Island, which is also the island with the second largest shallow-water shelf within the ecoregion. Rote Island takes second place with a seagrass extent estimate of 28 km^2, followed by SE Makulu (Roma, Kisar, Leti, Moa, Lakor Islands) with 21 km^2. Sumbawa Island, with the longest combined coastline (3000 km) and largest shelf area (446 km^2), is fourth in overall seagrass coverage (20 km^2). The five islands with highest seagrass cover relative to shelf size include Savu, SE Maluku Islands, Flores, Bali and Rote (Table 2).

Finally, we found that 23% of the detected seagrass was associated with both mangrove and coral reef habitats, 72% associated with coral reef only, 1% associated with mangrove only, and 4% occurred as isolated communities over sedimentary flats.

3.2. Map accuracy assessment

Bali and Komodo Islands, where a larger number of *in-situ* data were available, returned overall accuracies of 78% with 0.57 kappa and 76% with 0.53 kappa respectively (see Supplementary Data).

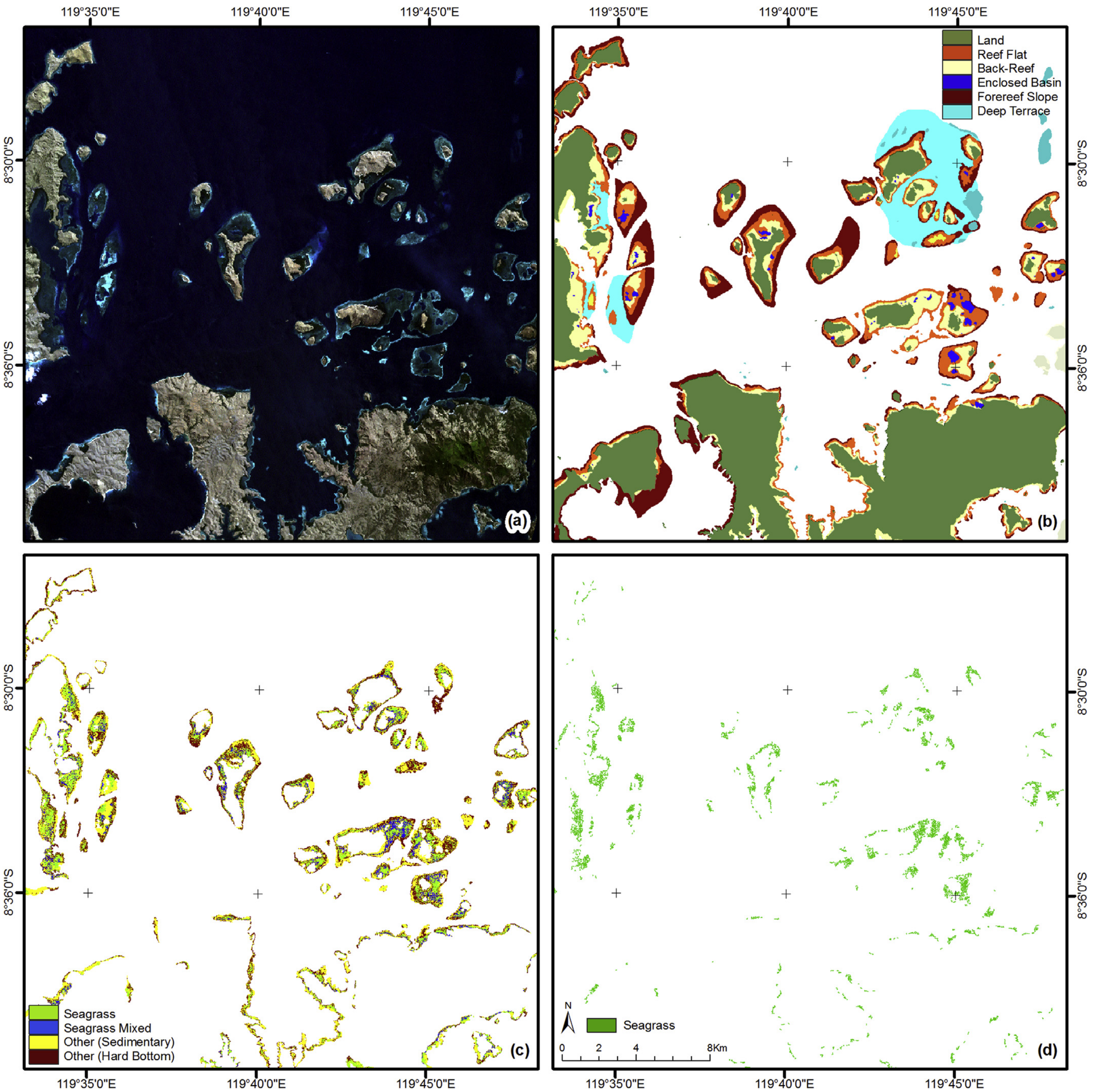

Fig. 4. Example of the seagrass classification process in part of Komodo National Park, (a) shows base Landsat imagery, (b) reef geomorphology zones considered for image segmentation, (c) ISODATA classification output after class labeling, and (d) resulting seagrass beds.

Because the ecoregional map was generated using systematic techniques, we could assume that these accuracy values are representative of the entire LSE (~77%). However this was based on a relatively low number of groundtruthing sites with limited distribution. With the addition of the *ex-situ* data a total of 126 additional "seagrass" or "other" class points were incorporated into the analysis. Overall accuracy based on 446 *in-situ* and *ex-situ* reference points, delivered a high 78% agreement with 0.58 kappa for the entire LSE seagrass map (Fig. 5). User and producer accuracy for "seagrass" were 88% and 69% respectively and, 72% and 89% for the "other" class.

3.3. Level of seagrass protection in the MPA network

The final MPA network design included 100 coastal and marine protected areas encompassing 98 000 km^2 of terrestrial, shallow coastal and 'deep sea yet near-shore' habitats (Fig. 5). Of the 273 km^2 of seagrass habitat identified during this study, 80 km^2 are encompassed within the final MPA network design. This includes about 41 km^2 identified for inclusion through the LSE design process for the first time, 6 km^2 in areas that were previously proposed but are not yet declared and, 33 km^2 that occur within existing MPAs or coastal reserves. In other words, the area of seagrass habitats under

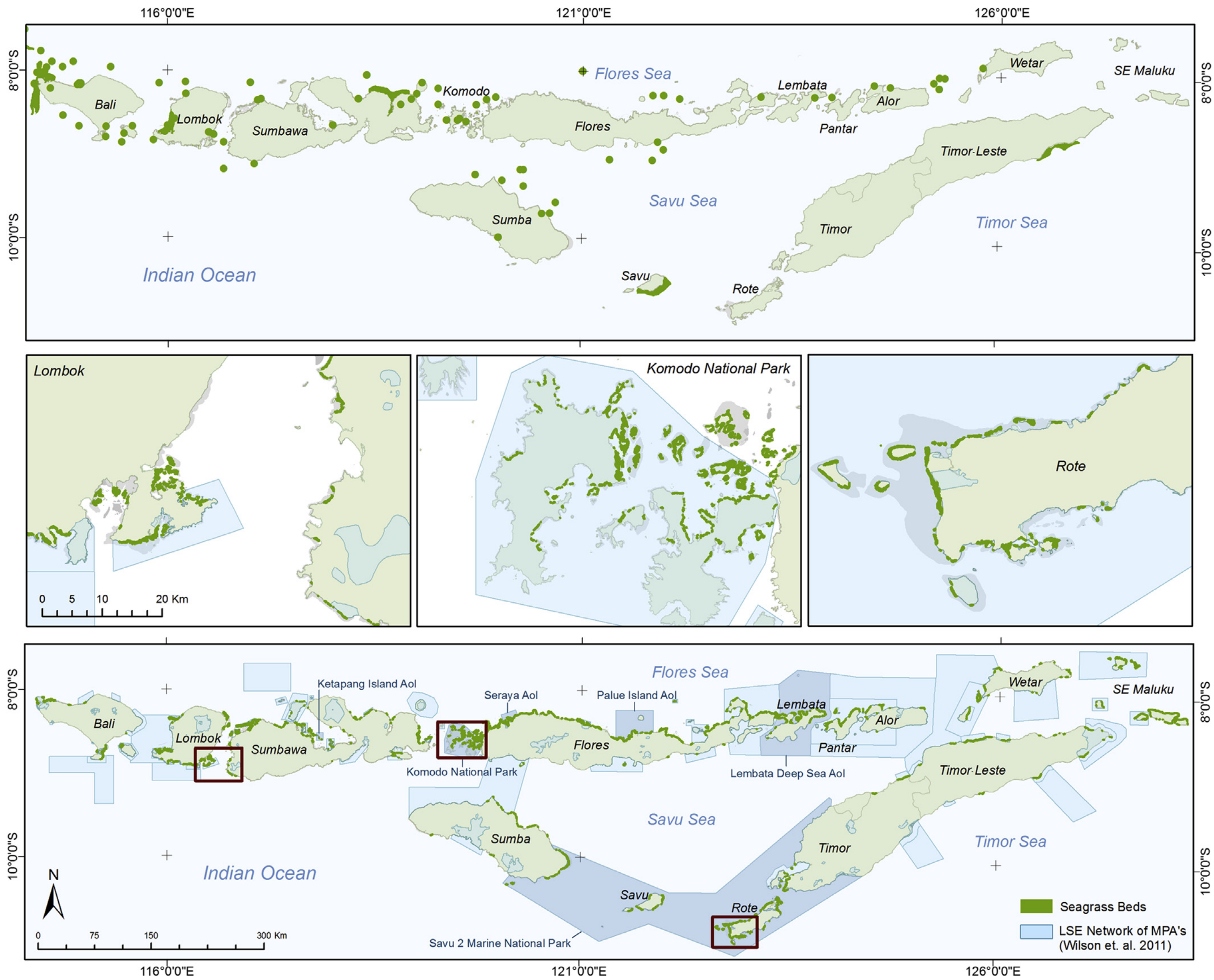

Fig. 5. The seagrass thematic layer (lower panel) for the LSE as derived from this study. Middle panels display in more detail the enhanced spatial coherence obtained using remote sensing techniques compared against the upper panel, which presents the seagrass data compilation made available by UNEP-WCMC (Green and Short, 2003). The latter was the only large-scale seagrass dataset available for the region before this study.

Table 2
Surface area of shallow water seagrass beds as estimated from Landsat imagery.

Province	Islands	LSE seagrass area (km^2)	LSE seagrass area (%)	Analyzed combined shelf area (km^2)
Bali	Bali	12	4	91
West Nusa	Lombok	18	6	160
Tenggara	Sumbawa	20	7	446
East Nusa	Komodo and Rinca	18	7	166
Tenggara	Flores	70	26	406
	Adonara, Solor	3	1	58
	Lembata	12	4	100
	Pantar	7	2	71
	Alor	2	1	46
	Sumba	19	7	213
	Savu	15	5	72
	Rote	28	10	240
Timor-Leste	Timor	19	7	229
Maluku	Roma, Kisar, Leti, Moa, Lakor	21	8	108
	Wetar	9	3	81

protection within the LSE should increase by about 51% with the proposed MPA network described in Wilson et al. (2011).

Of the 71 MPAs or AoIs in the network directly associated with shallow coastal ecosystems, 65 contained some seagrass. The MPA/AoI with the largest seagrass area (10 km^2), occurred within the Komodo National Park, followed by Savu 2 Marine National Park with 8 km^2, Lembata Deep Sea AoI with 7 km^2, Seraya AoI with 6 km^2 and Palue Island AoI with 6 km^2. The smallest coverage occurred in Ketapang Islands AoI with an estimate of 0.03 km^2. The design included 49 km^2 of seagrass beds that were part of a mangrove-seagrass-coral reef association. These habitat associations are represented in 24 of the 71 shallow coastal MPAs or AoIs within the MPA network.

In agreement with the maximum distance (100–200 km) design criteria principle to maintain genetic connectivity (McLeod et al., 2009), the most remote MPA or AoI containing seagrass has at least three other neighboring seagrass AoI within 200 km. The most populated or clustered seagrass AoI includes 31 AoI within the 200 km radius threshold. Within the network, the average distance among the AoI centroids containing seagrass is 109 km.

4. Discussion

In most coral reef areas worldwide, actual patterns of ecological and genetic population connectivity for numerous species remain unknown at ecoregional scales. It is assumed that accurate, spatially-explicit habitat representation at the appropriate scale should provide effective proxies to capture pathways of larval dispersal potential, patterns of migrations and exchanges at different ontogenetic stages, and access to reproduction sites and feeding grounds for a bulk of different taxa (Sale et al., 2010). The guidelines used to design the MPA network make use of the few hard facts on these ecological processes and patterns collected worldwide (McLeod et al., 2009; Grüss et al., 2011). It is known that physical and biological connectivity among coral reefs, mangroves and seagrass habitats support productivity and biodiversity (Unsworth et al., 2008). Improvements in data relating to seagrass spatial distribution and extent and proximity to coral reefs and mangroves throughout the region have benefited the MPA design by improving the accuracy of representation and connectivity of seagrass habitats. As such the LSE MPA design, partly developed using ecoregional maps of key habitats, species and uses, along with a resilient design criteria, offers a practical and reproducible example for other areas of the Coral Triangle and worldwide.

The LSE is rich in seagrass species with 13 known species occurring in mixed or monospecific beds (Kiswara, 1996; Tomascik et al., 1997; Humoto and Moosa, 2005). Seagrasses can be found in association with about seven seaweed species of economical interest (Kuriandewa et al., 2003a) and potentially with 117 species of seagrass-associated macroalgae (Verheij and Erftemeijer, 1993). This high species complexity sets LSE apart from Caribbean seagrasses, where 6 species have been reported (Hemminga and Duarte, 2000), and where monospecific beds are common (Tomascik et al., 1997). It also imposes major challenges for any statistically robust spectral discrimination approach, in particular considering the spatial and spectral detection capabilities of Landsat ETM + imagery (Andréfouët et al., 2003; Capolsini et al., 2003; Phinn et al., 2008). *A-priori* image segmentation based on reef-zones was especially useful to decrease the spectral heterogeneity inherent to the LSE ecological setting prior to automated image classification (Wabnitz et al., 2008; Phinn et al., 2012).

There are a number of potential sources of error in the image analysis and factors which may have affected the map accuracy assessment and final seagrass map product. Firstly, after image segmentation, some seagrass areas were unavoidably excluded from the analysis. That includes, seagrass growing in high-energy reef-crests, fore-reefs or deep terraces which are reef-zones omitted from the classification given the generally low probability of seagrass cover or that are outside the depth detection capabilities of Landsat. Secondly, seagrasses growing in highly turbid areas or very narrow shelf terraces (<90 m). That is the case of the Sumbawa Island, which narrow shelf and large sheltered embayments where the prevalence of turbid waters, as seen from available imagery, could have reduce the chance of detecting the presence or absence of seagrass beds. Thirdly, areas where very sparse (<10%) seagrass cover became spectrally undetectable were also inevitably omitted. Indeed, with Landsat images, it is impossible to detect very low density seagrass beds growing over mixed bottom types with good map accuracy (Wabnitz et al., 2008). Detailed analysis of classification results helped in detecting and contextually editing some areas where variations in substrate or sand composition (calcareous or volcanic) influenced the classification results or when seagrass was confused with spectrally similar seaweed aquaculture sites. Multi-date scenes were also useful in identifying potential misclassification due to nonpersistent cyanobacteria blooms and seasonal variations in algal growth (Wabnitz et al., 2008).

It is expected that temporal variation in seagrass distribution would have influenced the accuracy assessment since the imagery used for the analysis was collected at a different time than the groundtruthing surveys. Seagrass cover can change at different time scales, due to natural variations (growth, grazing, storms, etc.) and anthropogenic stressors (decrease in water quality, higher sedimentation, etc.) (Ertemeijer and Herman, 1994; De longh et al., 1995; Gullstrom et al., 2006). It is therefore possible that changes had occurred between the time of image acquisition and groundtruthing, or that changes had occurred between this assessment based on 1999–2003 images, and the current situation in the LSE. However, given the coarse level of seagrass bed description we used here, and given the relatively short period between the last images and groundtruth period compared to longer remote sensing change detection studies (e.g. 16 years in Gullstrom et al., 2006; 25 years in Knudby et al., 2010) we believe that bias in our assessment due to temporal changes is low. If changes have occurred, our accuracy assessment reports could be either optimistically or negatively biased, of an unknown magnitude, but the short time-scale suggests a small bias.

The number of *in-situ* sites that could be surveyed to groundtruth the map produced by image analysis was less than optimal (Congalton, 1991). This is why ISODATA unsupervised clustering per reef-zone was preferred over the more data-demanding supervised classification. Under a data-deficient scenario it freed available reference data necessary for accuracy assessment. *Ex-situ* data points complemented the *in-situ* data well primarily by introducing a quasi-random data distribution that better complies with probabilistic sampling protocols (Fig. 3). In addition, the simple seagrass presence-absence classification scheme applied here reduced the reference data quality requirements needed for map validation (Stehman, 2001).

Seagrass bed distribution and extent are presented here for remote islands never before included in regional compilations (e.g. Timor-Leste, SE Maluku). However, the lack of *in-situ* data at these remote locations implies some uncertainty in the results and future groundtruth surveys at these sites is highly recommended. In contrast to many studies that do not provide accuracy assessment (see review by Roelfsema et al., 2009), the error is quantified here for representative, albeit small areas, in terms of seagrass presence-absence. The accuracy achieved (78%) based on both *in-situ* and *ex-situ* groundtruth data is considered high and is indicative of the reliability or practical utility of the map (Congalton, 1991). It is comparable to Wabnitz et al. (2008) for the Caribbean after all the different seagrass classes were merged. It is also equivalent to the large-scale seagrass map achieved for the New Caledonia archipelago (Andréfouët, unpublished data). The accuracy assessment results support the qualitative utility of this large-scale seagrass habitat map for regional conservation planning. This seagrass map now complements a data base of ecologically sensitive habitat maps of coral reefs, estuaries and mangrove forests also digitized from Landsat imagery that together provides a greater understanding of the distribution and extent of coastal ecosystems in the LSE. It also represents a major improvement from previously available large-scale seagrass distribution and extent maps for the same area (Fig. 5). Further, in contrast to a static digital document, it can be dynamically improved as additional field data and imagery become available to better serve future management efforts.

This study demonstrates the capabilities of remote sensing to produce seagrass maps at relevant ecoregional scales, as part of conservation planning strategies within the Coral Triangle. The seagrass maps described here provide the first available ecoregional scale map of seagrass beds for the LSE. The maps were instrumental to identify potential conservation priority sites, assist expert decision making, guide conservation strategies and, ultimately design a resilient network of MPAs. The resulting design (Wilson et al., 2011) will be used as the main reference for establishing a network of MPAs in the LSE by the Indonesian Ministry of Marine Affairs and Fisheries and the Government of Timor-Leste and support sustainable use of marine resources for the benefit of local communities.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.ocecoaman.2013.04.005.

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