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Changes in the landscape pattern of the La Mesa Watershed – The last ecological frontier of Metro Manila, Philippines

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ABSTRACT

The La Mesa Watershed (LMW) is considered as the 'lungs' and the last ecological frontier of the Philippines' National Capital Region, Metro Manila. It is among the many watersheds in the country that suffered from severe deforestation in the past. Nevertheless, over the past few decades, reforestation programs for the LMW have also been initiated. The spatiotemporal monitoring of landscape pattern (composition and configuration) is needed to inform policy and support forward-looking management planning toward landscape sustainability. However, the changes in the landscape pattern of the LMW, including the extent of forest cover loss and gain over the past decades, have not been quantified; hence, this study. We used remote sensing data (Landsat) to classify the land use/land cover of the LMW in 1988, 2002 and 2016. We subsequently used spatial metrics to quantify the changes in the landscape pattern of the watershed. We found that between 1988 and 2002, a period that largely preceded the start of the LMW's major rehabilitation (c. 1999), the watershed had a net forest cover loss of 259 ha. From 2002 to 2016, it had a net forest cover gain of 557 ha. The detected increase in forest cover was supported by the percent tree cover change analysis results based on MODIS data. The deforestation of the LMW resulted in landscape fragmentation as indicated by the decrease in the area of forest and mean forest patch size, and the increase in forest patch density, etc. Forest restoration activities have helped improve the watershed's landscape connectivity as signified by the increase in the area of forest and mean forest patch size, and the decrease in forest patch density, etc. The results also revealed that rapid urbanization has been a major factor driving landscape changes around the LMW, and this requires proactive, forward-looking management planning. Overall, the LMW's case presents some valuable learning experience and insights regarding public-private partnerships toward watershed and forest-related rehabilitation initiatives. On a national scale, the Philippine government has embarked on a massive national greening program. The findings of this study suggest that such efforts could lead to the enhancement of denuded forest areas, if done properly.

1. Introduction

Watersheds are important sources of various ecosystem services – the benefits that people derive from ecosystems – including provisioning services, regulating services, supporting services, and cultural services (MEA, 2005; TEEB, 2010). Watersheds encompass biological, physical, social, and economic processes; thus, they are often used as units for landscape planning and natural resource management (Steiner, 2008). However, watersheds are also vulnerable to drastic landscape changes due to deforestation (Ziegler et al., 2004; van Noordwijk and Bruijnzeel, 2008; Maina et al., 2013; Qin and Gartner, 2016; Gao and Yu, 2017). Deforestation affects landscape pattern or structure as it can alter landscape composition (e.g. loss or decrease of forest cover) and landscape configuration (i.e. spatial connectivity and fragmentation of landscape elements, e.g. forest patches) (Haddad et al., 2015; Brinck et al., 2017; Reddy et al., 2018; Taubert et al., 2018).

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The effects of landscape fragmentation can be both positive and negative (Ibáñez et al., 2014; Mitchell et al., 2015; Rolo et al., 2018). For instance, in their review on the effects of landscape fragmentation on plants and plant communities, Ibáñez et al. (2014, p. 882) found that 'negative effects of fragmentation due to isolation, edge effects and fragment size were significant; but only edge effects and fragment size had significant positive effects... [and that] positive responses to edge effects were significant for density, fecundity, survival, growth and richness, and significantly negative for density, survival, colonization and richness'. In their study on the effects of fragmentation on carbon stocks, Rolo et al. (2018) found that forest fragmentation can directly reduce aboveground carbon and increase soil organic carbon. However, they also found that through decreasing functional diversity, forest fragmentation can indirectly increase aboveground carbon and decrease soil organic carbon. On their attempt to develop a framework for assessing the relationship between landscape fragmentation and ecosystem services, Mitchell et al. (2015) argued that while fragmentation generally has negative effects on ecosystem service supply, it can also have positive or negative effects on service flow. It is because landscape fragmentation can facilitate, but at the same time, interrupt movement of organisms, matter, energy, and people across landscapes (Mitchell et al., 2015).

More specifically, landscape fragmentation can result in an increased forest patch density and a decrease of mean forest patch size, consequently affecting the habitats for species (Andrén, 1994; Debinski and Holt, 2000; Haddad et al., 2015). The reduction of the spatial connectivity of forest patches leads to habitat fragmentation, consequently affecting biodiversity (Debinski and Holt, 2000; Laurance et al., 2011; Haddad et al., 2015; Taubert et al., 2018). Forest and habitat fragmentation also affect key ecosystem functions by decreasing biomass and altering carbon and nutrient cycles (Haddad et al., 2015; Brinck et al., 2017; Rolo et al., 2018). Deforestation often results in forest fragmentation and extends the forest edge wherein trees suffer increased mortality, substantially contributing to carbon emissions (Laurance et al., 2011; Brinck et al., 2017). The much-reduced areal extent of forest patches due to landscape fragmentation may also affect negatively the role of forests in mitigating hillslope overland flow (Ziegler et al., 2004). Forest fragmentation also exposes more edges to solar radiation, which can result in increased water loss due to transpiration (Gao and Yu, 2017).

Remote sensing technology has been, and continues to be, an important data source for many environmental studies and monitoring activities, including forest cover mapping and change and impact monitoring (Hansen et al., 2013; Shimada et al., 2014; Estoque et al., 2018; Reddy et al., 2018; Taubert et al., 2018). Remote sensing is the art and science of acquiring information about an object without being in direct physical contact with the object – information that is stored in satellite images and aerial photographs. Land use/land cover (LULC) maps derived from remote sensing data can provide valuable information and be of help in capturing landscape pattern, including landscape composition and spatial configuration (Nagendra et al., 2004; Haddad et al., 2015; Estoque and Murayama, 2016; Brinck et al., 2017; Estoque et al., 2017; Taubert et al., 2018).

In the Philippines, forest cover has declined from 90% in 1521 when Spanish colonizers arrived, to 70% in 1900 and then to 22% in 1998 due to deforestation (ESSC, 1999). Among the most important drivers of deforestation in the country are timber harvesting, agricultural expansion, urban growth, and population increase (Kummer, 1992; Liu et al., 1993; ESSC, 1999; Lasco et al., 2013). Driven by the alarming rate of deforestation in the country, the national government and other organizations have worked on the protection, conservation and improvement of the country's remaining forests (Estoque et al., 2018). As a result, numerous national reforestation projects and forest management policies have emerged over the years (Harrison et al., 2004; Lasco et al., 2013). The National Greening Program (NGP) is the most recent reforestation initiative by the Philippine government, with the objective to plant 1.5 billion trees on 1.5 million hectares from 2011 to 2016 (RP, 2011).

The La Mesa Watershed (LMW) is among the many watersheds in the country that have suffered from severe deforestation in the past. It is located in the Philippines' National Capital Region, Metro Manila. At present, it contains a valuable forest cover – the last remaining forest cover of its size in the metropolitan area. With its vegetation cover, the LMW functions as a major carbon sink in the area and serves as the 'lungs' of the highly urbanized metropolis. The LMW is the last ecological frontier of Metro Manila. Characterized and established in 1929, the LMW was envisioned primarily to serve as the main source of water supply for Metro Manila (Lasco and Pulhin, 2006; Malabrigo et al., 2015). With the demand for water in Metro Manila having increased over the years, the LMW has remained a crucial component of the entire network of water sources for the area (Calderon et al., 2006; Dizon et al., 2006; Lasco and Pulhin, 2006; see also http://mwss.gov.ph/learn/metro-manila-water-supply-system/). Since the 1960s, the LMW's whole landscape has undergone remarkable changes due to forest degradation and rehabilitation programs, including forest restoration activities (Lasco and Pulhin, 2006; Calderon et al., 2006; Malabrigo et al., 2015).

Recently, the LMW has gained attention and popularity from the media because of its purported improvement in terms of forest cover, among others, due to major rehabilitation efforts that started in 1999 (see, for example, France-Presse, 2015; GMA, 2015; Buan, 2015; Chan, 2015). However, the extent of forest cover changes in the LMW, i.e. loss and gain, and the changes in the watershed's landscape pattern over the past decades have not been quantified. Hence, this study sought to quantify the forest cover loss and gain and the consequent landscape pattern changes in the LMW over the past three decades (1988–2016) using state-of-the-art technologies, including remote sensing and geographic information systems (GIS). This study can help shed light on such a postulation that the LMW's landscape status has improved over the past decade or so. We argue that the spatiotemporal monitoring of forest cover changes is also needed to inform management planning on the magnitude or extent of the impacts of forest degradation and the various efforts that have been undertaken to restore and rehabilitate denuded areas. In this article, the drivers of forest cover changes in the LMW, including some important challenges on its on-going rehabilitation program, are also discussed.

2. Materials and Methods

2.1. Study site and data used

Geographically, the LMW is located between 14°42′34″ and 14°46′44″ N latitudes and 121°4′1″ and 121°8′4″ E longitudes in the north-eastern part of Metro Manila, Philippines (Fig. 1). The LMW has a total land area of 2659 ha, expanding across the cities of Quezon and Caloocan in Metro Manila and the municipality of Rodriguez in the province of Rizal (RP, 2007), with the greater part of the area located in Quezon City (Fig. 1b). Today, various institutions and agencies are involved in the management of the LMW, including the Metropolitan Waterworks and Sewerage System (MWSS), the Department of Environment and Natural Resources (DENR), the local government of Quezon City, the ABS-CBN Lingkod Kapamilya Foundation, Inc. (ALKFI), the La Mesa Executive Board, the La Mesa Resort Zone Executive Committee, and the Multi-Sectoral Watershed Management Council (MSWMC).

In terms of topography, the LMW has a gently undulating terrain, with elevations ranging from 46 to 256 m above sea level and slopes below 25% (Malabrigo et al., 2015). The climate in the area belongs to the Philippines' Climatic Type 1, characterized by two pronounced seasons: a dry season from November to April and a wet season during the rest of the year. The mean annual rainfall at the LMW from 1978 to 2007 was around 2515 mm, while the average monthly minimum and maximum temperatures were about 23 °C and 32 °C, respectively (Malabrigo et al., 2015).

We used remote sensing satellite imagery (Landsat; https://glovis.usgs.gov/) (Fig. 1c) to map the LULC of the area in 1988, 2002, and 2016. The 1988–2002 and 2002–2016 time periods were decided based on two

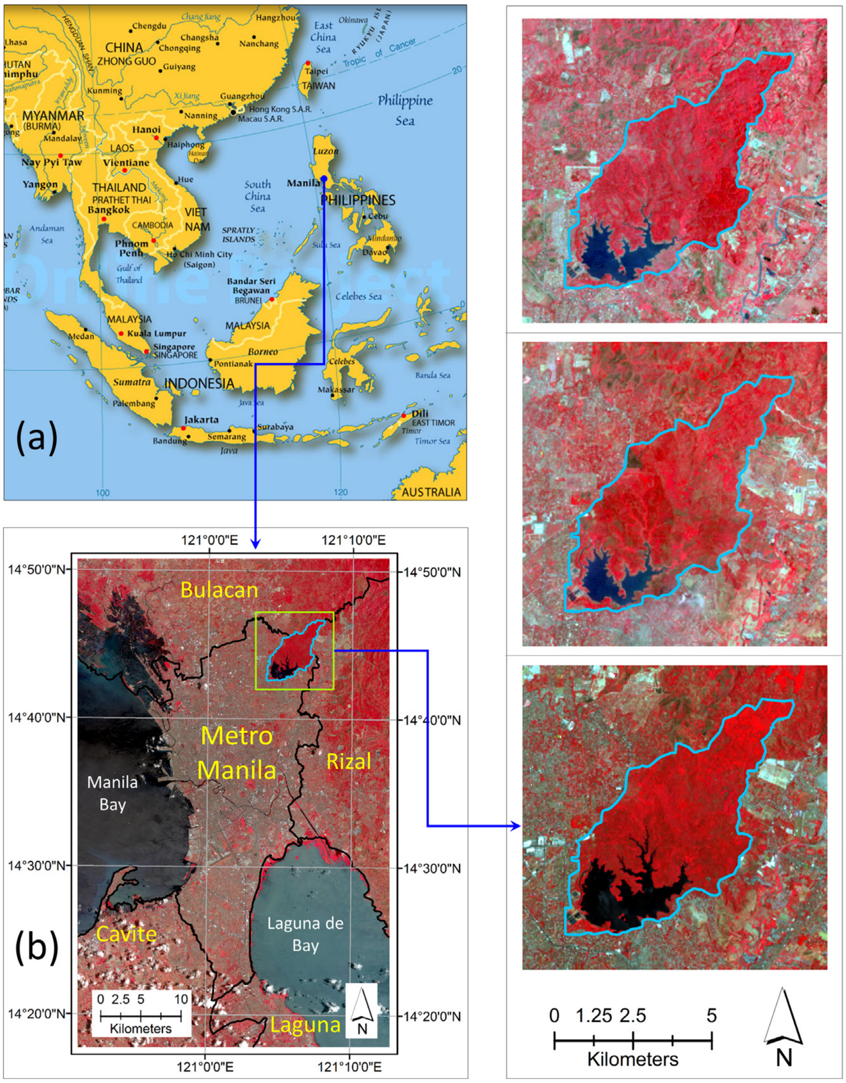

Fig. 1. Location of the study site, the La Mesa Watershed (LMW) and its surrounding areas, Metro Manila, Philippines. (a) Map of Southeast Asia showing the location of Metro Manila, Philippines (Map source: www.nationsonline.org); (b) Landsat 8 image of Metro Manila and its adjacent provinces; and (c) Landsat imagery used in this study showing the boundary of the LMW in light blue line. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

considerations: (i) in order to capture and compare the spatiotemporal landscape change dynamics in the LMW before and after its major rehabilitation that started in 1999; and (ii) availability of satellite imagery. The use of a 1999 image instead of the 2002 image would have pinpointed the starting year of the LMW's major rehabilitation. However, due to the unavailability of satellite imagery, the 2002 image was used instead. The two time periods have the same time extent of 14 years.

2.2. LULC change detection and analysis

Prior to LULC mapping, all the satellite images were subjected to a set of pre-processing procedures. More specifically, the digital number or DN values of the multispectral bands of the satellite images from the three different sensors (TM, ETM+, OLI/TIRS) had to be converted into surface reflectance values. Atmospheric correction was also performed using the dark object subtraction model to remove any atmospheric effects/interferences due to absorption and scattering. To accomplish all these pre-processing procedures, we used the 'Landsat archive import' module available in TerrSet, a geospatial monitoring and modeling software (https://clarklabs.org/terrset/).

The subset of the study area (Fig. 1c) was first clipped from the three pre-processed satellite images before classification. We used the maximum likelihood supervised classification technique to classify and map the LULC of the study site. This technique involves digitizing of training sites for each LULC class based on 'a priori knowledge' and using these training sites to train and eventually classify the pixels in the images. This method is commonly used in remote sensing-based LULC mapping (Thapa and Murayama, 2009; Rozenstein and Karnieli, 2011; Estoque and Murayama, 2016).

We classified six LULC classes, namely built-up, forest, grassland (including shrubs), cropland/bareland, water, and other. There were spectral confusions in some areas between built-up, cropland and bareland. We did our best to separate built-up from the other classes. Cropland and bareland were later combined into one class (cropland/bareland). In another study, it has also been observed that some subclasses of artificial surface (built-up), bareland and cropland have very similar spectral signatures (Chen et al., 2015). All the vegetated areas that were not classified as forest or cropland were classified as grassland (including shrubs). The 'other' class includes all other lands that do not fall into the other five classes. This class is dominated by burned areas, thus in this article it was labelled 'Other (burned areas)'.

The respective accuracies of the classified LULC maps were assessed using three sets (one set for each map) of 480 reference points generated through stratified random sampling. Our decision to consider this total number of reference points was based on previous studies. For instance, Zhou et al. (2014) used 256 reference points to assess the accuracy of their classified map containing six classes spread across a 1.47 million ha watershed landscape. In another study, Estoque and Murayama (2013) used 312 reference points to assess the accuracy of their classified map containing four classes in a 5700 ha study area. In comparison with these studies, our study used much higher density of points per unit area.

In the sampling scheme that we employed, i.e. 'stratified random', by using the spatial sampling module available in TerrSet, called 'SAMPLE', the input image (each of the LULC maps, in our case) was first divided (internally) into a rectangular matrix. The module then randomly chose a pixel within each matrix cell. The module did not consider the LULC classes when running its calculations, but instead assumed that pixels that are closer to one another are more similar than those farther away. This is in line with Tobler's first law of geography, i.e. "everything is related to everything else, but near things are more related than distant things" (Tobler, 1970, p. 236). Furthermore, the module did not determine the number of points for each class, but rather the number of points for the entire image based on the specified number of points and the rectangular matrix (TerrSet Help System).

For the 2002 and 2016 maps, Google Earth and the pan-sharpened Landsat images (Du et al., 2014; Estoque and Murayama, 2015a) were used as references during the assessment. Due to the lack of reference data for the 1988 map, we relied on our knowledge of the study area and visual interpretation of the original image, with the aid of Google Earth images especially for the areas whose cover had persisted over the years.

We detected the changes in the extent of each LULC class across the years 1988, 2002 and 2016 by calculating the area of each class for each time point. In addition, we overlaid (cross-tabulated) the 1988 LULC map with the 2002 LULC map, and the 2002 LULC map with the 2016 LULC map to determine the total area and location of the pixels of a particular class that transitioned to another class during the two time periods. In this particular analysis, we focused on the transitions that resulted in forest cover gains and losses.

2.3. Percent tree cover change analysis (2002–2016)

To complement the forest cover change detection between 2002 and 2016 based on Landsat data, we determined the changes in percent tree cover (PTC) over the same period. For this purpose, we used the PTC data derived from Moderate Resolution Imaging Spectroradiometer (MODIS) images in 2002 and 2016 (MOD44B V006) (Dimiceli et al., 2015).

The MOD44B V006 images were originally projected to the global sinusoidal projection system and had a spatial resolution of approximately 232 m in the study area. As part of our pre-processing procedure, we re-projected the images to WGS84 UTM 51N to be consistent with the Landsat data, but keeping their original spatial resolution. This process enabled us to clip the images with the extent of our study area and perform a change analysis. The change in PTC ($\triangle$ PTC) in each MODIS pixel was calculated by subtracting the 2002 PTC map from the 2016 PTC map.

2.4. Landscape connectivity and fragmentation analysis

The use of spatial metrics enables one to capture the composition and spatial configurations of landscapes and their patch elements at a certain point in time (Wu et al., 2011; McGarigal et al., 2012). To monitor the changes in the spatial configuration (connectivity and fragmentation) of the LULC classes (e.g. forest class) and the overall changes in the landscape pattern of the LMW, we used five class-level and three landscape-level metrics, respectively (Table 1). The class-level metrics included the percentage of landscape (PLAND), patch density (PD), mean patch area (AREA_MN), area-weighted mean fractal dimension index (FRAC_AM), and mean Euclidean nearest neighbor distance (ENN_MN). The landscape-level spatial metrics included the contagion (CONTAG), landscape shape index (LSI), and Shannon's diversity index (SHDI).

The class-level metrics were selected based upon the spatial features of the LMW's LULC classes we aimed to characterize, namely area or size (PLAND, AREA_MN), shape complexity (FRAC_AM), and fragmentation and connectivity (PD, ENN_MN). The landscape-level metrics aimed to capture patch aggregation (CONTAG), shape complexity (LSI) and patch types diversity (SHDI) in the LMW itself. These metrics are among the most commonly used metrics in landscape-related fields, such as landscape ecology, land change science, landscape sustainability science, and landscape and urban planning (e.g. Wu et al., 2011; Estoque and Murayama, 2016). All the metrics were calculated using FRAGSTATS v4.2 based on the eight-cell neighbor rule (McGarigal et al., 2012).

3. Results

3.1. LULC classification accuracy

The classified LULC maps of the study area are given in Fig. 3, while the accuracy assessment results are given in Table 2. The producer's accuracy row in Table 2 'indicates the probability of a reference pixel being correctly classified', and thus also quantifies omission error (Congalton, 1991, p. 36–37). On the other hand, the user's accuracy column is 'indicative of the probability that a pixel classified on the map/image actually represents that category on the ground', and is a measure of commission error (Congalton, 1991, p. 37). To illustrate, in the 2016 confusion matrix (Table 2c), there were 86 forest reference

Table 1
The spatial metrics used.

Metric	Description	Unit/Value range
Class-level		
Percentage of Landscape (PLAND)	Proportion of the landscape occupied by a particular patch type	%
Patch Density (PD)	Equals the number of patches of a particular patch type divided by the total landscape area	number per km^2 (or per 100 ha)
Mean Patch Area (AREA_MN)	Equals the average area of all patches of a particular patch type	ha
Area-Weighted Mean Fractal Dimension Index (FRAC_AM)	The patch fractal dimension weighted by relative patch area which measures the average shape complexity of individual patches for the whole landscape or a specific patch type	≥ 1; ≤ 2
Mean Euclidean Nearest Neighbor Distance (ENN_MN)	Equals the distance to the nearest neighboring patch of the same type, based on shortest edge-to-edge distance	m
Landscape-level		
Contagion (CONTAG)	An information theory-based index that measures the extent to which patches are spatially aggregated in a landscape	0–100
Landscape Shape Index (LSI)	A modified perimeter-area ratio of the form that measures the shape complexity of the whole landscape. It is also a measure of patch aggregation or disaggregation	≥ 1; without limit
Shannon's Diversity Index (SHDI)	A measure of the diversity of patch types in a landscape that is determined by both the number of different patch types and the proportional distribution of area among patch types	≥ 0; without limit

Source: Wu et al. (2011), McGarigal et al. (2012).

Table 2
LULC classification confusion matrix.

Classified Data	Reference Data							
	Built-up	Forest	Grassland	Cropland/Bareland	Water	Other (burned areas)	Total	User's accuracy (%)
(a) 1988								
Built-up	25	0	2	4	0	0	31	80.65
Forest	0	80	8	0	1	0	89	89.89
Grassland	3	7	141	10	0	0	161	87.58
Cropland/Bareland	6	0	7	84	0	1	98	85.71
Water	0	0	0	0	18	0	18	100.00
Other (burned areas)	0	0	3	5	0	75	83	90.36
Total	34	87	161	103	19	76	480	
Producer's accuracy (%)	73.53	91.95	87.58	81.55	94.74	98.68		
Overall accuracy (%) = 88.13; Overall kappa = 0.8471								
(b) 2002								
Built-up	74	0	7	8	0	0	89	83.15
Forest	1	78	6	0	1	0	86	90.70
Grassland	2	8	183	7	1	1	202	90.59
Cropland/Bareland	4	0	10	54	0	2	70	77.14
Water	0	0	0	0	6	0	6	100.00
Other (burned areas)	0	0	1	3	0	23	27	85.19
Total	81	86	207	72	8	26	480	
Producer's accuracy (%)	91.36	90.70	88.41	75.00	75.00	88.46		
Overall accuracy (%) = 87.08; Overall kappa = 0.8231								
(c) 2016								
Built-up	120	1	4	6	0	1	132	90.91
Forest	0	94	10	0	1	0	105	89.52
Grassland	4	5	116	9	1	1	136	85.29
Cropland/Bareland	3	0	5	66	0	1	75	88.00
Water	0	0	0	0	22	0	22	100.00
Other (burned areas)	1	0	0	1	1	7	10	70.00
Total	128	100	135	82	25	10	480	
Producer's accuracy (%)	93.75	94.00	85.93	80.49	88.00	70.00		
Overall accuracy (%) = 88.54; Overall kappa = 0.8516								

pixels, and of this number, eight were omission errors because they were not correctly classified as forest. In the same matrix and year, there were 105 classified forest pixels, and of this number, 11 were commission errors because these pixels were not forest as per reference data.

In our classifications, most of the confusions occurred between built-up, cropland/bareland and grassland. There was also a substantial amount of confusion between forest and grassland (including shrubs) and between cropland/bareland and grassland. Nevertheless, the assessment shows that the overall accuracy of each of the classified LULC maps is above the widely recognized minimum level of accuracy for thematic mapping from remotely sensed imagery, which is 85% overall (more on this in Section 4.3). The classified LULC maps had an individual overall accuracy of at least 87% and individual overall kappa of at least 0.82 (Table 2).

3.2. Landscape changes: forest cover gains and losses

The results show that the LMW and its surrounding areas had undergone dramatic landscape transformations over the past three decades (1988–2016) (Fig. 2). The forest cover of the LMW declined from 54.95% in 1988 to 45.22% in 2002, and increased to 66.16% in 2016. The decrease and increase, respectively, translate to a net forest cover loss of 258.68 ha from 1988 to 2002 and to a net forest cover gain of 556.74 ha from 2002 to 2016.

The results also show that rapid urbanization was a major factor driving landscape changes in the areas surrounding the LMW (Fig. 2). The area of built-up increased from 6.91% in 1988 to 19.72% and 27.67% in 2002 and 2016, respectively. From 1988 to 2002, built-up lands expanded at the rate of 85.86 ha per year, while during the 2002–2016 period it expanded at the rate of 53.36 ha per year. This temporal non-stationarity of the intensity of built-up land expansion is consistent with some previous findings for Metro Manila (Estoque and Murayama, 2015b).

The results also revealed that the extent of the 'water' class inside the LMW decreased and increased over the years (Fig. 2). The changes in the surface area of water inside the LMW, more precisely the La Mesa Dam (reservoir), could have been due to a number of factors, including the possible fluctuation in the volume of water fed by the Angat Dam and Ipo Dam for treatment and distribution. However, due to lack of available and reliable information, we could not verify this issue. Today, these three dams and watersheds (Angat, Ipo, and La Mesa) are the main sources of water for Metro Manila (http://mwss.gov.ph/learn/metro-manila-water-supply-system/), with the Angat Dam providing more than 90% of Metro Manila's potable water supply (Feliciano, 2014; Pulhin et al., 2018). Unfortunately, there is no

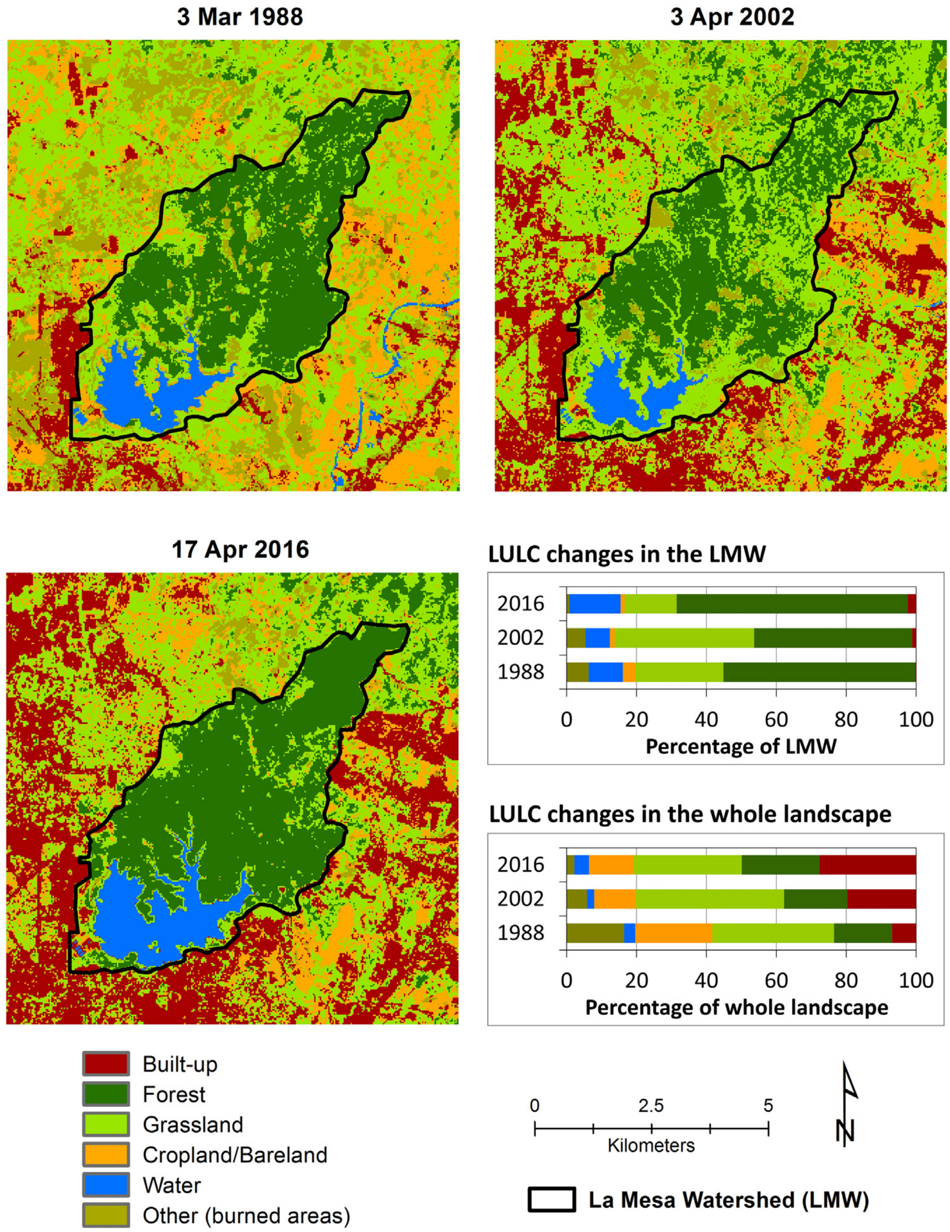

Fig. 2. LULC maps (and statistics) of the LMW and its surrounding areas. Note: 'whole landscape' refers to the whole area of the rectangular maps enclosing the LMW and its surrounding areas.

available information on the temporal changes in the contribution of the LMW.

Fig. 3 shows the spatial distribution and details of forest cover losses and gains in the LMW across the two time periods. Of the 252.75 ha gross forest cover gain during the 1988–2002 period, 80.11% came from grassland (Fig. 3a). On the other hand, 86.71% of the 511.43 ha gross forest cover loss was due to forest-grassland transition. During the 2002–2016 period, the LMW had a gross forest cover gain of 650.79 ha and 87.19% of which came from grassland (Fig. 3b). Of the 94.05 ha gross forest cover loss, 89.01% was also due to forest-grassland transition. These results show that grassland had been the most active LULC class for forest cover change, be it as a gaining category or as a losing category.

3.3. Changes in PTC

Fig. 4 presents the results of PTC change analysis based on MODIS data. At the spatial resolution of the MODIS data (c. 232 m), the entire LMW, excluding the water surface area, is composed of 386 pixels. Of this number, 77% experienced an increase in their PTC, while 20% experienced a decrease. The remaining 2% showed no change. The bar graph in Fig. 4b shows the △ PTC per individual pixel. The average PTC increase was 14.1%, while the average PTC decrease was 9.5%. Fig. 4c presents a frequency distribution per 5% interval of △ PTC. Overall, the results are consistent with the detected forest cover changes based on the Landsat images over the same period. There were indications of deforestation, but the

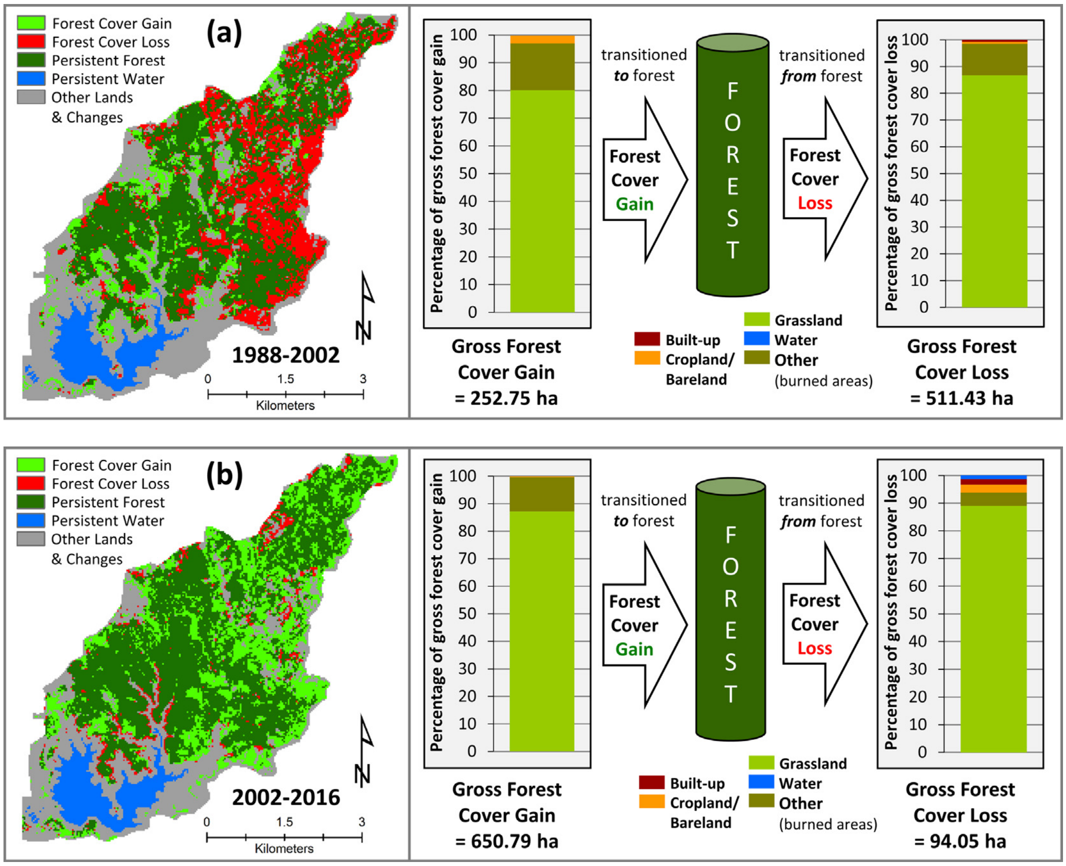

Fig. 3. Forest cover gains and losses in the LMW. (a) 1988–2002, and (b) 2002–2016.

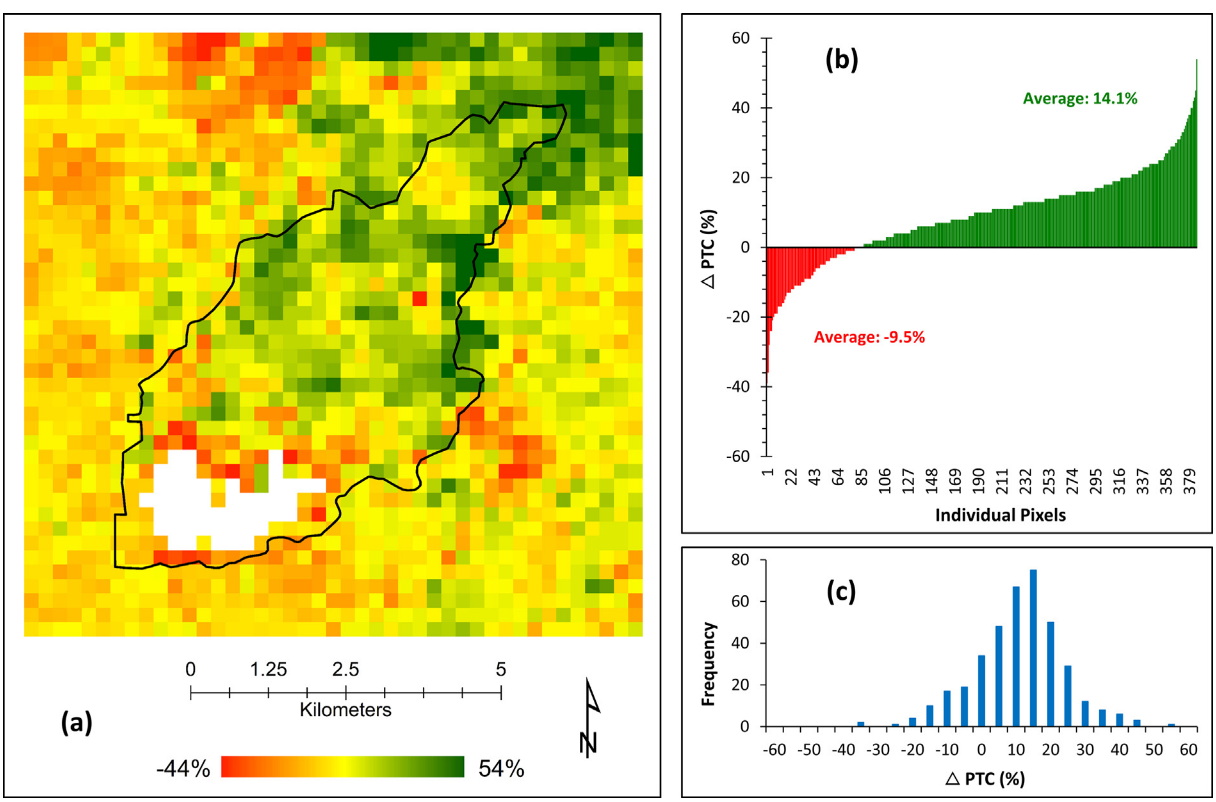

Fig. 4. Change in PTC in the LMW based on MODIS data (2002–2016). (a) Δ PTC map; (b) Δ PTC per pixel, where pixels are numbered and displayed according to Δ PTC; and (c) Frequency distribution per 5% interval of Δ PTC. Positive Δ PTC indicates net tree cover gain, while negative Δ PTC indicates net tree cover loss.

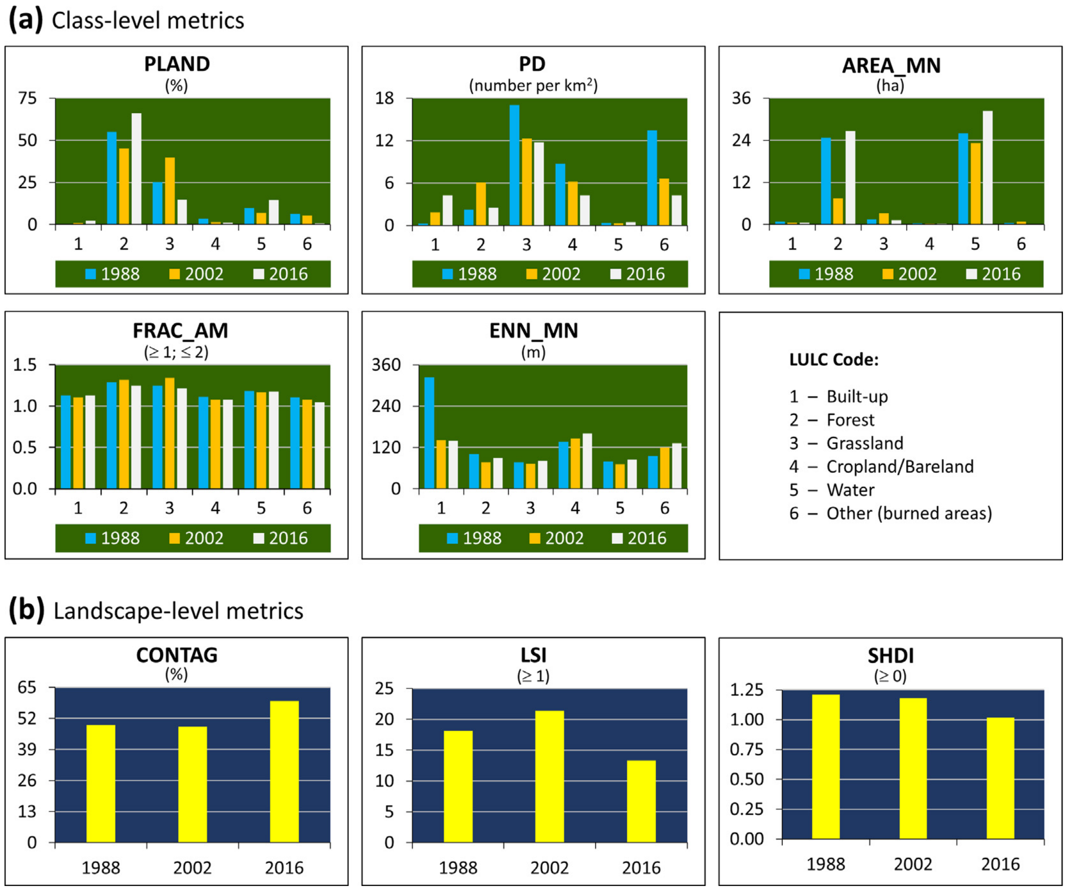

Fig. 5. Values of the spatial metrics used for examining the landscape connectivity and fragmentation of the LMW (excluding surrounding areas). Note: PLAND – percentage of landscape; PD – patch density; AREA_MN – mean patch area; FRAC_AM – area-weighted mean fractal dimension index; ENN_MN – mean Euclidean nearest neighbor distance; CONTAG – contagion; LSI – landscape shape index; and SHDI – Shannon's diversity index.

quantity of forest (tree) cover loss was outweighed by the quantity of forest (tree) cover gain.

3.4. Landscape connectivity and fragmentation

At the class level, the landscape pattern analysis revealed that during the 1988–2002 period, the LMW's forest cover became more fragmented and complex as indicated by the increase of PD and FRAC_AM and the decrease of AREA_AM and ENN_MN (Fig. 5a). ENN_MN decreases when (i) a large patch of a particular LULC class, located far from other patches of the same type, is broken down into several smaller ones; and (ii) a new patch of the same type is developed in between existing patches of the same type. The loss and gain of forest cover during this period are indicative of these two factors, though the former appears to be the stronger case for the LMW's forest cover. During the 2002–2016 period, the trends in the values of the class-level spatial metrics indicate that the LMW's forest cover became more aggregated or connected and less complex (Fig. 5a).

At the landscape level, our results show that from 1988 to 2002, the whole watershed became more fragmented and complex, as indicated by the decrease of CONTAG and the increase of LSI (Fig. 5b). By contrast, during the 2002–2016 period, the whole watershed became more aggregated, more connected, and less complex as signified by the increase of CONTAG and the decrease of LSI. The decrease in the area of water and cropland/bareland during the 1988–2002 period and the subsequent increase of forest cover and decrease of grassland during the 2002–2016 period resulted in SHDI's decreasing trend. All these findings are consistent with the degradation and improvement of forest cover in the LMW during the 1988–2002 and 2002–2016 periods, respectively.

4. Discussion

4.1. Changes in forest cover and landscape pattern

Our study captured some important indications of the impacts of the two land change processes that occurred in the LMW: forest degradation which resulted in a much greater forest cover loss during the 1988–2002 period and forest restoration/rehabilitation which resulted in a much greater forest cover gain during the 2002–2016 period (Figs. 2 and 3). During the 2002–2016 period, signs of deforestation (i.e. aside from those caused by the expansion of the water surface area) were also detected especially in the northern and eastern parts of the LMW (Fig. 3b). Nevertheless, the observed loss of forest cover during this period was much less than the observed gain, indicative of the overall positive impact of forest restoration activities, which was also supported by the result of the PTC change analysis (Fig. 4). Our results are also consistent with some other studies which observed that some portions of the LMW were still covered with natural vegetation with sparsely distributed primary and secondary growth vegetation, while most of the denuded parts had been reforested and converted into a young secondary forest (Malabrigo et al., 2015). In Fig. 3, these are respectively labelled as 'persistent forest' and 'forest gain'. The substantial increase of forest cover in the LMW during the second time period of the analysis can have a positive impact because most of the ecosystem services that watersheds generate and provide to people depend on forest cover (Kaiser and Roumasset, 2002; Lele, 2009; Locatelli and Vignola, 2009; Chen et al., 2015; Yi et al., 2017).

The spatial metrics used in this study were useful in determining whether the LMW's landscape had become more fragmented or more aggregated/connected over the years. At both levels (class and landscape), the spatial metrics indicated that LMW's forest cover and whole landscape became more fragmented or disaggregated during the 1988–2002 period and more aggregated or connected during the 2002–2016 period (Fig. 5). The detected landscape fragmentation or disaggregation between 1988 and 2002 was due to deforestation, resulting in much smaller forest patches, higher forest patch density, and more complex, less connected forest patches (Fig. 5a). By contrast, the detected landscape aggregation or connectivity between 2002 and 2016 was due to forest restoration, resulting in much larger forest patches, lower forest patch density, and more connected, less complex forest patches (Fig. 5a).

The much-improved condition of the LMW's vegetation cover today (Figs. 2-5) can further enhance its biodiversity and the provision and delivery of its ecosystem services. Ideally, detailed past-to-present data on flora and fauna are needed to aid the impact analysis of the restoration and rehabilitation activities. However, these types of data are not available for the LMW, and this constrained us in our analysis. Nevertheless, the LMW is now home to 520 plant species, of which 10 are vulnerable, seven are endangered, and four are critically endangered according to the International Union for Conservation of Nature (IUCN) (MWCI, 2012). From a mere nine species of trees that were planted in the mid-1970s to the mid-1980s (Chan, 2015), the LMW is now covered with at least 70 native tree species (Malabrigo et al., 2015; Chan, 2015 reports 99 indigenous species of forest trees). A total of 117 species of vertebrate wildlife are also found in the LMW (MWCI, 2012). From 31 bird species recorded in 2002, the LMW is now home to 120 species of birds (ALKFI, 2014; Chan, 2015), including the Osprey (*Pandion haliaetus* Linn.), a bird that is an uncommon migrant species listed under the Convention on International Trade in Endangered Species (CITES) (MWCI, 2012).

4.2. Drivers of forest cover changes in the LMW and future challenges

To better understand what could have possibly caused the spatio-temporal changes in the landscape pattern of the LMW, we traced some important major historical events concerning the watershed. Previous studies have reported that in the 1960s and 1970s, the LMW experienced severe land degradation (Tiburan et al., 2012; Malabrigo et al., 2015). Consequently, the deforestation of the LMW resulted in the conversion of many parts of a large natural forest into grassland (de Asis and Omasa, 2007). Our results, especially for the 1988–2002 period, are consistent with this previous observation, i.e. the transitioning of forest to grassland (Figs. 2 and 3). The presence of informal settlers and their slash and burn activities (kaingin system or swidden farming) have been considered as the major cause of the denudation of the watershed's forest cover (Lasco and Pulhin, 2006; de Asis and Omasa, 2007; Tiburan et al., 2012; Malabrigo et al., 2015). According to ALKFI (2014), by 1999, a large portion of the LMW were still occupied by informal settlers who were engaged in slash and burn farming and timber poaching. The abandonment of slashed and burned areas after some time due to nutrient depletion might have resulted in the increase of grassland. It can be noted that the second most dominant LULC class inside the watershed as shown by the remote sensing-derived LULC maps was grassland (Figs. 2 and 3).

Recognizing the high importance of the LMW, the Manila Seedling Bank Foundation, Inc. (MSBFI), a non-governmental organization founded in 1977, undertook reforestation activities in the area from 1978 to 1983 (Malabrigo et al., 2015). However, when MSBFI left the LMW, illegal loggers exploited the area (Malabrigo et al., 2015). Thus in 1999, under a public–private partnership, the Philippine government had to start another rehabilitation program for the LMW, this time through the assistance of the Bantay Kalikasan (Nature Watch) of the ABS-CBN Lingkod Kapamilya Foundation, Inc. (ALKFI), formerly ABS-CBN Foundation, Inc. (Lasco and Pulhin, 2006; Tiburan et al., 2012; ALKFI, 2014). As of this writing, the LMW is still being managed by Bantay Kalikasan, in cooperation with the Metropolitan Waterworks and Sewerage System (MWSS), the local government of Quezon City, and other institutions and agencies (Section 2.1).

The landscape change analysis during the 1988–2002 period (Figs. 2 and 3a) provides indications of the negative consequence of the Philippine government's failure to continue the rehabilitation program for the LMW soon after the MSBFI left. But even during the 2002–2016 period, as mentioned above, there were still signs of deforestation (Fig. 3b), and this needs to be taken into account in the continuing rehabilitation program for the LMW. Nevertheless, the much higher observed forest cover gain during this period is indicative of the overall positive impact of the Bantay Kalikasan rehabilitation program.

The case of the LMW is an example that shows that the real success of a watershed rehabilitation program depends on the 'continuity' of the program. Short-term goals are important, but a long-term goal is necessary. In general, the results of this study provide indications of the improving condition of the LMW since Bantay Kalikasan took over. As of 2014, a total of 1552 ha have been reforested, a total of 52 km of roads and trails are currently maintained, and the incidence of illegal squatting or settling has been contained (ALKFI, 2014). In addition, the ALKFI has also developed a 33-hectare eco-park within the LMW, called the LA MESA ECOPARK, which aims to provide the general public nature appreciation, recreation, and learning opportunities while generating resources for the rehabilitation of the watershed (ALKFI, 2014).

However, another important challenge is emerging for the LMW and its vanguards, and that is the rapid urbanization of the watershed's surrounding areas (Fig. 2). Owing to its relatively small area, Metro Manila is already fully urbanized, with high population density and limited urban green spaces remaining (Estoque and Murayama, 2015b; Estoque et al., 2017). The expansion of Metro Manila's urban area is limited to the north-south direction only due to Manila Bay on the west and Laguna de Bay on the south-eastern side (Fig. 1b). Because of this, the areas surrounding the LMW located in the north-eastern part of the metropolis have been rapidly urbanized (Fig. 2). We suggest that this current pattern of landscape changes around the LMW must also be given consideration in its rehabilitation program. Where possible, a forest buffer zone outside LMW's boundary must be established.

4.3. Methodology-related discussion

We recognize that the results of our landscape pattern analysis were sensitive to the input LULC maps which were not free of classification errors (Table 2). Thus, the potential limitations of the landscape pattern analysis should be considered whenever the results are used. Medium to moderate spatial resolution satellite images, such as those used in this study, are among the most commonly used for regional land change studies. However, these types of satellite images also have some limitations, including the mixed pixel problem (Lu and Weng, 2004; Xie et al., 2008).

In remote sensing, the reflectance value of a mixed pixel is the average of the reflectance values of several ground cover types. This means that in a pixel-based classification, like the one employed in this study, a pixel that has been classified as forest or non-forest may not be 100% forest or non-forest on the ground as there could be parts of the mixed pixel with or without trees. Thus, there could be some errors in the detected forest cover changes due to this factor. Image interpretation during the collection of training sites could be another source of error in the LULC classification.

Nevertheless, the quality and accuracy levels of the derived LULC maps (i.e. $> 87\%$ overall and $\geq 70\%$ per class) (Table 2) generally exceed the widely recognized minimum level of accuracy for remotely sensed classified LULC maps of 85% overall (Anderson et al., 1976; Thomlinson et al., 1999; Foody, 2002, 2008) and 70% per class (Thomlinson et al., 1999). That said, we also recognize that this 85% minimum level for an overall accuracy has been criticized (Congalton and Green, 1999; Foody, 2008; Pontius and Millones, 2011).

5. Conclusions

In this study, we quantified the forest cover loss and gain and the consequent landscape pattern changes in the LMW over the past three decades (1988–2016) using remote sensing data and spatial metrics. Between 1988 and 2002, a period that largely preceded the start of the LMW's major rehabilitation in 1999, the watershed had a net forest cover loss of 259 ha. From 2002 to 2016, it had a net forest cover gain of 557 ha. The result of the PTC change analysis supports the detected forest cover gains during the 2002–2016 period. The deforestation of the LMW, primarily due to slash and burn farming and timber poaching, resulted in landscape fragmentation, while forest restoration activities resulted in better landscape connectivity. Rapid urbanization has been a major factor driving landscape changes around the LMW, and this must also be taken into account in its (still on-going) rehabilitation program.

It is important to sustain the rehabilitation, conservation, and improvement of the LMW. A continuous monitoring of land changes in the LMW is also needed to keep management planning up to date. Since this type of study is new for the LMW, it can be used as a reference for other similar studies and remote sensing-based monitoring activities in the future. In general, the LMW's case presents some valuable learning experience and insights regarding public-private partnerships toward watershed and forest-related rehabilitation initiatives. We support the notion that the Philippine government's partnership with Bantay Kalikasan can be used as a model for other forestry and watershed-related rehabilitation programs in the country given the enormous task of protecting and conserving the country's remaining forests with the very limited resources of the government (Malabrigo et al., 2015). On a national scale, the Philippine government has embarked on a massive national greening program. The findings of this study suggest that such efforts could lead to the rehabilitation or restoration of denuded forest areas, if done properly.

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References

ALKFI (ABS-CBN Lingkok Kapamilya Foundation, Inc.), 2014. Accomplishment Report 2014. ABS-CBN Corporation, Quezon City, Philippines.
Anderson, J., Hardy, E.E., Roach, J.T., Witmer, R.E., 1976. A land use and land cover classification system for use with remote sensor data. USGS Professional Paper 964, Sioux Falls, SD.
Andrén, H., 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: a review. Oikos 71, 355–366.
Brinck, K., Fischer, R., Groeneveld, J., Lehmann, S., De Paula, M.D., Putz, S., et al., 2017. High resolution analysis of tropical forest fragmentation and its impact on the global carbon cycle. Nat. Commun. 8, 14855.
Buan, L.N., 2015. After 16 years: reforestation of QC watershed now 90-percent done. Available at: < http://www.gmanetwork.com/news/news/metro/525906/after-16-years-reforestation-of-qc-watershed-now-90-percent-done/story/ > (retrieved 24 July 2018).
Calderon, M.M., Camacho, L.D., Carandang, M.G., Dizon, J.T., Rebugio, L.L., Tolentino, N.L., 2006. Willingness to pay for improved watershed management: evidence from Metro Manila, Philippines. For. Sci. Technol. 2, 42–50.
Chan, J., 2015. The birds of La Mesa. Available at: < https://corporate.abs-cbn.com/lingkodkapamilya/bantay-kalikasan/article/pid-1473299148918/the-birds-of-la-mesa/ > (retrieved 24 July 2018).
Chen, M., Lu, Y., Ling, L., Wan, Y., Luo, Z., Huang, H., 2015. Drivers of changes in ecosystem service values in Ganjiang upstream watershed. Land Use Pol. 47, 247–252.
Congalton, R.G., 1991. A review of assessing the accuracy of classifications of remotely sensed data. Rem. Sens. Environ. 37, 35–46.
Congalton, R.G., Green, K., 1999. Assessing the Accuracy of Remotely Sensed Data: Principles and Practices. Lewis Publishers, Boca Raton.
de Asis, A.M., Omasa, K., 2007. Estimation of vegetation parameter for modeling soil erosion using linear spectral mixture analysis of Landsat ETM data. ISPRS J. Photogramm. Rem. Sens. 62, 309–324.
Debinski, D.M., Holt, R.D., 2000. A survey and overview of habitat fragmentation experiments. Conserv. Biol. 14, 342–355.
Dimiceli, C., Carroll, M., Sohlberg, R., Kim, D.H., Kelly, M., Townshend, J.R.G., 2015. MOD44B MODIS/Terra Vegetation Continuous Fields Yearly L3 Global 250m SIN Grid V006. NASA EOSDIS Land Processes DAAC.
Dizon, J.T., Calderon, M.M., Camacho, L.D., Carandang, M.G., Rebugio, L.L., Tolentino, N.M., 2006. Institutionalization of a water user fee for the watershed management. For. Sci. Technol. 2, 51–56.
Du, Z., Li, W., Zhou, D., Tian, L., Ling, F., Wang, H., Gui, Y., Sun, B., 2014. Analysis of Landsat-8 OLI imagery for land surface water mapping. Rem. Sens. Lett. 5, 672–681.
ESSC (Environmental Science for Social Change, Inc.), 1999. Decline of the Philippine Forest. ESSC, Quezon City, Philippines.
Estoque, R.C., Murayama, Y., 2013. Landscape pattern and ecosystem service value changes: implications for environmental sustainability planning for the rapidly urbanizing summer capital of the Philippines. Landscape Urban Plann. 116, 60–72.
Estoque, R.C., Murayama, Y., 2015a. Classification and change detection of built-up lands from Landsat-7 ETM+ and Landsat-8 OLI/TIRS imageries: a comparative assessment of various spectral indices. Ecol. Ind. 56, 205–217.
Estoque, R.C., Murayama, Y., 2015b. Intensity and spatial pattern of urban land changes in the megacities of Southeast Asia. Land Use Pol. 48, 213–222.
Estoque, R.C., Murayama, Y., 2016. Quantifying landscape pattern and ecosystem service value changes in four rapidly urbanizing hill stations of Southeast Asia. Landscape Ecol. 31, 1481–1507.
Estoque, R.C., Murayama, Y., Myint, S.W., 2017. Effects of landscape composition and pattern on land surface temperature: an urban heat island study in the megacities of Southeast Asia. Sci. Total Environ. 577, 349–359.
Estoque, R.C., Pontius Jr., R.G., Murayama, Y., Hou, H., Thapa, R.B., Lasco, R.D., Villar, M.A., 2018. Simultaneous comparison and assessment of eight remotely sensed maps of Philippine forests. Int. J. Appl. Earth Obs. Geoinform. 67, 123–134.
Feliciano, C.M.C., 2014. Gov't races to boost water supply for Metro Manila. Available at: < https://ppp.gov.ph/?in_the_news=govt-races-to-boost-water-supply-for-metro-manila > (retrieved 24 July 2018).
Foody, G.M., 2002. Status of land cover classification accuracy assessment. Rem. Sens. Environ. 80, 185–201.
Foody, G.M., 2008. Harshness in image classification accuracy assessment. Int. J. Rem. Sens. 29, 3137–3158.
France-Presse, A., 2015. Forest paradise re-emerges in the Philippine capital. Available at: < https://www.rappler.com/science-nature/environment/91004-la-mesa-watershed-philippine-capital > (retrieved 24 July 2018).
Gao, Q., Yu, M., 2017. Reforestation-induced changes of landscape composition and configuration modulate freshwater supply and flooding risk of tropical watersheds. PLoS One 12, e0181315.
GMA, 2015. SONA: La Mesa Watershed, binuhay muli sa pamamagitan ng 16 na taong reforestation [English: State of the Nation (SONA): La Mesa Watershed, revived through 16 years of reforestation]. Available at: < https://www.youtube.com/watch?v=bMYZVzPcTKI > (retrieved 24 July 2018).
Haddad, N.M., Brudvig, L.A., Clobert, J., Davies, K.F., Gonzalez, A., Holt, R.D., et al., 2015. Habitat fragmentation and its lasting impact on Earth's ecosystems. Sci. Adv. 1, e1500052.
Hansen, M.C., Potapov, P.V., Moore, R., Hancher, M., Turubanova, S.A., Tyukavina, A., et al., 2013. High-resolution global maps of 21st-century forest cover change. Science 342, 850–853.
Harrison, S., Emtage, N.F., Nasayao, E.E., 2004. Past and present forestry support programs in the Philippines, and lessons for the future. Small-scale For. Econ., Manage. Pol. 3, 303–317.
Ibáñez, I., Katz, D.S.W., Peltier, D., Wolf, S.M., Barrie, B.T.C., 2014. Assessing the integrated effects of landscape fragmentation on plants and plant communities: the challenge of multiprocess–multiresponse dynamics. J. Ecol. 102, 882–895.
Kaiser, B., Roumasset, J., 2002. Valuing indirect ecosystem services: the case of tropical watersheds. Environ. Dev. Econ. 7, 701–714.
Kummer, D.M., 1992. Deforestation in the Postwar Philippines. Ateneo de Manila University Press, Quezon City, Philippines.
Lasco, R.D., Pulhin, F.B., 2006. Assessing the role of watershed areas in mitigating climate change in the Philippines. The case of the La Mesa Watershed. J. Environ. Sci. Manage. 9, 19–29.
Lasco, R.D., Veridiano, R.K.A., Habito, M., Pulhin, F.B., 2013. Reducing emissions from deforestation and forest degradation plus (REDD+) in the Philippines: will it make a difference in financing forest development? Mitig. Adapt. Strat. Glob. Change 18, 1109–1124.
Laurance, W.F., Camargo, J.L.C., Luizao, R.C.C., Laurance, S.G., Pimm, S.L., Bruna, E.M., et al., 2011. The fate of Amazonian forest fragments: a 32-year investigation. Biol. Conserv. 144, 56–67.
Lele, S., 2009. Watershed services of tropical forests: from hydrology to economic valuation to integrated analysis. Curr. Opin. Environ. Sustain. 1, 148–155.
Liu, D.S., Iverson, L.I., Brown, S., 1993. Rates and pattern of deforestation in the Philippines: application of geographic information system analysis. For. Ecol. Manage. 57, 1–16.
Locatelli, B., Vignola, R., 2009. Managing watershed services of tropical forests and plantations. Can meta-analyses help? For. Ecol. Manage. 258, 1864–1870.
Lu, D., Weng, Q., 2004. Spectral mixture analysis of the urban landscape in Indianapolis with Landsat ETM+ imagery. Photogramm. Eng. Rem. Sens. 70, 1053–1062.

Maina, J., de Moel, H., Zinke, J., Madin, J., McClanahan, T., Vermaat, J.E., 2013. Human deforestation outweighs future climate change impacts of sedimentation on coral reefs. Nat. Commun. 4, 1986.
Malabrigo Jr, P.L., Tiburan Jr, C.L., Galang, M.A., Saizen, I., 2015. Tree diversity at La Mesa Watershed in Luzon, a reforested urban watershed. Asian J. Biodivers. 6, 22–40.
McGarigal, K., Cushman, S.A., Ene, E., 2012. FRAGSTATS v4: Spatial pattern analysis program for categorical and continuous maps. Computer software program produced by the authors at the University of Massachusetts, Amherst.
MEA (Millennium Ecosystem Assessment), 2005. Ecosystems and Human Well-Being: Current State and Trends. Island Press, Washington, DC.
Mitchell, M.G., Suarez-Castro, A.F., Martinez-Harms, M., Maron, M., McAlpine, C., Gaston, K.J., Johansen, K., Rhodes, J.R., 2015. Reframing landscape fragmentation's effects on ecosystem services. Trends Ecol. Evol. 30, 190–198.
MWCI (Manila Water Company Inc.), 2012. Championing Sustainability: 2012 Sustainability Report. MWCI, Quezon City, Philippines.
Nagendra, H., Munroe, D.K., Southworth, J., 2004. From pattern to process: landscape fragmentation and the analysis of land use/land cover change. Agric. Ecosyst. Environ. 101, 111–115.
Pontius Jr., R.G., Millones, M., 2011. Death to Kappa: birth of quantity disagreement and allocation disagreement for accuracy assessment. Int. J. Rem. Sens. 32, 4407–4429.
Pulhin, J.M., Ibabao, R.A., Rola, A.C., Cruz, R.V.O., 2018. Water supply and demand and the drivers of change. In: Rola, A.C., Pulhin, J.M., Hall, R.A. (Eds.), Water Policy in the Philippines: Issues, Initiatives, and Prospects. Springer Nature, Cham, Switzerland, pp. 15–40.
Qin, Y., Gartner, T., 2016. Watershed lost up to 22% of their forests in 14 years. Here's How it Affects Your Water Supply, World Resources Institute.
Reddy, C.S., Saranya, K.R.L., Pasha, S.V., Satish, K.V., Jha, C.S., Diwakar, P.G., et al., 2018. Assessment and monitoring of deforestation and forest fragmentation in South Asia since the 1930s. Global Planet. Change 161, 132–148.
Rolo, V., Olivier, P.I., Pfeifer, M., van Aarde, R.J., 2018. Functional diversity mediates contrasting direct and indirect effects of fragmentation on below- and above-ground carbon stocks of coastal dune forests. For. Ecol. Manage. 407, 174–183.
Rozenstein, O., Karnieli, A., 2011. Comparison of methods for land-use classification incorporating remote sensing and GIS inputs. Appl. Geogr. 31, 533–544.
RP (Republic of the Philippines), 2007. Proclamation No. 1336: Declaring certain parcels of land situated in Quezon City, Caloocan city in the Municipality of Rodriguez, Province of Rizal, as the La Mesa Watershed reservation for purposes of protecting and improving its water quantity, quality and security. Office of the President of the Republic of the Philippines, Manila City, Philippines.
RP (Republic of the Philippines), 2011. Executive Order No. 26: Declaring an Interdepartmental Convergence Initiative for a National Greening Program. Office of the President of the Republic of the Philippines, Manila, Philippines.
Shimada, M., Itoh, T., Motohka, T., Watanabe, M., Shiraishi, T., Thapa, R.B., Lucas, R., 2014. New global forest/non-forest maps from ALOS PALSAR data (2007–2010). Rem. Sens. Environ. 155, 13–31.
Steiner, F., 2008. The Living Landscape: An Ecological Approach to Landscape Planning. Island Press, Washington, Covelo, London.
Taubert, F., Fischer, R., Groeneveld, J., Lehmann, S., Müller, M.S., Rödig, E., et al., 2018. Global patterns of tropical forest fragmentation. Nature 554, 519–522.
TEEB (The Economics of Ecosystems and Biodiversity), 2010. Mainstreaming the Economics of Nature: A Synthesis of the Approach, Conclusions and Recommendations of TEEB. UNEP TEEB Office, Geneva, Switzerland.
Thapa, R.B., Murayama, Y., 2009. Urban mapping, accuracy, & image classification: a comparison of multiple approaches in Tsukuba City, Japan. Appl. Geogr. 29, 135–144.
Tiburan Jr, C.L., Saizen, I., Kobayashi, S., 2012. Spatial relationship and ecosystem management. In: Uy, N., Shaw, R. (Eds.), Ecosystem-based Adaptation. Emerald Group Publishing Limited, Bingley, UK, pp. 197–222.
Thomlinson, J.R., Bolstad, P.V., Cohen, W.B., 1999. Coordinating methodologies for scaling landcover classifications from site-specific to global: steps toward validating global map products. Rem. Sens. Environ. 70, 16–28.
Tobler, W., 1970. A computer movie simulating urban growth in the Detroit region. Econ. Geogr. 46, 234–240.
van Noordwijk, M., Bruijnzeel, S., 2008. Deforestation and the Multiple Functions of Tropical Watersheds. ASB Policy Briefs 08, ASB Partnership.
Wu, J., Jenerette, G.D., Buyantuyev, A., Redman, C.L., 2011. Quantifying spatiotemporal patterns of urbanization: the case of the two fastest growing metropolitan regions in the United States. Ecol. Complex. 8, 1–8.
Xie, Y., Sha, Z., Yu, M., 2008. Remote sensing imagery in vegetation mapping: a review. J. Plant Ecol. 1, 9–23.
Yi, H., Guneralp, B., Filippi, A.M., Kreuter, U.P., Guneralp, I., 2017. Impacts of land change on ecosystem services in the San Antonio River Basin, Texas, from 1984 to 2010. Ecol. Econ. 135, 125–135.
Zhou, P., Huang, J., Pontius Jr., R.G., Hong, H., 2014. Land classification and change intensity analysis in a coastal watershed of Southeast China. Sensors 14, 11640–11658.
Ziegler, A.D., Giambelluca, T.W., Tran, L.T., Vana, T.T., Nullet, M.A., Fox, J., et al., 2004. Hydrological consequences of landscape fragmentation in mountainous northern Vietnam: evidence of accelerated overland flow generation. J. Hydrol. 187, 124–146.